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TECHNICAL GUIDANCE DOCUMENT ON THE USE OF SOCIO-ECONOMIC ANALYSIS IN CHEMICAL RISK MANAGEMENT DECISION MAKING

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TECHNICAL GUIDANCE DOCUMENT ON THE USE OF SOCIO-ECONOMIC ANALYSIS IN CHEMICAL RISK MANAGEMENT DECISION MAKING

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INTER-ORGANIZATION PROGRAMME FOR THE SOUND MANAGEMENT OF CHEMICALS

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The work of the OECD related to risk management is carried out by the Joint Meeting of the Chemicals Committee and the Working Party on Chemicals, Pesticides and Biotechnology, with Secretariat support from the Environmental Health and Safety Division of the Environment Directorate. As part of its work on risk management, the OECD has issued 'status report' monographs on five substances that were, or continue to be, the subject of review: lead, cadmium, mercury, selected brominated flame retardants and methylene chloride. It has also published two volumes of the proceedings of the OECD Cadmium Workshop held in Saltsjöbaden, Sweden, in 1995 and a survey report on methylene chloride, supplementing the information presented in the Risk Reduction Monograph on methylene chloride (see list of publications on page 4). In 1996, OECD Environment Ministers endorsed a Declaration on Risk Reduction for Lead to advance national and co-operative efforts to reduce the risks from lead exposure.

OECD has also published as part of its work on risk management, workshop reports and guidance documents concerning methodologies on non-regulatory initiatives, collection and recycling of nickel-cadmium batteries, sustainable chemistry and socio-economic analysis.

The Environmental Health and Safety Division publishes documents in several different series, including: Testing and Assessment; Good Laboratory Practice and Compliance Monitoring; Pesticides; Risk Management; Harmonization of Regulatory Oversight in Biotechnology; PRTRs (Pollutant Release and Transfer Registers); and Chemical Accidents. More information about the Environmental Health and Safety Programme and EHS publications is available on the OECD's web site.

This publication was produced within the framework of the Inter-Organization Programme for the Sound Management of Chemicals (IOMC).

This publication is available electronically, at no charge.

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The Inter-Organization Programme for the Sound Management of Chemicals (IOMC) was established in 1995 by UNEP, ILO, FAO, WHO, UNIDO, UNITAR and the OECD (the Participating Organizations), following recommendations made by the 1992 UN Conference on Environment and Development to strengthen co-operation and increase international co-ordination in the field of chemical safety. The purpose of the IOMC is to promote co-ordination of the policies and activities pursued by the Participating Organizations, jointly or separately, to achieve the sound management of chemicals in relation to human health and the environment.

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FOREWORD

In January 1998, the OECD sponsored a Workshop on the Integration of Socio-Economic Analysis (SEA) in Chemical Risk Management. One of the recommendations made by Workshop participants concerned the development of a flexible framework for integrating socio-economic analysis in chemical risk management. This document was produced in response to that recommendation.

This is the third in a series of documents produced by the OECD as part of its work on socio-economic analysis. The other two are: *Guidance for Conducting Retrospective Studies on Socio-Economic Analysis* (1999) and *Framework for Integrating Socio-Economic Analysis in Chemical Risk Management Decision Making* (2000).

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The principal authors were Meg Postle (of Risk & Policy Analysts Limited) and Matthew Hickman (who was at Risk & Policy Analysts Limited but has since become an Environmental Economist at Environment Waikato, New Zealand) who extend their thanks to a number of individuals who contributed to the document by authoring particular sections:

- Vic Adamowicz and Peter Boxall, University of Alberta, Canada (Section 3.2.6, Part 3 on attribute-based stated preference methods);
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EXECUTIVE SUMMARY

This document is the third in a series of OECD publications on integrating socio-economic analysis (SEA) into chemical risk management decision making. The first, *Guidance for Conducting Retrospective Studies on Socio-Economic Analysis,*[1] provided guidance for those who have already conducted a SEA and implemented risk management measures, and who wish to determine, for example, whether *ex post* impacts match the *ex ante* prediction made in the SEA, and how important key assumptions affecting the analyses were to the decision and to the actual outcomes of the decision.

The second document, *Framework for Integrating Socio-Economic Analysis in Chemical Risk Management Decision Making*[2] provides a general discussion of the key elements of socio-economic analysis, drawing on the guidelines that already exist in OECD countries and the experience that they have gained in applying them. It has been designed to:

- assist decision makers who use the results of SEA to understand the processes underlying the results;
- help providers of the information used in SEA, including risk analysts, to understand the context in which their information is used; and
- act as a general source of reference for practitioners of SEA and stakeholders interested in the SEA process.

The aim of this *Technical Guidance* document is to further elaborate on the theory and principles underlying the different methodologies used in SEA and on how these are then put into practice, with a focus on the specific requirements of chemical risk management decision making. It has been designed to provide a detailed understanding of the various techniques applied by economists and other policy analysts who are asked to prepare such analyses. This includes information on ‘best practice’ in terms of applying the various techniques, and an indication of how appraisal practices are likely to develop over time.

The discussion draws on the technical guidelines currently being used by a number of OECD countries and the experiences of analysts in these countries. It is hoped that this sharing of information on both techniques and experiences will help foster a more common approach to the assessment of risk management strategies.

It should be noted that while this document provides guidance as to the preparation of SEAs, such analyses will vary in terms of the types of issues that may need to be considered and, hence, in the types of techniques that may need to be called upon and the way those techniques are applied.

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1. *Guidance for Conducting Retrospective Studies on Socio-Economic Analysis*, OECD Series on Risk Management No 11 (1999).
2. *Framework for Integrating Socio-Economic Analysis in Chemical Risk Management Decision Making,* OECD, Series on Risk Management No 13 (2000), Risk & Policy Analysts Ltd. (2000).

The target audience for this document includes all of those practitioners who may be involved in the detail of preparing SEAs within the context of chemical risk management. Although the discussion is focused on economists and policy analysts, much of the material presented here may also be of relevance to those involved in the risk assessment and risk communication stages of the decision making process. Risk assessment usually provides the starting point for SEA and the outputs of such assessments act as inputs to the SEA. Developing a more detailed understanding of how SEA uses these data should help risk assessors in providing the types of information required by economists and other policy analysts. Similarly, if those tasked with communication of the strategy have some appreciation of the techniques that have been applied, they will be better able to address some of the concerns voiced by stakeholders and the general public.

The document has been developed to act as a reference source for practitioners, for use alongside other country-specific guidelines. It has been organised into four parts, to allow users to focus on different levels of detail:

- Part 1: this part provides an introduction to SEA, a general overview of the decision making process (including the linkage to risk assessment as the starting point) and of the types of issues which may arise in preparing a SEA.

- Part 2: this part of the document reviews the main methodological tools which provide the frameworks for SEA. This includes a discussion of the theory and principles underlying each of the methodologies and an indication of how they are applied and the issues that may arise using hypothetical case studies.

- Part 3: this part builds on Part 2 by providing greater detail on the analytical techniques that may be called upon within the above frameworks. It examines in detail the approaches used for assessing specific types of impacts, how these are applied in practice, and the types of data required for their application.

- Part 4: this part then focuses on some of the common issues that arise in SEA, including the management of risk and uncertainty, equity analysis, discounting and presenting the findings of assessments so as to best inform the decision making process.

GLOSSARY

Abatement cost function	The cost of abating a perceived problem, such as pollution or congestion, in the form of an equation/curve. Curves will tend to slope upwards at an increasing rate.
Appraisal	The process of defining objectives, examining options and weighing up the costs, benefits, risks and uncertainties before a decision is made.
Benefits transfer	The method of transferring benefit estimates from past valuation studies to a current study (usually) in order to reduce appraisal costs.
Bequest value	The value placed by people on the continued existence of an asset for the benefit of future generations.
Cardinal	A mathematical term; if one number is double another on a cardinal scale then it follows that there is twice as much of whatever is being measured.
Catastrophic event	That which has a sudden, dramatic and widespread impact upon the environment.
Conjoint analysis	A form of multi-criteria analysis in which people are asked to rate different scenarios according to a variety of criteria. The aim is to find which criteria matter the most.
Consumer surplus	The difference between the amount currently paid for a good or service and the maximum amount that an individual would be willing to pay.
Contingency	An allowance included in the estimated cost of a project to cover unforeseen circumstances.
Contingent valuation	The valuation of environmental change based on estimation of people's willingness to pay (or to accept compensation) for a specified effect elicited *via* questionnaires and statistical techniques.
Cost-benefit analysis	A form of economic analysis in which costs and benefits are converted into money values for comparison over time.
Cost-effectiveness analysis	A form of economic analysis which compares the costs of alternative means of achieving pre-set goals *e.g.* lives saved.
Cost of capital	The cost attributed to money raised for investment, expressed as an annual percentage rate.
Derogation	The exclusion or alternative treatment of those heavily impacted by a policy.

Desktop assessment	An assessment of projects or policies which uses valuation data from past studies to arrive at ball-park figures for (usually) benefits.
Discount rate	The annual percentage rate at which the present value of a future unit of currency (or other unit of accounting) is assumed to fall through time.
Discounting	The comparison of quantities which are distributed over time by converting them to a present value through applying a discount rate.
Dose-response technique	Determines the economic value of environmental changes by estimating the market value of the resulting changes in output.
Economic analysis	Aimed at evaluating all of the effects of a policy or project and valuing them in national resource terms. This takes place in a 'with' and 'without' framework.
Economic appraisal	An umbrella term which includes cost-benefit analysis and cost-effectiveness analysis. Should be contrasted with financial appraisal which is restricted to cash flows.
Evaluation	The retrospective analysis of a project, programme or policy to assess how it has performed, and what lessons can be learned for the future.
Ex ante	Refers to predicted (or expected) values prior to say, the introduction of a risk reduction measure.
Ex post	Refers to actual values as a result of say, the introduction of a risk reduction measure.
Existence value	The value placed by people on the continued existence of an asset for the benefit of present generations.
Expected utility theory	Foundation of decision theory meaning that any course of action with the highest expected utility should be chosen.
Expected value	The sum of individual outcomes times their probability of occurrence.
Exposure	The means by which someone/something is exposed to a hazard.
Externalities	Goods which remain unpriced and thus are external to the market (*i.e.* 'free' goods such as those relating to the environment). An externality is said to exist when the actions of one individual affect the well-being of other individuals without any compensation taking place through the market.
Financial analysis	The determination of the cash flow implications of a policy or project and ensuring that these are sustainable in that sufficient funds are generated to meet outflow.
Free-riding	A phenomenon whereby an individual chooses not to contribute to a good with public characteristics since he/she can gain the benefits from the provisions of others without paying the costs.

General equilibrium	Type of analysis that looks at the economic system as a whole and observes all changes in prices and quantities simultaneously. Usually relies upon complex mathematical techniques.
Gross domestic product	The total output produced in an economy (annually).
Hazard	A situation or activity with the potential for harm.
Hedonic pricing	Technique that infers valuations by using market prices which reflect a range of different criteria, *e.g.* change in environmental quality and property prices.
Input-output model	Type of analysis that examines the inter-relationships between sectors of the economy. Usually represented as a series of linear production functions.
Intergenerational equity	One of the tenets of sustainable development, ensuring fair treatment between different generations.
Irreversible effects	The loss of an irreplaceable environmental feature (*e.g.* an ecosystem or species) and very long-term changes to the natural environment.
Least cost alternative	The lowest cost alternative means of providing the same goods and services as the asset under consideration.
Market price approach	The valuation of physical or qualitative environment or health changes using market prices, for example, changes in fisheries productivity or changes in health care costs as a result of changes in environmental quality.
Multi-criteria analysis	A decision making aid which relies on the use of non-monetary valuation methods (such as scoring and weighting techniques) to aid in problem evaluation.
Net present value	A term used to describe the difference between the present value of a stream of costs and a stream of benefits.
Non-use value	Values which are not related to direct or indirect use of the environment (consists of existence and bequest values).
One-off cost	A cost that is incurred only once throughout the analysis period; an example may be the replacement of capital equipment as a response to a risk reduction measure.
Opportunity cost	Value of a resource in its next best alternative use.
Option appraisal	A term used in some areas to describe any form of appraisal.
Option value	The value of the availability of the option of using an environmental or other asset (which is usually non-marketed) at some future date.
Ordinal	A mathematical term; on an ordinal scale the factors are simply ranked in order, *e.g.* first, second, third, *etc.*

Peer review	Review of an appraisal by experts (in the public, private or academic sectors) to determine if good practice has been followed, quantification is accurate, and so on.
Perfect markets	An economic concept in which firms produce a homogenous product using identical production processes and buyers and sellers possess perfect information. There is also a large number of buyers and sellers and freedom of entry to the market ensuring that normal profits are being earned.
Present value	The discounted value of a stream of future costs or benefits.
Public good	Where the provision of the good for one individual necessarily makes it available to others, and where it is not possible to prevent others from using it.
Quality Adjusted Life Year	A measure of health status in terms of the quality of life associated with a state of health, and the number of years during which that health status is enjoyed.
Real price	The nominal (*i.e.* cash) price deflated by a general price index or GDP deflator relative to a specified base year or base date.
Real terms	The value of expenditure at a specified general price level: that is, a cash price or expenditure divided by a general price index.
Residual value	The expected value of a capital asset at the end of the analysis or at some future date.
Resource costs	The cost of marketed goods or services (adjusted to economic prices) used as inputs to, or consumed as a consequence of, an action.
Revealed preference	Willingness to pay for something which is non-marketed, as revealed by other expenditure choices.
Risk	The likelihood of a specified (adverse) consequence.
Sensitivity analysis	The analysis of how an appraisal will be affected by varying the projected values of the important variables.
Shadow price	The price that represents the opportunity cost to society of engaging in some economic activity (such as producing or consuming goods).
Socio-economic impacts	Any impacts upon society/the economy as a result of a policy or project, such as price changes, welfare changes, employment, reduction in health impacts, and so on.
Stakeholder	Any interested or affected party with regard to a policy or project.
Stated preference	The willingness to pay for something which is non-marketed, as derived from responses to questions about preference for various combinations of situations and/or controlled discussion groups.

Stocks and flows	Stocks are goods which will experience a one-off change in value. Flows occur annually and stem from the stock good (*e.g.* agricultural yields per annum).
Switching point/value	The value of an uncertain cost or benefit where the best way to proceed would switch, *i.e.* from including to excluding some extra expenditure to preserve some environmental benefit.
Time preference rate	Preference for consumption (or other costs or benefits) sooner rather than later, expressed as an annual percentage rate.
Total economic value	The sum of use values (direct, indirect and option) plus non-use values (bequest and existence).
Uncertainty	Stems from a lack of information, scientific knowledge or ignorance and is characteristic of all predictive assessments.
Use value	The value of something which is non-marketed (such as the environment or good health) generated by people's actual use of it.
Value of a statistical life	Collective willingness to pay to reduce mortality risk by those at risk (aggregated up to the level where one fewer death would be expected).
Welfare cost/benefit	Any effect on human well-being.

ACRONYMS

ABSPM	Attribute-based Stated Preference Method
AC	Average Costs
AHP	Analytic Hierarchy Process
B/C	Benefit-cost Ratio
CBA	Cost-benefit Analysis
CEA	Cost-effectiveness Analysis
CERCLA	Comprehensive Environmental Response, Compensation and Liability Act, US
C/E	Cost-effectiveness Ratio
CGE	Computational General Equilibrium
CRI	Consumption of Rate of Interest
COI	Cost of Illness
C/U	Cost-utility Ratio
CVM	Contingent Valuation Method
DALY	Disability Adjusted Life Year
DETR	Department of the Environment, Transport and the Regions, UK
EC	European Commission
EPA	Environmental Protection Agency, US
EVRI	Environmental Valuation Research Inventory
GDP	Gross Domestic Product
GE	General Equilibrium
HALY	Health Adjusted Life Year
HPM	Hedonic Pricing Method
HRQL	Health Related Quality of Life

HSE	Health & Safety Executive, UK
HYE	Healthy Years Equivalent
IIA	Independence of Irrelevant Alternatives
I-O	Input-Output
MC	Marginal Costs
MCA	Multi-criteria Analysis
MNL	Multinominal Logit
NOEL	No Observable Effect Level
NPV	Net Present Value
OCC	Opportunity Cost of Capital
OECD	Organisation for Economic Co-operation and Development
OMB	Office of Management and Budget, US
PDF	Probability Density Function
PEC	Predicted Environmental Concentration
PNEC	Predicted No Effect Concentration
PTO	Person Trade-off
PV	Present Value
QALY	Quality Adjusted Life Years
QWB	Quality of Well Being
RPA	Risk & Policy Analysts Ltd
RUM	Random Utility Model
SEA	Socio-economic Analysis
SG	Standard Gamble
SMART	Specific, Measurable, Agreed, Realistic, Time-dependent
SOC	Social Opportunity Cost of Capital
STPR	Social Time Preference Rate of Discount

TC	Total Costs
TCM	Travel Cost Method
TEV	Total Economic Value
TTO	Time Trade-off
UK	United Kingdom
US	United States of America
VAS	Visual Analog Scale
VLYL	Value of Life Years Lost
VOI	Value of Information
VSL	Value of a Statistical Life
WTA	Willingness to accept
WTP	Willingness to pay
ZOZ	Imaginary chemical providing basis for case study

WEB SITES OF INTEREST

Note: This is a general list that may be useful to those undertaking SEAs. Due to the number involved, it does not list each specific academic institution undertaking research in this area. Instead, the list should be viewed as a good starting point in the subject area of economics and environment/health effects.

OECD	www.oecd.org
European Commission	www.europa.eu.int
US EPA	www.epa.gov
US EPA Economics	www.epa.gov/economics
Environment Canada	www.ec.gc.ca
Health Canada	www.hc-sc.gc.ca
UK Government Departments	www.open.gov.uk
Governments on the WWW	www.gksoft.com/govt/en
EVRI database	www.evri-ec.gc.ca/evri
Resources for the Future	www.rff.org
Association of Environmental and Resource Economists	www.aere.org
European Association of Environmental Economists	www.eaere.org
WWW Resources in Economics	www.netec.mcc.ac.uk/WebEc.html
Resources for Economists on the Internet	www.rfe.wustl.edu
Online Journals (including JEEM)	www.idealibray.com
Chemical Industry Search Engine	www.chemindustry.com

PART 1: CONDUCTING A SOCIO-ECONOMIC ANALYSIS

1. INTRODUCTION TO SOCIO-ECONOMIC ANALYSIS: WHAT IS SOCIO-ECONOMIC ANALYSIS?

Decision making is about making choices as to the best way forward. Such choices can be made using a variety of decision criteria and taking into account a range of different information. As a general rule, decision makers will want information on the impacts of choosing one course of action over another. This includes information on the impacts to industry, regulators, consumers, the environment and society more generally. Within the field of chemical risk management, relevant information will include details of, for example:

- the nature and characteristics of the risks of concern;
- the types of regulatory and non-regulatory measures that could be adopted to reduce or mitigate damage;
- the costs of risk reduction and their distribution, where this includes costs to industry, consumers, regulators and society more generally;
- the benefits of risk reduction and the distribution of these, where these may relate to environmental and human health gains or to increased technical/product innovation; and
- the wider trade, competition, and economic development implications of adopting a change in policy.[3]

The theory underlying how such information is developed and how the tools are used in analysing the information falls under the general heading of socio-economic analysis (SEA). The aim of socio-economic analysis within the field of chemical risk management is to assist the decision making process by making explicit the implications of choosing one risk management option over another. By having information on the trade-offs associated with different options, decision makers are able to make a more informed choice as to the best way forward.

A range of different methodological tools may be used within a SEA. Commonly used methodologies (described in Part 2) are cost-benefit analysis (CBA) and cost-effectiveness analysis (CEA), with some forms of multi-criteria analysis (MCA) also applied by OECD countries. These methodologies themselves call upon the use of a number of different analytical techniques. For example, preparation of a cost-benefit analysis may require that a number of different valuation techniques are applied, and will involve the use of discounting procedures.

[3] Given the increasingly global nature of the world's economy, the chemicals industry and chemical risk management, it may be important to extend an analysis beyond national frontiers to consider the implications of risk management decisions for developing countries and their future economic development.

2. PREPARING FOR THE SOCIO-ECONOMIC ANALYSIS

2.1 Overview of the Decision Making Process

The findings of a risk assessment will usually provide the starting point for the SEA. In this regard, decision making on risk issues relates to one of three levels of risk:

- a level of risk that is so low as to be regarded as trivial, with this being referred to as an 'acceptable' or 'negligible' or '*de minimis*' risk;

- a level of risk that is so high that there is general agreement that the risks are 'unacceptable' or 'intolerable' (the '*de manifestis*' level of risk); and

- a level of risk between these extremes, where there is greater uncertainty as to the benefits of risk reduction and agreement is lacking as to the need for action, the most appropriate form of action, or resource availability.

It is decision making with regard to the latter two levels of risk that is the main focus of the SEA tools set out in this document. For example, SEA can assist in developing a regulation from the first proposal stage, identifying new and alternative options for risk management, eliminating options that would not be cost-effective or acceptable for other reasons, adjusting options to account for differences between industries or industry segments, and supporting end decisions.

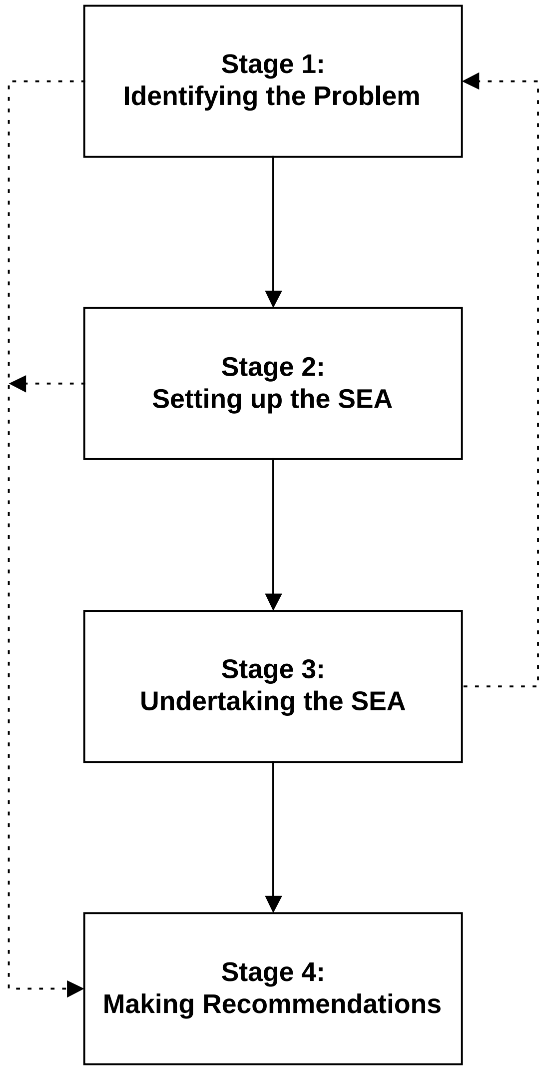
The type of approach to SEA that may be applied within the above contexts vary among OECD countries. There is broad agreement, however, on the need for a systematic approach to such analyses to make explicit the implications of adopting a particular risk management action. By being systematic, the SEA should enable decision makers to understand many of the implications of risk management, to determine the adequacy of the information available to them and to identify any important gaps.

Figure 2.1 provides a generalised framework outlining the role that SEA can play in chemical risk management decision making. This example framework has four key stages:

- Stage 1: Identifying the Problem - this step relates to a risk issue entering onto the decision making agenda and the factors that give rise to the need to consider risk management (for example, as a result of risk characterisation in terms of the potential acceptability or unacceptability of the level of risk);

- Stage 2: Setting up the SEA - the objectives of risk management are elaborated in the second step, with this also entailing the identification and involvement of the relevant stakeholders and integrating the efforts of economists, risk assessors and others whose work will inform the decision;

Figure 2.1: A Generalised SEA Framework



- Stage 3: Undertaking the SEA - the analysis will require identification of alternative risk management options, the collation of data on impacts associated with the various alternatives, and assessment of the predicted negative and positive costs and negative and positive benefits; again this is likely to involve input from stakeholders; and

- Stage 4: Making Recommendations - this final step includes the comparative analysis of the alternative options, peer or expert review of analysis results and involvement of stakeholders in order to provide a comprehensive set of recommendations to decision makers; this step also incorporates implementation of any decision and monitoring its success.

The framework presented in Figure 2.1 is obviously a simplified one, with each of the Stages comprising a range of different activities. The Annex to this Part provides a series of Figures that sets out in more detail these activities and their relationship to the overall decision making process. A brief overview of the key analytical activities within each of the Stages is presented below.[4]

2.2 Stage 1: Identifying the Problem

Risk issues may enter onto a decision maker's agenda through a number of different avenues, including: legislative requirements, previous government policy decisions, public concerns raised by the media, experts, interest group pressure, the availability of new scientific information or the availability of new technologies. How an issue arises is likely to affect the way in which risk management is approached. When a problem arises owing to legislative requirements or other government decisions, then many of the characteristics of the risk issue may be well understood, together with the potential options for addressing it. When instead, issues arise as a result of pressure from the media, interest groups or selected experts, then the characteristics of the issue may be less clear, with more work required to fully define the problem. On the other hand, the availability of new scientific information or of new technologies may make it important to re-examine risk issues that have been addressed in the past.

In any event, this first stage of problem identification may be particularly challenging for chemical risks, owing to the variety of uses to which a chemical may be put, uncertainty over a chemical's potential for harm, a lack of hazard and exposure data, and potential synergies and antagonisms with other chemicals. Problem identification is likely to include the following activities:

- identifying the potential risk generating activities for the substance or uses of the substance of concern;

- collecting preliminary data on the societal value of the substance or uses of concern, including current market levels and trends in usage, and the availability of substitute chemicals or processes;

- identifying why policy intervention may be necessary; for example, whether it will be necessary to correct market failures (*e.g.* there being no cost penalty for discharging a chemical to the environment) or to correct distortions introduced into markets as a result of government policy (*e.g.* when a policy aimed at protecting the environment leads to changes in health risks);

- undertaking a screening level risk assessment to determine if further, more detailed analysis is required; and

- where further analysis is required, preparing a detailed risk assessment (according to national and/or international requirements).

These activities will usually be carried out as part of the risk assessment, but it is likely that those responsible for undertaking the SEA will also want to be involved. In particular, co-operation on the collection of data on the market for the substance and the availability of substitutes is likely to be of value to both the risk assessors and those preparing the SEA. In contrast, questions concerning why regulatory action to reduce risks may be necessary will be of more concern to those preparing the SEA. In this regard, analysts will want to establish why current regulations are not adequate, why market forces alone

[4] For a more detailed discussion of this framework, see also the *Framework for Integrating Socio-Economic Analysis in Chemical Risk Management Decision Making* (OECD Series on Risk Management, No. 13).

are insufficient to correct the problem, and the degree to which other government policies have influenced the risk generating activities (*e.g.* distorted certain markets thus inducing use of a particular chemical).

The findings from this stage provide what is often the starting point for the SEA and may give an indication of potential risk management options or lead to other decisions regarding the appropriate approach to risk management. Where the risk assessment and SEA are conducted in parallel, the decision-maker, risk analyst and socio-economic analyst may jointly develop options to be considered in both assessments. In addition, as work progresses on the SEA, it may be important to revisit some of the assumptions made in the risk assessment (for example, the potential for chemical exposure may change as assumptions are refined).

2.3 Stage 2: Setting up the SEA

Once a decision has been made to proceed with a SEA, the next phase of work involves establishing the framework for the analysis. This involves:

- defining the risk management objectives underlying the SEA;
- undertaking initial data collection to allow a screening assessment of the options; in some cases, the screening exercise may identify a risk management option which is acceptable to all stakeholders, while in other cases it will provide the basis for focusing the more detailed analysis (Stage 3); and
- developing specifications for the more detailed SEA.

A further activity that is likely to be undertaken within this stage is identifying the relevant stakeholders and determining how they are to be involved in the risk management process, including what type of involvement is expected when developing the SEA. Given the importance of this aspect to SEA, it is addressed separately below in Section 3.

Setting the Objectives

The findings of the risk assessment will be useful in establishing what should be examined in the SEA. The objectives of the SEA itself should then be specified. At the highest level, the goal is to provide information on the costs and benefits of the alternative chemical risk management options. How this goal is to be achieved is likely to vary, depending on the factors driving the SEA. In some OECD countries, there will be legal requirements for a particular form of analysis, while in others such requirements may not exist or may be much more general in nature.

More specific objectives are also likely to be required, provided an indication is given as to the expected scope of the analysis and the nature of the outputs required from it. The manner in which these objectives are defined is likely to be vital to the success of the SEA. The UK Health & Safety Executive and the UK Treasury suggest that objectives should be SMART (HSE, 1995):

- Specific;
- Measurable;
- Agreed;

- Realistic; and
- Time-dependent.

To the degree possible, the objectives should also accurately state the intentions or commitment of the commissioning agency (and the final decision makers) and reflect stakeholder concerns and the level of agreement between all stakeholders regarding the objectives.

Identifying Options

For any risk management issue, there is likely to be a number of different options that can meet decision maker's objectives. In this early stage, it is important that a wide range of potential regulatory and non-regulatory options is considered to avoid the premature rejection of viable options. For example, the types of options that may be important in the chemical risk context are:[5]

- voluntary approaches, such as product stewardship and/or environmentally friendly product design, or the setting in ISO or other standards;
- employee health and safety programmes;
- information tools, such as classification and labelling, and risk communication, environmental education and public awareness programmes;
- market based tools (economic instruments) such as emission/effluent charges, product taxes, tradable permit systems and subsidies aimed at shifting producer, user and/or consumer behaviour; and
- command and control tools, such as marketing and use restrictions, emission or discharge limits (for example, to meet environmental quality standards), and best available technology requirements.

The level of risk posed by a particular chemical will depend on a range of factors, including its hazard potential, the nature of its use (or uses) and the degree of exposure that results from that use. Risk management activities need to be correctly targeted to ensure that they are addressing the real risks of concern, which may relate to particular applications rather than the use of the chemical in its entirety across all uses. It may be important, therefore, to consider options for controlling risks arising from different stages within a chemical's life cycle. For example, if the risks are related to manufacturing activities then command and control tools or voluntary agreements may be appropriate, as may market based tools (economic instruments). If the concern is over risks from product use, then health and safety programmes or classification and labelling may be appropriate. Linking options to the relevant life-cycle stages should help ensure that risk reduction is properly targeted and that an appropriate balance is achieved between risks, costs and benefits.

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5 For a more detailed discussion of the types of options available see: EC, 1998; OECD, 1991; OECD, 1994. In addition, the guide produced by the Treasury Board of Canada Secretariat titled *Assessing Regulatory Alternatives* provides a question and answer approach to identifying possible regulatory and non-regulatory forms of policy intervention.

In addition to what life-cycle stage an option might apply, it might also be important to consider whether it provides a short- or longer-term solution to the risk problem. For example, command and control measures and voluntary agreements may provide only short- to medium-term solutions, while measures which change consumer behaviour (*e.g.* some economic instruments and information/education tools) may provide longer-term solutions.

Once the range of potential risk management options has been identified, the next step is to collect the preliminary data on each to enable a simple screening exercise to take place. The aim of such an exercise is to provide general information on aspects such as technical feasibility, effectiveness, economic impacts, change in environmental and health risks, and likely stakeholder acceptability. In a few cases, screening may indicate that one option performs better than the others against the various criteria, delivering an acceptable balance between negative and positive costs and negative and positive benefits. Where there is general agreement to the adoption of an option (taking into account the limited accuracy or prediction capability of such exercises), it may be the case that no further analysis is required (unless there are legal requirements or other drivers underlying the need for a more detailed appraisal). However, it must be recognised that this does not constitute a proper SEA.

In other cases, the information developed through the screening should make it possible to reduce the initial, wide-ranging set of options to a smaller number for more detailed analysis. In so doing, however, it should be remembered that options that are feasible and perform better than all of the other options on at least one important criterion should not be eliminated at this stage. They should be carried forward to the next stage to allow a proper examination of their relative advantages and disadvantages.

Specifying the Detailed SEA

The outcome of the screening exercise will be valuable information for determining what is required of any more detailed assessment (although as indicated above it may also be possible to make risk management recommendations on the basis of the screening results). In this regard, the exercise should inform:

– any refinements to the objectives of the SEA;

– the manner in which stakeholders are to be involved in the detailed SEA, where this includes consideration of both who and how;

– the selection of the appraisal methodology(ies) for the analysis;

– the timing of key deliverables from the analysis; and

– any key issues that may need to be given extra attention within the analysis.

2.4 Stage 3: Undertaking the SEA

The third stage within the overall process involves preparation of the detailed SEA. This will include:

– detailed specification of the key risk management options;

– selection of the appraisal methodology(ies) to be used within the SEA;

- further data collection activities;
- analysis of the data; and
- comparative appraisal of the options.

It is likely that there will be some iteration both within this stage and between this stage and some of the activities comprising Stage 2. In particular, data collection may lead to the identification of new risk reduction options. As the analysis of alternative options proceeds, it is likely that new issues will arise, leading to further data collection. Similarly, these new issues may lead to the need to re-define the risk issues and repeat some of the Stage 1 activities such as revisiting the need for intervention and screening level risk assessments.

Identification of Key Risk Management Options

It will be important to ensure that the options are defined well enough so that the key parameters affecting the assessment of costs, risks and benefits are understood. Such parameters may include the timing of specific actions, the risk generating activities that would be affected, any sub-options available in terms of how an option could be implemented, and any restrictions that may be placed on the way in which the risk generators responded to an option. The aim in specifying these details is not to reduce the flexibility open to industry or others in responding to a proposed option. Instead, it is hoped that, by specifying such parameters at the start of the detailed analysis, this will help ensure that everyone involved in the appraisal has a shared understanding of what the different options would entail. This should help not only in data collection, but also in identifying possible variations in options that may have different cost, risk or benefit outcomes.

Selecting the SEA Methodology

In selecting the methodology(ies) that will provide the basis for the SEA, a number of factors should be considered:

- the regulatory context within which the SEA is being undertaken and whether this suggests or requires the use of a certain methodology or set of techniques (*e.g.* cost-benefit analysis is implicitly required under some US legislation);
- the stated objectives of the SEA and the requirements of decision makers with regard to having quantitative versus qualitative information;
- the number of negative and positive costs and negative and positive benefits of concern; and whether any specific health or environmental targets or thresholds have to be met for an option to be considered feasible or acceptable;
- the nature of the information available from the risk assessment (whether a full consequence analysis or more limited information on hazard or risk potential);
- the period of time and resources (staff and money) available to the analyst (bearing in mind that both are limited and that the analysis should be proportionate to the characteristics of the decision); and

– the capability of the methods under consideration to the support decision making process.

Also of key importance in selecting the methodology is understanding the questions that the different methodologies should answer. The types of questions that decision makers may want answered include: How well do the options perform in terms of economic efficiency? What are the trade-offs associated with choosing one option over the others? Are there any equity (distributional) issues of concern? What will the impacts of the option be on the health of the industry (or market) or on the economy more generally? Is the industry very competitive - nationally and/or internationally?

A wide range of different issues, therefore, may need to be taken into account, with these being dependent upon the varying interests of the stakeholders and any differences that they may have in their priorities for risk management. The greater the differences in stakeholder positions, the more important it will be to adopt an approach that highlights the trade-offs in meeting one set of criteria over another.

Depending on the requirements, a SEA may take one of three possible forms:

– a systematic qualitative analysis, where the magnitude, significance and relative importance of the risks, negative and positive costs and negative and positive benefits are described but not quantified;

– a semi-quantitative analysis, where some aspects of the risks, negative and positive costs and negative and positive benefits are assessed in quantitative terms while others are treated qualitatively; or

– a fully quantitative analysis, where all risks, negative and positive costs and negative and positive benefits are quantified in physical/natural units and/or, in some cases, in monetary terms.

A qualitative analysis may be sufficient where there are readily affordable solutions and there is common agreement that risk management is required. In other cases, such analyses may not provide enough detail to indicate trade-offs clearly nor determine whether the benefits from risk management outweigh the costs. As a result, a more quantitative analysis may be required. However, it may not be possible to undertake a quantitative analysis owing to a lack of data. Preparing a qualitative analysis in these instances will remain important because having some information will be better than having no information.

In general, the more quantitative the approach, the more informative the analysis is likely to be, yet also the more resource-intensive. Any analysis will inevitably involve management of uncertainty and will require informed, professional judgements to be made. As a result, achieving a balance between the thoroughness of the analysis and practical limits to carrying out an analysis is essential (US Office of Management and Budget, 1997). Indeed, most chemical risk management SEAs will involve combining both qualitative and quantitative information on impacts, as it may not always be possible to value the full range of impacts.

Research undertaken for the Nordic Council of Ministers (Hokkanen & Pellinen, 1997) promotes a step-by-step approach to SEA, starting with the application of qualitative assessment techniques. Semi-quantitative or more fully quantitative techniques are then applied as warranted by the magnitude of the trade-offs involved in selecting one course of action over another. The research goes further to suggest that for many chemical risk management problems, given the complexity of the decisions and the number of factors that need to be taken into account, the combined use of a number of approaches may prove the most valuable.

The various methodological approaches that can assist in preparing either qualitative or more quantitative assessments are discussed further in Section 4 below (and in Parts 2 and 3).

Detailed Data Collection

The methodology that is used in preparing the SEA will, to a large extent, determine the type and form (whether qualitative, quantified in physical terms or related to monetary values) of the data required at this stage. In many cases, a wide range of data is likely to be required, for example:

- information on the number of companies using a substance, levels of use, and expected trends in use;

- details of the implications of the proposed option in terms of any changes required to existing processes and/or end-products (technologies used, chemicals used, level of treatment, product quality and/or availability, *etc.*), reporting, monitoring, enforcement or other requirements;

- data on the capital and/or recurrent costs (and/or savings) associated with the introduction of a proposed option;

- information on rates of and potential for technological change for the sector of concern; and

- predictions of effects, in terms of human health and environmental risks.

For any particular assessment, some of the above data may not be available either because they do not exist or because they are not readily accessible to the analyst. In the latter case, where stakeholders have been involved from an early stage in the risk management process, this may increase access to the data that they hold. It does not necessarily follow, however, that the ready availability of extensive quantities of data means that a highly detailed analysis should be carried out. Instead, the scope and nature of the analysis should be determined by the characteristics of the risk management decision.

It may also be important for data collection to be extended beyond national/regional boundaries. Decisions concerning the management of both inorganic and organic chemicals are increasingly international in effect. As a result, actions taken at the national level can raise significant economic development issues for other countries. For example, mining is increasingly becoming a developing country activity, and such economies may be affected by the manner in which developed countries choose to manage the risks associated with mined materials. It may, therefore, be important for appraisals to examine the implications of proposed policy options for these wider stakeholders.

Data Analysis and Appraisal of Options

Analysis of the above data will generate information on the predicted impacts of adopting alternative risk management options. For most chemical risk decisions, it is likely that the analysis will be expected to produce information on:

- the costs (or savings) to industry/business (usually broken down by sector or type of activity) of adopting a given option, taking into account any one-off costs (*i.e.* capital costs of new plant) and changes in operating/revenue costs;

- impacts on trade and the competitiveness of industry, across the relevant supply chain (*e.g.* from raw materials producers to formulators to secondary manufacturers);

- impacts on small and medium sized enterprises, where these may vary from those on larger companies;

- impacts on consumers in terms of increased product prices, changes in quality of end-products, reduced availability of particular products, *etc.*;

- impacts on regulators associated with changes in regulations and their monitoring and enforcement;

- predictions of the environmental and human health benefits (and costs) arising from an option; and

- predictions of indirect or secondary effects related to changes in employment or impacts on other sectors of the economy.

It should be possible in most cases to develop rough estimates at least of the compliance costs (with these acting as a proxy for the social costs) involved in adopting and implementing any particular risk management option, where this includes the costs to industry and regulators (and potentially consumers). Similarly, the outputs of the risk assessment can provide information on the potential change in human health and environmental risks, where these represent the benefits arising from risk management.

However, the methodological framework chosen for the SEA will determine the manner in which the data are analysed and, thus, the way in which the alternative options are comparatively assessed. For example, where cost-benefit analysis provides the analytical framework, then the aim is to compare estimates of the social costs of risk management with the social benefits to determine whether a particular option delivers net benefits (in economic efficiency terms). If data are available on the probability of risk outcomes, it may be possible to place a monetary value on the changes in environmental or human health risks. Where such valuation is feasible, the expression of safety benefits in the same units (money) as the costs of control allows the direct comparison of the trade-offs associated with alternative options. The manner in which the different methodologies treat data and thus allow for the comparison of options is discussed further in Part 2 of this document.

In some countries, analysis of the distributional impacts of proposed options is also required, with such analyses considering impacts across both different industry sectors and sizes of companies and different stakeholder groups (with stakeholders potentially being defined to account for any impacts arising at the international level). The aim of such analyses is to help ensure that any significant equity considerations are brought into the decision making process, including possible impacts on future generations (see Part 3 for a more detailed discussion on such analyses). On the basis of this analysis and the information on costs and benefits, it should be possible to identify complementary measures for mitigating or minimising significant negative effects on particular stakeholders.

2.5 Stage 4: Presenting Results and Making Recommendations

As the aim of SEA is to inform the decision making process, the final step requires that information is brought together in a form that communicates effectively and assists decision makers, stakeholders and the general public in understanding trade-offs. The activities involved in this stage are likely to include:

- presentation of information in a clear format, which allows the various parties to understand and interpret the results;
- submitting the analysis to a peer review, undertaken by government reviewers, external experts or others; and
- formulating conclusions and, as appropriate, recommendations (for example, on further analysis requirements).

Presenting Information

Decision makers, stakeholders and the general public need the results of the analysis to be presented in a clear and concise manner, stating assumptions, data sources and any uncertainties contained within them. It is essential, therefore, that both the analysis and any conclusions reached are transparent. Not only will this help ensure that the results are correctly interpreted, but also that users of the results have confidence in them and are able to understand whether there are any significant gaps in the analysis.

Obviously, the manner in which the results are reported will depend on the methodology(ies) that has formed the basis for the analysis. For all analyses, however, it will be important to provide a comprehensive overview of the findings and to present a summary of the trade-offs associated with adopting one option, or mix of options, over another, including:

- the associated risks, benefits and costs for each option, or mix of options;
- the risks associated with the use of substitutes, where these may be technologies/techniques or chemicals;
- the key parameters affecting the decision, in particular the key uncertainties and the sensitivity of the end results; and
- the relative impacts on different groups from the different risk management options.

A range of different presentation formats can be adopted as a means of conveying information on trade-offs, including the use of qualitative descriptions, matrices, charts and other quantitative data. See also Section 5 of Part 4 for further discussion on the presentation of results.

Peer Review

Peer review of SEAs may be important to ensuring that the analysis is robust and will stand up to external scrutiny. The aim of such reviews is to validate both the data and assumptions used within an analysis, and the manner in which those data are analysed. As one might expect, different countries adopt different approaches to the peer review process:

- other government agencies with specialists in particular fields *(e.g.* economics or eco-toxicology) may act as peer reviewers of either part or all of an analysis;
- external specialists may be asked to peer review an analysis prior to its more public release; and/or

– internal specialists not directly involved in the analysis may be asked to undertake a peer review prior to going to other government agencies or external specialists.

A peer review is likely to be of most value in those cases where there are significant trade-offs involved in the choice of risk management options, where the analysis has been particularly difficult, where there are a number of key uncertainties affecting the results, or where new/novel methodologies have been used. In particular, when the costs of risk management are expected to be high and may have significant impacts on a particular industry sector, a number of sectors or the economy more generally, it is likely to be an important part of the process. In such cases, external reviewers should be selected so as to provide specific technical expertise relevant to the issues at hand.

Formulating Risk Management Recommendations

The results of the SEA will form only one set of data to be taken into account by decision makers. The SEA, therefore, does not make the decision but informs it. As a result, it may be appropriate for the SEA to make a number of different recommendations for consideration by decision-makers (with what is considered appropriate obviously varying across different national and international decision making contexts).

Such recommendations may take many forms concerning the need for further research and analysis, the timing of the introduction of a risk reduction option, the examination of new options or the implications (merits and demerits) of exempting certain companies or sectors from a proposed option. In general, such recommendations should take into account any feedback provided by the peer review and/or through a final consultation with stakeholders.

It may also be appropriate for the recommendations to touch on issues other than chemical control. For example, in cases where the public's perception of risks does not reflect actual risk levels, the conclusions may include suggestions on measures aimed at improving risk communication.

3. STAKEHOLDER INVOLVEMENT

3.1 Introduction

The framework presented above is obviously a simplified one. It does not highlight, for example, the need for dialogue between decision makers, risk assessors and socio-economic analysts throughout the risk management process. There will be several stages throughout the process where discussion amongst these parties is important to ensuring that the overall process is effective and that the decision makers' requirements will be addressed.

Stakeholder involvement in the risk management process should help improve the quality of decision making and it may also help avoid damaging and time-consuming confrontations later on in the decision making process (although involvement is not a guarantee that such confrontations and challenges to decisions will not take place). Clearly the extent of such stakeholder involvement will depend upon a range of factors, including the nature of the decision, the time scale and the overall decision making approach. The approach taken will also vary depending on the cultural differences among OECD countries; an approach that has proved successful in one country may not be applicable in another.

Stakeholders may also be involved directly in the SEA, for example, through the provision of data or by participating in a 'steering group' overseeing progress of the analysis itself. In this context, effective stakeholder involvement will require a clear understanding of:

- content: what the issues are that need to be addressed by the analysis;
- outcomes: what the analysis is intended to achieve;
- people: who should be involved, how they should be involved and when; and
- process: how can those involved contribute so as to help achieve the objectives.

3.2 Whom to Consult?

Stakeholders are those organisations or individuals who will be impacted in some way by a decision. Any decision on chemical risk management is likely to affect a wide range of stakeholders. Some of these, for example industry, may face significant costs as a result of risk management decisions, whilst the benefits may be distributed more widely across different groups. Conversely, failing to manage risks may result in a continuation of current impacts on the environment or people, with only a small group benefiting from inaction.

The Environment Council[6] (1999), which reflects both private and public sector interests in the UK, notes that it is essential to identify:

- participants who must be involved if the outcome is to have credibility;
- participants who would bring particular expertise or creativity;
- people who could represent those who might otherwise be excluded; and
- people who might oppose the outcome if they are not included.

The stakeholders that meet these criteria will vary from decision to decision. The first step is therefore to identify key stakeholders. These are likely to include government departments at various levels, employees and labour organisations, organisations such as environmental, consumer and human rights groups, representatives of any ethnic or cultural interests, representatives of regional and local government, as well as the industry sectors that may be affected.[7] It may be particularly important for stakeholders to be defined at the regional and international levels for chemical risk issues as production and consumption often take place at different locations. This will help ensure that the wider national/global implications of a proposed action are understood (and in particular for the economies of developing countries).

Different OECD countries appear to include different groups in stakeholder involvement. In Canada, industry, aboriginal groups and non-governmental organisations are the key stakeholders included in the Options Evaluation Process. In the development of risk management policy in the UK, advisory groups are formed for specific analyses, which consist of government departments and external interest groups, with a wider Stakeholder Forum being in place to prioritise chemical risk management issues. These approaches are similar to that adopted in Finland, where specialist advisory committees, with representatives from government, industry, trade associations and labour organisations, are commonly used to support decision making. In contrast, the Danish government relies on a 1,500-strong business panel, co-ordinated by an agency within the Trade and Industry Department, to consult on the impacts of regulation.

Particular care may be needed where stakeholders involve those outside organised groupings. For example, involving stakeholders beyond activists and pressure groups may be difficult as they may lack the financial resources and technical/scientific expertise required to provide the input sought. In order to ensure effective participation, such issues will need to be identified and addressed at an early stage.

3.3 When to Consult?

The timing of stakeholder involvement will be a critical factor in determining whether stakeholders can effectively participate in the SEA, and the degree of that participation. Sufficient time needs to made available to incorporate the needs of non-governmental organisations that may meet

6 The Environment Council is a registered UK charity. Its activities include promotion of the message that ‘environmental sense is commercial sense’ through publications, seminars and workshops, and training courses. It is supported by a range of public sector organisations (regulatory agencies, government departments, educational institutions) and over 800 private companies.

7 For example, the World Business Council for Sustainable Development (1999) identifies the following stakeholders: shareholders and investors; employees; customers/consumers; suppliers; communities; and legislators.

infrequently, or for stakeholder groups not represented by established organisations to develop a means to participate. Similarly, where stakeholders are asked to contribute data to the analysis, a reasonable amount of time needs to be allowed for them to do this given that this will pull resources away from other activities.

Where the time available for input is limited, stakeholders may either be unable to provide the desired inputs to the analysis or their involvement may be restricted to a post-analysis consultative role. Many interest groups meet irregularly and may not have teams in place capable of collating data and /or reviewing a document before the required date.

Consideration needs to be given to the involvement of stakeholders in the early stages of planning the SEA. Initial discussions with potential stakeholders during the planning process may assist in identifying potential timing problems, which can then be addressed. It may also help induce the involvement of those who would otherwise work against the process (by trying to delay any decision making unnecessarily), as may giving them an explicit role in providing input to the SEA.

3.3 What Type of Input?

Once the key stakeholders have been identified, the next stage is to determine what form of input is most appropriate to the decision and what methods can be used to achieve the objectives of stakeholder involvement. Again, the selection of methods will need to take account of the nature of the decision, the abilities of stakeholders and the cultural background to ensure that useful contributions are not excluded.

The nature of stakeholder input to decision making on chemical risk management can vary widely, for example, it can consist of:

- providing data for the analysis;
- preparing some component of the analysis;
- considering the acceptability of different levels of risk;
- providing information on preferences between different types of negative and positive costs and negative and positive benefits;
- commenting on distributional and equity issues; and
- participating in the formulation of the outputs, thus adding credibility to the final decision taken.

Regardless of the assessment methodology forming the basis for the SEA, stakeholders are likely to be in a position to provide data that are valuable to the decision but otherwise difficult to obtain. Other government departments and local authorities may provide unpublished data while trade associations, interest groups, employers' organisations and individual firms may provide more detailed case-specific information. An important example is that of industry providing information on the extent and nature of use of a particular chemical in a particular application (or set of applications of concern).

The method of seeking stakeholder input can take many forms, with differing requirements in terms of time and resources. A number of governments encourage stakeholder consultation by making information on proposed decisions widely available and inviting public comments. Methods include formal conferences, round tables, expert symposia, facilitated workshops, public meetings and working group meetings. Surveys, questionnaires and focus groups can also be used to gather feedback and gauge public opinion.

Other fora to encourage more involvement from the lay public, such as ‘Citizen’s Juries’ and ‘Stakeholder Dialogues’, are also increasingly being used. The former involves the use of a ‘jury’ taken from the general population which, after listening to a range of expert witnesses, considers an issue in an objective and informed way before making a (usually non-binding) recommendation to decision makers. Stakeholder Dialogues use consensual processes, such as mediation and facilitation, to deepen, broaden and extend the process of communication and negotiation. The argument underlying the use of this process is that it helps people with different interests, values and resources build mutual understanding, find common ground, resolve shared problems and reach agreement in complex situations. Stakeholder dialogue has been used mainly in situations where there is complexity, uncertainty and differences of interest, or where decision makers need input from a range of stakeholders. As can be seen from Table 3.1, the different stages comprising the dialogue process are similar to the progression of activities involved in preparing a SEA.

Table 3.1: Stages of a Stakeholder Dialogue Process

1. Preparation and design: involving extensive discussion with many stakeholders.
2. Opening events: *e.g.* a public meeting or a series of private meetings with stakeholders.
3. Story telling: a facilitated process to enable participants to understand each other’s points of view.
4. Identifying issues: clarification, classification and structuring issues identified through story telling.
5. Identifying interests, values, needs and fears: often involves breaking into sub-groups to focus on particular issues.
6. Exploring options and managing uncertainty: the exploration stage of the dialogue, sometimes lasting over many meetings.
7. Agreement: written agreements which are specific, timed, balanced, positive and realistic, including contingency plans and processes to resolve future difficulties.
8. Evaluation and review of decisions: to ensure that the decisions are working as intended or to negotiate any variations that are required.

Source: Environment Council (1999)

4. THE METHODS OF ANALYSIS

4.1 Introduction

For the purposes of this document, we have organised the material according to the types of information that the different methodologies provide about alternative risk reduction options:

- tools for assessing economic efficiency: cost-effectiveness analysis and cost-benefit analysis;
- tools for assessing macroeconomic impacts: input-output models and general equilibrium models; and
- tools for assessing options against a range of decision criteria: screening methods, multi-criteria techniques and financial analysis.

This categorisation has been adopted to make more explicit the differences inherent in the information provided by the various methodological tools. The methodologies vary by the types of impacts that can be assessed through their application. Table 4.1 sets out the types of tools that may assist with the different stages in SEA. A brief review of the main methodological frameworks is provided here, with a more detailed discussion on the theoretical underpinnings provided in Parts 2 and 3. For reference, an indication is also given on where to find discussion on the various tools or forms of analysis within this document. As will be noted from the table, the tools relate to only some of the tasks involved in preparing a SEA, as many of the tasks are more process-related.

4.2 Tools for Assessing Economic Efficiency

The tools falling under this heading have been developed to assess whether a proposed option is economically efficient or not, and to indicate the most efficient option from a range of alternatives. The main methodologies are:

- **Cost-effectiveness analysis (CEA)** is widely used to determine the least cost means of achieving pre-set targets or goals, with these targets defined by government guidelines or legislation. In such analyses, the costs relate to the estimated resource costs (or opportunity cost) of meeting the target or of adopting a particular option, while the effectiveness relates to a physical outcome (*i.e.* cost per statistical life saved); alternatively, CEA can be structured to determine the means of achieving the maximum effectiveness for a given cost.
- **Cost-benefit analysis (CBA)** provides a framework for comparing both the social costs and social benefits of a proposal as they would be measured in economic resource terms; the nature of the analysis may range from one which is mainly qualitative to one which is fully quantitative with the aim being to determine if a proposal is efficient from a social perspective (in terms of the benefits to society being greater than the costs).

Table 4.1: Stages, Tasks, Relevant Tools and Associated Information				
Stage	**Task**	**Tool**	**Scope of Information and/or Impacts**	**Relevant Parts (P) and Sections (S)**
Stage 1	Screening Level Risk Assessment	MCA	• Screening techniques	Part 2 and Part 3, Sec 4
Stage 2	Identify Options	CEA, CBA, Macro-economic Analyses, MCA	• Screening against key criteria • Simple rankings of options	Parts 2 and 3
Stage 3	Appraisal of Options	CEA	• Compliance costs • Qualitative and physical data on health and environmental effects	Part 3, Sec 2 Part 2, Sec 2 and Part 3, Sec 3 + 4
		CBA	• Compliance costs • Monetary valuation of health and environmental effects • Qualitative and other quantitative data on health and environmental effects	Part 3, Sec 2 Part 3, Sec 3 + 4 Part 3, Sec 3 + 4
		Macro-economic Analyses	• Compliance costs • Employment effects • Impacts on industry sectors and wider economy (macro-economic effects)	Part 3, Sec 2 Part 2, Sec 3 and Part 3, Sec. 2 Part 2, Sec 3 and Part 3, Sec 2
		MCA	• Compliance costs • Qualitative or quantitative data on health and environmental effects • Impacts on wider economy • Employment effects • Other issues	Part 3, Sec 2 Part 3, Sec 3 + 4 Part 3, Sec 2 Part 3, Sec 2
		Equity Analyses	• Distributional effects by population or location	Part 3, Sec 5
		Discounting	• Impacts occurring over time	Part 4, Sec 2
		Uncertainty Analyses	• Sensitivity of decision to uncertainties, distribution of possible outcomes, value of information	Part 4, Sec 3 Part 4, Sec 3 Part 4, Sec 3
Stage 4	Presenting Information	Presentational Tools		Part 4, Sec 4

The principles underlying these methodologies are elaborated further in Part 2. In general, the principles underlying CBA appear to provide the preferred framework for many countries that have established procedures involving the application of SEA to chemical risk management. For many regulatory decisions, however, it will be important that both an economic appraisal (using CEA or CBA) and a financial appraisal (see Section 2.6 of Part 3) are undertaken in order to provide decision makers with information on both the degree to which a proposed option yields net benefits to society as a whole and to which a market sector can 'afford' to implement a proposed option.

The focus of these methodologies also tends to be on estimating the direct effects of a change in regulatory policy, although indirect effects in a few related markets can also be taken into account. The analyses examine the implications of adopting a new regulation at a detailed, sector by sector level, assuming that all other sectors in the economy remain unaffected (in other words, they act as a partial equilibrium analysis). Aggregation to a national level using these types of methods then requires that a range of assumptions are made as to the impacts of the action across all firms within a sector and on any relationships between sectors.

4.3 Tools for Assessing Macroeconomic Impacts

The tools available for assessing impacts at the macroeconomic level involve some modelling of the interactions between different economic agents in the economy. These models are aimed at describing how the implementation of new legislation by businesses affects their behaviour as ‘buyers’ and ‘sellers’, as such changes in behaviour will affect other sectors, the interactions between sectors, and ultimately the functioning of the entire economy.

The key top-down modelling approaches are:

- **Input-output (I-O) analysis** which is based on the development of a set of accounting relationships that show the linkages between the purchase of outputs by different sectors of the economy, as well as for labour, capital and raw materials (primary inputs).

- **General equilibrium (GE) analysis** which considers both supply and demand interactions, and allows quantification of the direct and indirect effects of policies on the structure of an economy, product mix, economic growth, the allocation of resources and the distribution of income.

In contrast to the use of CEA and CBA, the above models provide no information on whether or not a proposed option is efficient in economic terms (*i.e.* the social benefits outweigh the social costs). They only indicate what the impacts would be at a macroeconomic level. In addition, because these models operate at the macroeconomic level, they sacrifice technical detail for greater spatial scope.

The use of these tools is unlikely to be of value for most chemical risk management issues, where the main concern is likely to be that of economic efficiency. Where macroeconomic modelling may be more valuable is in the context of transboundary air quality issues, for example, in order to understand the implications of climate change policies which may impact on a wide range of market sectors and hence the economy as a whole. However, any policy that is likely to have a major economic impact and any policy that would result in the loss of a major product has the potential to affect other sectors and the wider economy.

4.4 Tools for Assessing Impacts against a Range of Criteria

The other main category of tools are those which allow examination of risk reduction options against a range of decision criteria, where these may include both economic and non-economic criteria. Within the context of these guidelines and chemical risk management, this heading refers to the various analytic approaches falling under the heading of multi-criteria analysis (MCA) and the use of financial analysis.

The term multi-criteria analysis is used here to refer to the range of techniques that have been developed to allow impact information expressed in different units of measure to be combined for decision making purposes. The techniques range from the more simple qualitative tools provided by screening, ranking and pairwise comparison methods, to the more complex (utility theory based) scoring and weighting methods, which allow aggregation of information across different attributes into an overall indicator of relative performance.

In the more sophisticated MCA techniques, a distinction is made between the impact of an action on a given attribute or criterion and the importance that is assigned to that attribute or criterion. The impact of an action is determined through scoring procedures, while importance is determined by the weight assigned to a criterion. This contrasts with the economic appraisal methods, where such a distinction does not exist since the two aspects are measured jointly through the monetary valuation process. In addition, the ability to include separate and explicit importance weights may be of value where there are widely varying viewpoints. This aspect of MCA means that such analyses can allow for consideration of a range of different value systems when assessing the relative cost, benefit and risk trade-offs.

The different tools available under this heading are discussed in more detail in Parts 2 and 3. To date, MCA techniques have not been widely applied as part of chemical risk management. This is due to both legal and methodological considerations. As noted earlier, in some countries (notably the US), there are legal requirements to undertake certain forms of analysis, in particular CBA, for some aspects of chemical risk management.

However, in other cases (with the EU being an example), no such constraints exist and the reasons are more methodological in nature. A key factor is that half of the cost versus benefit equation - the cost side - is readily measured in money terms. Because decision makers and others are used to thinking in these terms, the preference of many is to consider the benefits in the same unit of measure as this allows direct comparison. Hence, the greater emphasis placed on CEA and CBA as the most appropriate methodologies. In addition, the question of how one develops a politically agreed set of weights for use in the scoring and weighting techniques is regularly raised, with this identified as a key constraint when analyses are international in scope. However, as there are likely to remain difficulties in deriving money estimates for all environmental and human health risks, the potential exists for greater use of MCA techniques as part of chemical risk management (DETR, 2000; Hokkanen & Pellinen, 1997).

Also included under this heading is the use of financial analysis. Financial analysis is aimed at determining the impact that a proposed regulation will have on industry and its competitiveness. It differs from CEA and CBA in that it focuses on changes in cash flow (rather than social costs) to different industry/business sectors and on how these affect the financial performance of companies, government or individuals impacted by policies and programmes. As indicated above, such analyses are often prepared alongside CEAs and CBAs, and indeed they may share some of the same cost information (particularly where compliance costs are used as a proxy for social costs).

5. ANALYTICAL ISSUES

5.1 Overview

Regardless of the methodology adopted as the basis for the SEA, several analytical issues may need to be addressed during the assessment. A number of these are likely to be common to all SEAs, with the key issues concerning:

- setting the baseline for the analysis;
- specifying the analysis time horizon;
- the reliability of impacted predictions;
- the impacts of substitute processes or chemicals and technological change;
- issues in quantification and conversion of predictions to a comparable basis;
- distributional and equity issues; and
- the management of risk and uncertainty.

A brief overview of these issues is provided below. More substantive discussions on how each should be approached are provided in Parts 2 and 4. See also G*uidance on Conducting Retrospective Studies on Socio-Economic Analysis* (OECD Series on Risk Management, No. 11, 1999) for further discussion on some of the common criticisms levied against SEAs.

5.2 Setting the Baseline

The baseline refers to the status of health, environmental and economic conditions in the absence of further risk management; in other words, it describes the present situation and anticipated outcomes with no new actions being taken by government. Determination of what constitutes the baseline is central to conducting any form of SEA, as the costs and benefits of risk management should be assessed in terms of the marginal changes stemming from the introduction of a proposed option. In defining the assumptions that will form the baseline, the following issues may need to be considered:

- the behaviour of the target industry in the absence of new government action in relation to the chemical risks in question; of key importance will be any trends associated with changes in technology or product design, market re-structuring, *etc.* as such changes may render a baseline defined in terms of current conditions unrepresentative of future conditions;
- the impacts of customer demands on the target industry and thus on product innovation and development;
- the level of current compliance with existing risk management regulations; and

– the potential implications of future initiatives that may have an indirect effect on the chemical risks of concern.

With regard to the above, it is interesting to note that different regulatory agencies may treat compliance issues differently when defining the baseline for the analysis. It is most common for the baseline to assume that industry complies fully with existing regulations (in order to ensure that the benefits of complying with existing regulations are not incorrectly attributed to a new regulation). The potential need to consider more than one set of baseline assumptions is also noted in cases where the baseline is uncertain as one moves into what would happen in the future, for example, owing to shifts in technology or the potential for increased integration within industry.

In all cases though it will be important to ensure that the same baseline is used when assessing costs and benefits (particularly when responsibility for these assessments is assigned to different analysts). This is likely to be particularly important where different analysts are assigned different components of the SEA (*i.e.* one person takes responsibility for estimating costs while another is responsible for benefits).

5.3 The Project Timescale

In any analysis, decisions have to be made over the time period to be used in analysing the costs and benefits. If the time period adopted for the analysis is considered to be too short for the life of the policy (*e.g.* 5 to 10 years), it may fail to take proper account of impacts that will occur further into the future (such as environmental and health impacts).

There are no absolute criteria for establishing what the time period should be for a particular assessment, but a number of factors may be taken into account:

– the life of relevant capital equipment;

– the timing associated with implementation of different risk management options;

– the remaining expected life of the chemical under examination; and

– the nature of the environmental or health risks, and whether the benefits of risk reduction would be realised in the short-term or the longer-term.

Perhaps the most commonly used criterion is that of the life of any capital equipment that would be required as a result of risk management. However, as the environmental benefits arising from reduced chemical exposure may take 20 years or more to be realised, it is essential that the time period taken for the assessment is adequate to reflect the full magnitude of future benefits. This is a particular concern where such effects are valued in money terms.

5.4 Predicted versus Actual Impacts

Analysts are increasingly aware that the impacts predicted as part of pre-policy appraisals (referred to as *ex ante* estimates) do not always correspond with what happens after a regulation is adopted (referred to as *ex post* impacts). This type of finding has led to increasing emphasis being placed on the use of retrospective analyses, which can help identify why such differences have arisen.

These findings can then help in reformulating policy as appropriate to address any significant unpredicted effects.[8]

A number of reasons have been hypothesised as to why *ex ante* estimates may differ from actual outcomes. With regard to costs, estimates may be incorrect due to the failure of the analysis to consider all cost elements. Similarly, actual costs may vary as a result of assumptions concerning contingencies within estimates or when budgets are based on other than median estimates, as under or over runs may be much more likely where some other estimation basis is chosen. Even where industry is able to provide some indication of the potential for technological change and for cost efficiencies to result from 'learning by doing', the resulting figures may still represent under or overestimates given the lack of knowledge of the future.

Just as such improvements in productivity may reduce the estimated costs, technological change could affect predictions of the benefits stemming from a proposed regulation. Hence, the potential implications of technological change on both costs and benefits is likely to be one of the areas leading to the greatest level of uncertainty within an analysis.

The following are all further reasons as to why differences between predicted and actual effects may arise:

- the failure to identify all of the potential health and/or environmental effects associated with a particular chemical;
- as a result of scientific uncertainty surrounding dose-response relationships and the environmental population currently at risk;
- because of a lack of information on the role that one chemical plays as part of a multi-pollutant problem;
- due to the perspective adopted within the risk assessment and whether this is based on worst-case, best-case or median value estimates; and
- the failure to consider the impacts that would arise from the adoption of substitute processes or chemicals, or from their adoption having unintended consequences.

Given the importance of recognising and addressing uncertainty within the SEA process and associated with the analysis results, a separate section discussing key issues and the techniques available for managing uncertainty is provided in Part 4.

5.5 The Impact of Alternatives

Key Considerations

Section 2 highlighted the fact that there is a range of potential options which can be considered when developing chemical risk management strategies, varying from command and control approaches to market instruments, worker safety programmes, product stewardship, engineering controls, product and packaging design, information tools and voluntary agreements. It also highlighted that each of these different options may address different life-cycle stages, resulting in the need to consider a mix of options for some chemical risk management issues. Each option or mix of options is likely to have different risk, cost and benefit implications. Although the actual

[8] See also *Guidance for Conducting Retrospective Studies on Socio-Economic Analysis* (OECD Series on Risk Management, No. 11, 1999).

requirements associated with the different options vary, industry responses may involve changing production techniques, changing the nature of the end product, or adopting a replacement chemical.

It will, therefore, be important that adequate consideration is given to the implications associated with the adoption of alternatives, and in particular to the use of substitute chemicals as replacements or in the reformulation of a new product aimed at achieving the same end. The European Commission seeks answers to the following questions when assessing the impacts of a risk reduction option and the introduction of substitutes (EC, 1998):

- What substitutes might be used in place of the substance in question? Are these substitute chemicals or 'not-in kind' alternatives, such as new production methods? What is their market situation?

- Do these substitutes (chemical or other) present a new set of risks? If so, what is the nature of these risks?

- Are the substitutes effective for all of the same situations as the original substance? Will new technology, equipment or processes be required by industry to achieve the required results using the substitutes? Will adoption of the substitute increase costs or reduce costs? Is the option likely to foster technical innovation?

- Will there be a loss of production facilities and other specialised capital and technology used in the manufacture of the restricted chemicals or products?

- What research and development is necessary in order to adopt the substitute(s)? Will such activities require significant expenditure? Will retraining of personnel on use of the substitute(s) be required?

- Will the end product be of higher or lower quality? Will the consumer have the same level of satisfaction? and

- Will some products disappear due to a lack of substitutes? Or, is it likely that the overall range available for a given product type will increase?

In an ideal world, a full analysis of the range of potential substitutes would be undertaken to determine the risks, costs and benefits associated with their adoption. Such an analysis would consider not only the costs (or savings) associated with the adoption of a substitute but also the availability, efficacy and risk potential of possible substitutes:

- **availability**: in some cases, chemicals will already be in the process of being phased-out because of the availability of improved production methods or the existence of lower cost, less toxic, or better performing substitutes; while in other cases, little effort will have been put into developing alternatives, owing either to the cost of the associated research and development or the lack of a current demand for alternatives. In both cases, restrictions on use in the short-term could have severe effects on specific industry sectors;

- **efficacy**: the move to an alternative may lead to a deterioration in the quality of the final good produced unless there is sufficient flexibility in the technology currently in use. In such cases, the losses to consumers may be significantly greater than indicated by the costs of replacement products or of process changes alone; and

- **risk assessment**: many substitute chemicals are likely to be new, and there may be only limited data on the risks associated with their use. This has been identified as a problem when undertaking SEAs of pesticide use, for example, where the data on the alternatives are generally inadequate to complete full risk assessments (Reinert *et al*, 1990). In such cases, it may be necessary to use the limited information available to undertake a comparative risk assessment, while highlighting the main uncertainties; further research needs concerning risks should also be highlighted.

In practice, there are constraints on the time available to prepare a SEA and thus the detailed examination of substitutes is generally not included in the study. Care is therefore required to ensure that implicit assumptions are not being made about the substitutes.

5.6 Issues in Quantification and Conversion to a Comparable Basis

Within any SEA, there are likely to be constraints on the degree to which predicted impacts can be quantified. For example, a lack of data or knowledge may affect the degree to which the costs of introducing an option can be quantified. Similarly, there are likely to be constraints on the degree to which environmental and human health risks can be quantified. As a result, it will be important that any non-quantified effects are not automatically given less weight and, thus, overshadowed by those which have been quantified and valued. Similarly, care should be taken to ensure that non-quantifiable effects are not automatically assumed to be of greater value than the quantified negative and positive costs and negative and positive benefits.

Many of the reasons for the difficulties in quantification relate to scientific uncertainty and a lack of information/knowledge. With regard to the quantification of environmental and human health risks, this relates to:

- current exposure rates at regional and local levels;
- dose-response relationships for the various risk end-points of concern; and
- the population (environmental or human) currently at risk (or likely to be at risk in the future).

Some of the above data may be available for certain risk end-points, but it is unlikely that such data are comprehensively available for most chemicals of concern. This absence of data means that SEAs are constrained to using more qualitative methods for describing many of the environmental and human health benefits (or will need to rely on the use of multi-criteria techniques).

Where the above data are available (even if uncertain), some quantification should be possible. Within the context of chemical risk management, it will be important that changes in risk are quantified in physical terms using the most appropriate units of measure first. Where there is a desire to convert all units to a common basis for aggregation, this can then proceed through either monetary valuation or through the use of one of the MCA techniques. In either case, it will be important that decision makers are provided with data (whether qualitative or quantitative) on both the actual impacts and the implied ‘value’ of those impacts.

With regard to monetary valuation as part of a cost-benefit analysis, there is a range of techniques available for deriving monetary values of human health and environmental effects, with value being defined by individuals’ preferences. These preferences are obtained by measuring either individuals’ (and hence society’s) *willingness to pay* for a benefit (or to avoid a cost) or individuals’ *willingness to accept* compensation for a loss. These techniques involve the examination of the market value of gains and losses, of individuals’ *revealed* preferences or of individuals’ *stated* preferences (with the different techniques discussed in detail in Part 3). It is unlikely, however, that all impacts can be valued in monetary terms for direct inclusion in a CBA.

In contrast, it should be possible using MCA to develop impact scoring systems for most environmental and human health effects, although commonly cited problems in developing such systems are:

- defining individual impacts (criteria) so as to be independent and separable to avoid double counting when specifying the impact categories;
- developing the scoring system so that it properly respects the relative proportionality of different levels of effect; and
- where the aim is to aggregate impact information, eliciting a set of relative importance weighting factors (for use in aggregating impact scores to a common unit of measure) that will reflect society's values rather than those of a few experts.

Discounting to Account for Temporal Differences

One of the aims of using either monetary valuation or scoring and weighting techniques is to allow aggregation of predicted negative and positive costs and negative and positive benefits. However, given the wide range of these that may need to be brought together, there are likely to be significant temporal differences in when they occur. Typically, the costs of risk management will be incurred upon implementation of an option, while the benefits occur further into the future.

Whether performing a cost analysis, CEA, CBA or MCA, such aggregation requires that the streams of costs and/or benefits occurring over time are reduced to a single figure in order to allow their conversion to a common basis. This is achieved through the use of discounting procedures, and should apply equally to all types of impacts (*e.g.* monetary and non-monetary).

Discounting reflects the assumption that individuals would prefer to have money or other types of gains now rather than some time in the future; future costs and benefits are therefore 'discounted' and attributed a lower weight than those occurring immediately. The higher the discount rate used, the lower the importance placed on future costs and benefits. At any positive discount rate, costs and/or benefits that accrue more than 50 years into the future will have a very small 'present value'. As the impacts of environmental changes (such as the concentration of persistent chemicals, CO_2 concentrations, *etc.*) may last for a longer time period than other social changes, the rate of discounting can have a significant effect on the appraisal.

A key question in preparing a SEA is therefore likely to concern the choice of discount rate. In financial analyses, the discount rate prevailing in financial markets may be more appropriate (as this reflects the opportunity costs which companies face when investing in one capital use today as opposed to another in the future). In economic appraisals, the correct discount rate is what is referred to as the *social discount rate*, which reflects the rate at which society as a whole is willing to trade-off present for future negative and positive costs and negative and positive benefits (the rate of time preference). The market rate and social discount rate are usually different and most governments appear to assume a single number, lying within a plausible range between them (with rates tending to vary from between 3% and around 6%).

Although various regulatory agencies have established groups to consider issues such as the discounting of environmental and health effects, and treatment of inter-generational equity considerations (*e.g.* the US EPA and the UK), current practice between countries varies only slightly. The issues that arise in these contexts are reviewed in more detail in Part 3, in the sections discussing prediction of environmental impacts, health effects and distributional effects.

5.7 Distribution and Equity Issues

Policy makers generally place great importance on equity and distributional issues, wishing to consider the fairness of a proposed option in terms of the incidence and distribution of benefits and costs, as well as its overall net benefits. Information on distribution effects can be incorporated into a SEA either through qualitative discussions or through the use of more formal quantitative analyses.

Formal distributional analyses involve the partitioning out of information on the negative and positive costs and negative and positive benefits by sub-population within an overarching appraisal, or by supplemental analyses conducted for some or all of the options available to the decision maker. Although such formal analyses are used in North America, they tend not to be in other OECD regions.

One of the options available for improving the ability of socio-economic analyses to identify and evaluate environmental justice is through the use of scoring and weighting systems. The use of scoring and weighting techniques to allow for the incorporation of equity and distributional effects into SEAs is by no means new and has the advantage of being adaptable to the particular issue at hand (see Part 3 for further discussion).

5.8 Managing Uncertainty

Uncertainty stems from a lack of information, or a lack of knowledge about the consequences of a given action. A range of different types of uncertainty may exist, including uncertainty on:

- future relative prices of key cost components and the most appropriate values for use in the analysis, in particular as part of benefit transfer;
- rates of technological change and the way in which affected industries will adapt to a new regulation;
- the wider economic consequences of taking a particular action;
- the science surrounding base environmental and health data (exposure rates, dose-response functions, *etc.*);
- the time-frame over which costs and benefits will occur;
- the range of assumptions made by modellers as part of modelling and estimation activities;
- policy goals and how to weigh one decision factor against another; and
- related future decisions and how they may affect the outcome.

Sensitivity analysis provides a means for determining the importance of uncertainty to the end results and is a standard requirement of most SEA guidelines. The aim of such analyses is to assist risk managers and stakeholders in understanding the level of confidence that can be placed in the assessment results. At its simplest level, sensitivity analysis can be carried out by varying the values assumed for key uncertain variables, to test the sensitivity of the end results to changes in these values. More sophisticated techniques are also available, such as Monte Carlo analysis, Delphi methods and meta-analysis.

Where uncertainty surrounding one or more key assumptions is critical to the choice of the risk reduction option, 'value of information' analysis can be used as part of an economic appraisal. Such analyses help determine whether or not it is more appropriate to base a decision on incomplete information or to delay making a decision until sufficient data have been collected to reduce the key uncertainties. The 'value' of this additional information is derived by considering the difference in the expected outcomes with and without the information. The approach relies upon the use of conditional probabilities and expected values. If the additional cost of gaining the extra information is outweighed by its expected benefits, then it is worthwhile obtaining the additional information.

All of these tools can be of value in determining the ‘robustness’ of different options in terms of their ability to withstand errors in prediction, assumptions and other forms of uncertainty. The concept of robustness and further details on the various techniques described above are reviewed further in Part 4, Section 2.

6. LINKS TO RISK ASSESSMENT AND RISK MANAGEMENT

As highlighted earlier, the outputs of risk assessments form critical inputs to SEAs. Similarly, some of the information that economists use is an important input to risk assessments; for example, data about markets and uses of chemicals are essential when considering exposure potential and alternatives. The overall process of developing risk and socio-economic information for chemical risk management purposes, should be seen as being interrelated, with there being iterations between the two types of analysis.

However, the need to improve the links between risk assessment and SEA has been recognised by both disciplines. For example, a report by the US Presidential/Congressional Commission on Risk Assessment and Risk Management (1997) recommended greater co-ordination between risk assessors and economists to reduce inconsistencies in approaches. More specifically, a recent multi-organisation sponsored workshop (OECD, 2000) identified the following issues as requiring examination to improve co-ordination:

- identification of the types of information about exposure, toxicity, chemical or process substitution that is required by both types of analysis and how this can be shared to improve both;

- examination of approaches for better integrating the outputs of risk assessments with the risk estimates desired by economists. Risk assessments often evaluate risk by comparing exposures to a reference standard, such as a hazard quotient or actual exposure levels, to those known to produce an adverse effect. Economists, on the other hand, seek information on changes in risk expressed as probabilities. The issue is whether there are ways to interpret non-probabilistic risk data for use in SEAs;

- the approaches used by risk assessors and economists to model risks and socio-economic effects often rely on conflicting assumptions. SEAs require data on population risk estimates (*i.e.* statistical lives saved or percentage change in population level); while risk assessments frequently provide estimates of changes in individual risks, for instance risks to the individual most exposed;

- the outputs of both risk assessments and SEAs are uncertain and it is therefore important that both disciplines understand how these uncertainties affect the other type of analysis so that uncertainty as a whole can be better managed; and

- the degree to which the two types of analysis are consistent in the conservatism adopted. For example, the risk assessment may include a margin of safety and use upper bound estimates in order to adopt exposure levels that are protective. SEAs on the other hand attempt to provide best estimates of the benefits of reduced risks for comparison to best estimates of costs.

7. REFERENCES

DETR (2000): Multi-Criteria Analysis: A Manual, London, Department of the Environment, Transport and the Regions.

EC (1998): Technical Guidance on Development of Risk Reduction Strategies, Directorate General XI, Brussels, European Commission.

Hokkanen & Pellinen (1997): The Use of Decision-Aid Methods in the Assessment of Risk Reduction Measures in the Control of Chemicals, Copenhagen, Nordic Councils of Ministers.

Health & Safety Executive (1995): Policy Appraisal and Evaluation, General Administrative Procedures No.23, Economic Advisors Unit, HSE, UK, September 1995.

OECD (1991): Environmental Policy: How to Apply Economic Instruments, Paris, OECD.

OECD (1994): Managing the Environment - The Role of Economic Instruments, Paris, OECD.

OECD (1999): Guidance for Conducting Retrospective Studies on Socio-Economic Analysis, OECD Environmental Health and Safety Publications, Series on Risk Management No. 11, Paris, OECD and also Risk & Policy Analysts Ltd (1999).

OECD (2000a): Proceedings of the Workshop on the Convergence of risk assessments and Socio-Economic Analyses to Better Inform Chemical Risk Management Decisions, Workshop Proceedings, May 1-2 Arlington, Virginia.

OECD (2000b): Framework for Integrating Socio-Economic Analysis in Chemical Risk Management, OECD Environmental Health and Safety Publications, Series on Risk Management No. 13, Paris, OECD and also Risk & Policy Analysts Ltd (2000).

Office of Management and Budget (1997): Report to Congress on the Costs and Benefits of Federal Regulations, Washington DC, OMB.

The Presidential/Congressional Commission on Risk Assessment and Risk Management (1997): Risk Assessment and Risk Management in Regulatory Decision Making, Final Report Volume 2, Washington DC.

Reinert et al (1990): A Discussion of the Methodologies used in Pesticide Risk-Benefit Analysis, The Environmental Professional, Vol.12, No. 2, pp. 94-100.

Treasury Board of Canada Secretariat (undated): Assessing Regulatory Alternatives, http://www.pco-bcp.gc.ca/raoics-srdc/raoic_e.htm.

The World Business Council for Sustainable Development (1999): Meeting Changing Expectations, Geneva, WBCSD.

ANNEX TO PART 1:

STAGES IN THE SEA PROCESS

Figure A.1: Identifying the Problem

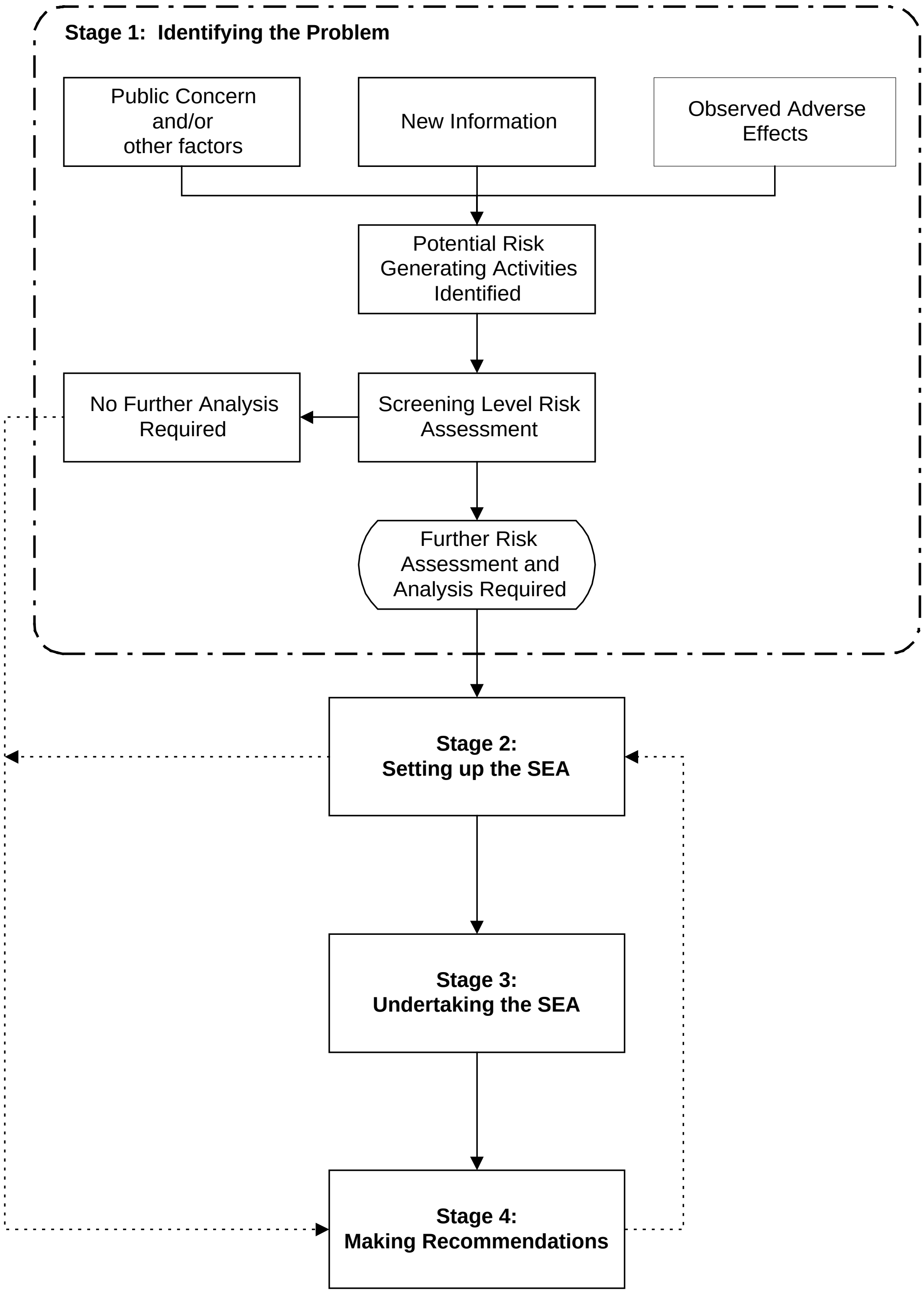


Figure A.2: Setting up the SEA

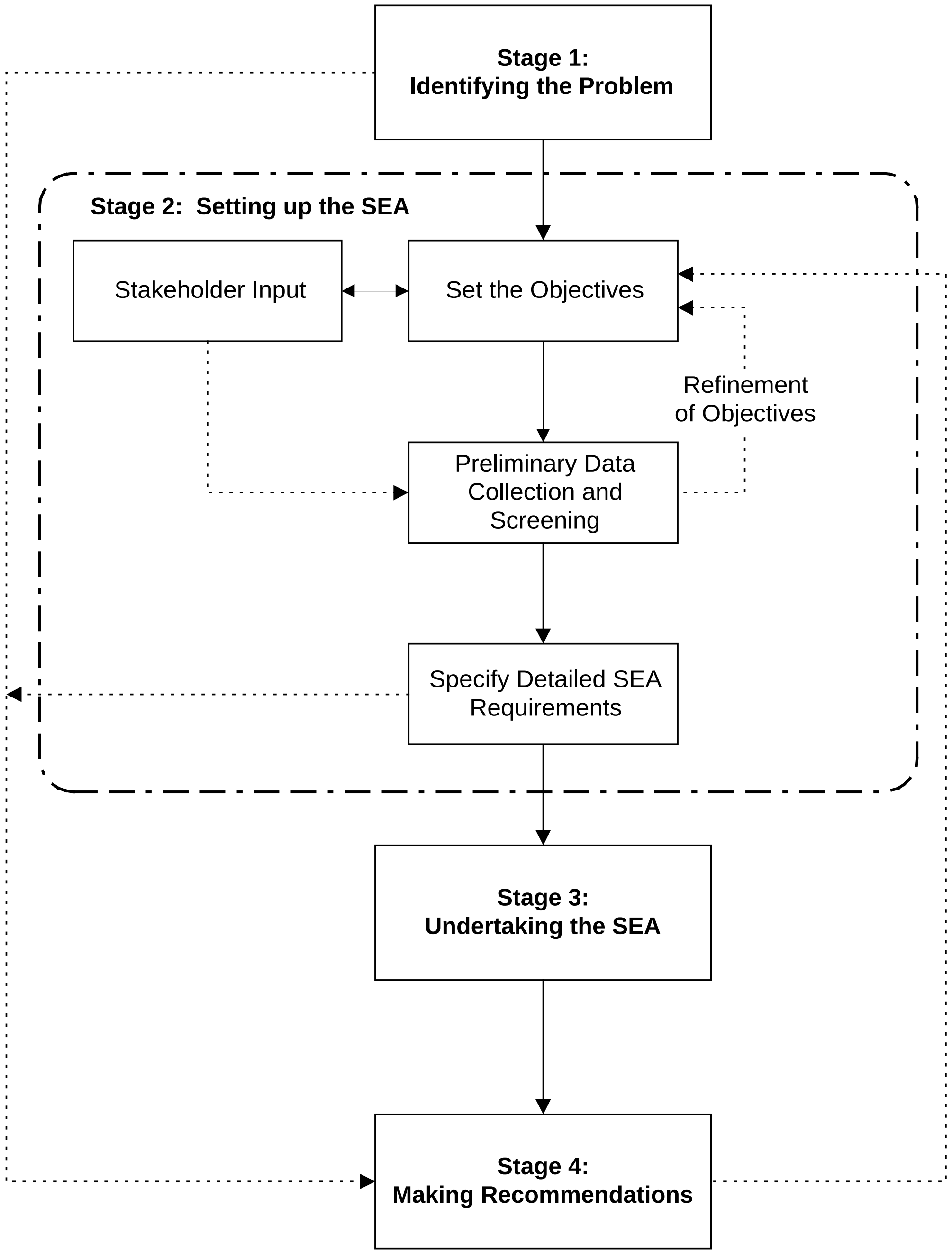


Figure A.3: Undertaking the SEA

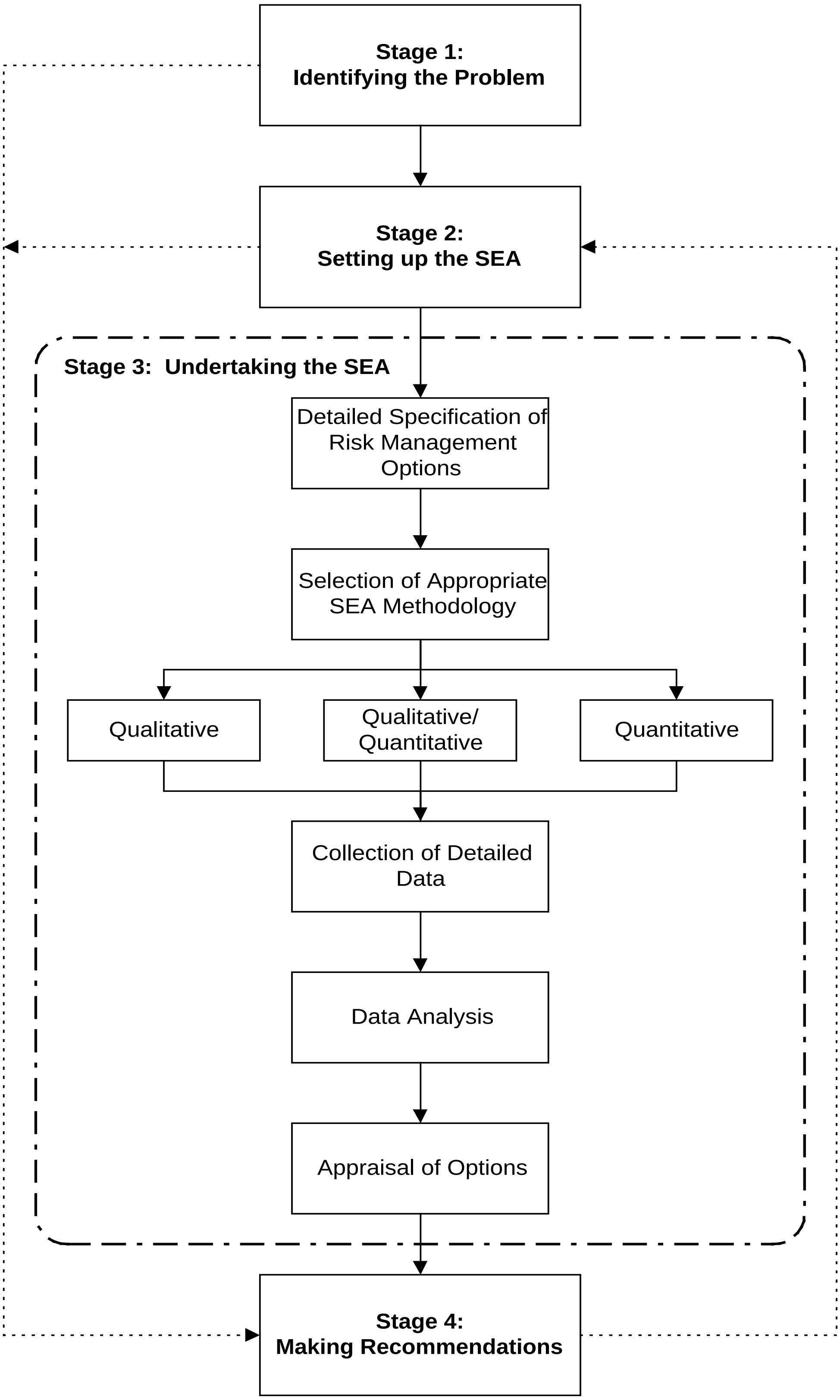
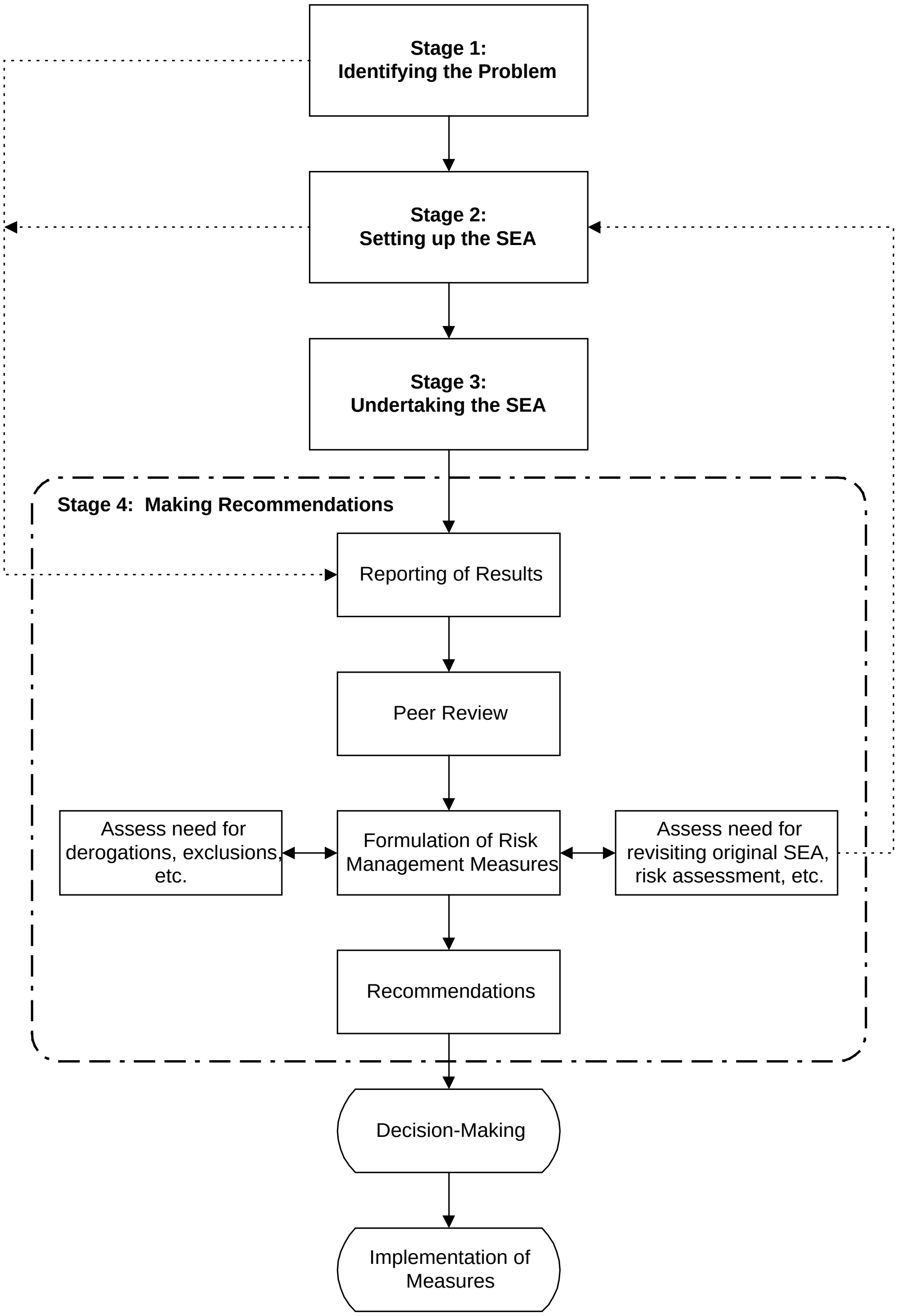


Figure A.4: Making Recommendations



PART 2: THE TOOLS FOR SOCIO-ECONOMIC ANALYSIS

1. INTRODUCTION

1.1 Overview

This part (Part 2) of the guidance sets out the key tools for undertaking a SEA. It starts by giving a brief introduction to the three categories of methodological tools that can be called upon as part of conducting SEA, with these being:

- tools for assessing economic efficiency (with these tools generally used to assess impacts at the micro-economic level);

- tools for assessing economic impacts at the macro level; and

- tools which can assess impacts at either the macro or firm/sectoral level and embrace wider, non-economic decision criteria.

A brief introduction to the different tools and when one tool might be better than the others is provided below.

Sections 2 to 4 then discuss the concepts underlying the tools in more detail, in order to provide an understanding of the basic principles, of what is required in applying the different tools, and of some of the issues that may arise in so doing. These discussions draw on the use of a hypothetical risk management issue, based on an imaginary set of chemical compounds commonly referred to as ZOZ. An indication is given for each tool as to how it would be applied as part of decision making concerning ZOZ.

1.2 The Different Tools

Most countries' approaches to SEA are based on the principles of economic appraisal and the desire to demonstrate that the benefits of proposed policies outweigh the costs. The two most common methods used for these purposes are cost-effectiveness analysis (CEA) and cost-benefit analysis (CBA). The principles underlying these two methods form the basis for most chemical risk related socio-economic analyses. Proponents of economic appraisal argue that this is because these methods provide a systematic framework for collecting and organising information. The template character of such analyses enables the decision maker to determine the adequacy of the information collected, to identify whether and what important information is missing, and to incorporate the conclusions into the risk management decision making process.

While CEA and CBA may be more commonly used or suggested for use within SEAs, the full range of tools is much wider. As a result of their focus on economic efficiency as the key information to be developed, CEA and CBA are effectively 'bottom-up' approaches in that they start with the sectors that will be directly affected by the regulation. For practical reasons, they tend to focus on estimating the direct effects of a proposed option on the regulated activities, although they can also be applied to capture many of the secondary or indirect effects that may result.

The next category of economic appraisal tools (comprising input-output analysis and general equilibrium models) are not designed to provide information on efficiency. Rather, these 'top-down' macroeconomic approaches focus on the impacts of a proposed option at a multi-sectoral level, covering the structure and functioning of the economy as a whole. Such approaches are much less commonly used as part of chemical risk management, and are most relevant in cases where an option would change the prices and costs faced by a large number of production activities (whether in private or public ownership).

It should be noted that the above forms of analysis are not exclusive in nature. There may be many cases where information on both economic efficiency and effects at the macroeconomic level, in addition to other information, will be of value to decision making. This may be the case where regulations are likely to affect a chemical that is either used by or a pollutant emission of a highly integrated product or production process (such as a fuel or energy source). Examples of such regulations are related to air quality and global warming, which may impact on a group of specific industry sectors (such as large users of fossil fuels) and also on those market sectors supplying to and buying from this group. In such cases, because a regulatory option may have a widespread effect on the economy, it may be important to consider not only the economic efficiency of an option but also its impacts on the economy more generally. In this regard, the results of macroeconomic modelling could be fed into a CBA, for example.

The final category of techniques are those which do not rely on the use of economic appraisal methods. Multi-criteria analysis can embrace a wide range of different information, including both quantitative and qualitative measures of performance. The most relevant techniques can be used to rank the relative performance of different measures against one or more criteria. Or, techniques can be applied which, similar to CBA, allow the aggregation of information on costs and benefits[9], in this case, however, there is no requirement that the costs and benefits be measured in the same units. As with CEA and CBA, MCA can be applied from either a ‘bottom up’ or a ‘top down’ perspective, and can be used to supplement economic analyses (*i.e.* where monetary valuation of environmental and human health benefits. whether positive or negative in nature, is not possible). In such cases, MCA may be a valuable tool for assessing the relative performance of different regulatory options against different objectives and impact criteria.

Table 1.1 summarises the types of impacts that can be covered by the different assessment tools. It should be noted that MCA differs from the other approaches. Economic analysis (CBA and CEA in this context) has been developed to provide an ‘objective’ analysis based on a single decision criterion, while MCA has been developed specifically to enable decision makers to incorporate subjective judgements into the analysis and to allow analysis against multiple objectives.

[9] It is important to recognise that ‘costs’ to industry may include not only increased expenditure (positive costs) but also savings (negative costs). Similarly, it is usual to assume that chemical risk reduction leads to health and environmental benefits (positive benefits). However, shifts to other products or other chemicals may also result in health and environmental damages (negative benefits). In this document, costs relate to the impacts on industry, consumers and regulators arising from the implementation of a risk reduction option, while benefits relate to the changes in health and environmental effects that stem from risk reduction.

Table 1.1: Information Provided or Included by the Various Assessment Tools

Information	Tool		
	CEA and CBA	**Macro**	**MCA[1]**
Direct and indirect impacts on industry	Y	Y	Y
Increases in costs of end products to consumers	Possibly	Y	Y
Change in costs of administration	Y	Y	Y
Changes within and between sectors	Possibly[2]	Y	Y
Economic efficiency	Y		Possibly[6]
Economy-wide price impacts		Y	Y
Distributional effects	Possibly[3]	Y	Y
Employment	Possibly[4]	Y	Y
Change in relative competitiveness	Possibly[4]	Y	Y
Environmental impacts (use and non-use values)	Y	Possibly[5]	Y
Improvement/deterioration in human health	Y	Possibly[5]	Y

1. MCA is not typically used to develop the information listed in the first column, but is able to incorporate such information into a single framework for use by decision makers in comparing alternative options.
2. It is possible within a partial equilibrium framework to consider the impacts of a few linked markets.
3. This can be accomplished with the use of distributional weights.
4. This is possible; however, it tends to be undertaken separately from the CBA and CEA.
5. Environmental quality matrices can be built into the analysis.
6. If information from a CBA or CEA is fed into the MCA.

1.3 The Risk Management Case Study

1.3.1 Introduction

As indicated above, in order to illustrate how the various tools are applied in practice, a hypothetical case study concerning the management of ZOZ is used in the sections that follow. ZOZ is an imaginary group of isomeric compounds which all share the same general formula. It is assumed here that the group of compounds was identified a few years ago as a priority for risk assessment and, potentially, risk reduction. This listing was based on the following:

- annual production volumes are large, with use being spread across a wide range of different applications;
- ZOZ has been found to be toxic to aquatic organisms and is not readily biodegradable;
- ZOZ appears to be widely dispersed throughout the environment; and
- use of ZOZ is expected to continue at current levels or increase in the future.

Given the above characteristics, there are obviously concerns over the risks that ZOZ's production and use may pose to the health of workers, to consumers of products containing ZOZ, to the general public, as well as to the environment.

For this case study, it is assumed that as work progressed on the preparation of environmental and human health risk assessments, input was required from the analysts responsible for preparing SEAs. In addition, it became clear that an assessment of the implications of adopting some form of risk reduction would be required:

- Section 2 examines how efficiency-based CEA and CBA might be applied to this case study;

- Section 3 discusses how macroeconomic modelling techniques could be applied; while

- Section 4 considers how MCA based techniques could be applied.

Because SEA should be seen as a process and not just as a series of techniques, it is important to also discuss here the types of activities which would be undertaken under the various Steps outlined in Part 1 of this document. Since Steps 1 and 2 should be viewed as common to all SEAs, these are considered first below, with the remaining Steps examined in Sections 2 to 4.

1.3.2 Step 1: Identifying the Problem

As will be recalled from Part 1, this first step in applying the suggested SEA framework relates to determining that there is a need for some form of risk reduction. In most cases, this will have been determined by the outputs of a risk assessment. However, these assessments require, as input data, information on:

- the manner in which the chemical(s) of concern is used;

- current levels of use across all applications and sectors of concern; and

- the substitute chemicals which already exist or are in development, and any alternative technologies which reduce the need for a chemical replacement.

So, for example, a market analysis would indicate the quantities of ZOZ used in different applications currently, in the past and expected in the future. Details would also be collected on the exact manner of use to ensure that the processes are well understood. This would be likely to include discussions with trade associations and individual companies, with the latter including both large and smaller operators to identify whether there are any significant differences in ZOZ use within a given sector.

With regard to the likely trends in use, both producers of ZOZ and downstream formulators of ZOZ based products may be valuable to highlighting where the market is going for the applications of concern. Similarly, producers of replacements for, or substitutes to ZOZ, are also likely to have views on trends in the sector with regard to the use of ZOZ, alternatives to it and technological development more generally.

The findings of the market analysis are likely to be important not only for identifying possible risk reduction options and assessing their implications, but also for the risk assessment. Such data are essential to ensuring that the assumptions used in the risk assessment provide a good reflection of the existing situation.

1.3.3 Step 2: Setting up the SEAs

Setting the Objectives

The findings of the risk assessment are likely to determine whether or not a full SEA will be undertaken for ZOZ. The next step in the process then is for decision makers to agree and specify the objectives for the SEA. For example, in the case of ZOZ, the objectives of the SEA might be as follows:

1. Identify the range of options that could be implemented as part of an overall risk reduction strategy; this should be done through consultation with both industry and non-industry interest groups who have expressed concern over the chemical.

2. Consider the costs (positive and negative) and benefits (positive and negative) associated with each risk reduction option, and provide an indication of the most appropriate form of risk reduction.

3. Indicate the implications of adopting the preferred option for industry/businesses, highlighting in particular the likely impacts on small and medium sized enterprises, consumers and regulators.

4. Examine the more practical implications of the strategy in terms of the practicability of its implementation, administrative requirements, monitorability and enforceability.

5. Highlight any key uncertainties surrounding the analysis.

6. Establish a series of recommendations concerning the implementation of the strategy.

The above objectives have been defined so as to provide flexibility in choosing the actual form of the SEA to be adopted following the screening of potential options. As highlighted in Part 1, however, such flexibility may not always be available as the regulatory/legal frameworks that exist in some countries may dictate or severely restrict the form of the analysis.

Identifying Options

Once the objectives have been defined, the next task is to identify potential risk reduction options, taking into account the activities during the life-cycle of ZOZ that give rise to human health and environmental risks of concern. It is likely that several different types of options can be identified, with their applicability varying across different life-cycle stages. Table 1.2 illustrates this point, setting out a generalised list of options for managing the risks associated with ZOZ.

Table 1.2: Potential Types of Options for ZOZ Risk Management

Types of Options	**Production of ZOZ**	**Production of Derivatives**	**Use of ZOZ-based Products**
Voluntary Approaches	✓	✓	✓
Employee Health & Safety			✓
Information Programmes			✓
Market Instruments		✓	✓
Command and Control	✓	✓	✓

Some of these types of options could take more than one form in risk management. For example, under the heading of ‘command and control’, options to manage human health and environmental risks could include:

- limits on the concentration of ZOZ in particular products;
- performance-based or criteria-based requirements;
- emission limits;
- best available technology requirements; or

– restrictions on marketing and use of ZOZ or ZOZ-based products.

In addition, the same option may be applicable to more than one life-cycle stage. For example, limiting concentrations of ZOZ in products may help reduce risks associated with both production and use, having benefits for workers, consumers of products containing ZOZ, the environment and the general public who may be exposed to ZOZ in the environment.

It is likely that some form of screening analysis would be undertaken to develop a short list for more detailed consideration by analysts. Such analyses generally require consideration of both the market data produced for the risk assessment and further data on technical feasibility, effectiveness, economic impact, change in environmental and/or health risks and stakeholder acceptability. Much of the information considered at this stage is likely to be qualitative in nature.

Based on the screening analysis, a more refined and specific list of options can be identified, such as that presented in Table 1.3. This short-list takes into account the practicality of an option (based on the screening criteria). There is no reason to expend effort in considering options in more detail if they are clearly unsuitable.

Table 1.3: Short-listed Options for examination in ZOZ SEA

Short-listed Options	Production of ZOZ	Production of Derivatives	Use of ZOZ-based Products
Voluntary approaches	✓	✓	✓
Worker training	✓	✓	✓
Improved protective clothing		✓	✓
Expanded product labelling			✓
Best available technology			✓
Marketing and use restrictions			✓

The next step in the SEA is a transitional one as it involves specifying in more detail the form of the full SEA. It is at this point that the process becomes more specific to the tool adopted.

2. TECHNIQUES FOR ASSESSING THE EFFICIENCY OF OPTIONS

2.1 Introduction

This section focuses solely on those SEA techniques that are concerned with analysis of the economic efficiency of proposed risk reduction options.

The two main forms of efficiency-based analysis are:

- cost-effectiveness analysis (CEA); and
- cost-benefit analysis (CBA).

These analyses are carried out at the sectoral (or individual company) level. This basically means that these approaches are based on the use of partial equilibrium analysis,[10] an economic term meaning that the focus is on specific markets and the analysis effectively 'freezes' economy-wide impacts (such as changes in employment or price levels). Within this type of approach, analysts do not need to concern themselves with impacts in inter-related markets, unless they are likely to be significant.

Economic efficiency-based analyses are contrasted with the macroeconomic level of analysis (discussed in Section 3) which examines the economic impacts of a policy or project for the economy as a whole. The key difference being that the macro level approaches are not designed to provide information on the efficiency of options. Instead, they focus on measuring economy level impacts, such as employment and trade, which may or may not be included in the efficiency based approaches. However, concepts of welfare can be used at the aggregate level but a balance is usually required in terms of level of resources used against relatively little added value.

In general, it should be possible to capture most economic impacts within a micro-level analysis based on CBA, but this may depend upon the nature of the impacts generated by a specific policy or project. Some degree of scoping may be required to determine whether an appraisal aimed at determining the efficiency implications of a policy at the sectoral level will be sufficient to indicate how widespread such impacts may also be at the macro level given inter-linkages between sectors. Clearly, where a policy may affect only a small number of markets then a partial equilibrium approach is preferred; such an approach may also be capable of providing information on the most significant inter-sectoral effects.

Assuming that economic efficiency is a key criterion, there is then the choice to undertake the analysis using either CEA or CBA. In some cases, this choice may be constrained by legal or other requirements. Where no such constraints exist, the decision is likely to be determined by the objectives of decision makers, the decision making context, and the nature of the risk management decision to be made. This section examines the application of CEA and CBA in more detail to provide a better understanding of what applying these approaches actually entails, the issues that may arise and the types of outputs they produce.

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[10] This technique was used by the economist Alfred Marshall (1842 – 1924) and ignores the effects of changes in the price of a commodity on all other related market places (including the prices of factors of production).

To achieve this, the discussion has been organised as follows:

- a review of the theoretical principles underlying the use of these techniques;
- an introduction to CEA;
- an introduction to CBA; and
- the application of CEA and CBA to the ZOZ case study.

2.2 Key Principles of Economic Appraisal

2.2.1 Overview

Economic appraisal (distinct from financial analysis) requires that resources are used in a manner so as to maximise 'total social welfare' (or to achieve the greatest net welfare gain to society). To assess whether this objective has been met requires that the social opportunity costs of resource use (see discussion on scarcity below) are compared to the economic benefits generated by those resources, with different patterns of benefits and costs being generated by different patterns of resource use.

The key principles underlying economic appraisal are that:

- resources are scarce and thus need to be allocated among competing uses;
- human demands, wants and needs are driven by preferences; and
- scarce resources can be allocated according to criteria designed to achieve specific goals and objectives in line with individuals' preferences.

2.2.2 Scarcity

The so-called 'factors of production' (*i.e.* land, labour, capital [and may also include entrepreneurship]) available to satisfy human needs and wants are limited. This state of affairs is referred to as scarcity, *i.e.* the needs and wants of society exceed the resources available to satisfy them. This means that choices have to be made as to which needs and wants to satisfy.

This concept of scarcity directly relates to the most important principle in economics, that of 'opportunity costs'. Opportunity costs reflect true economic values in terms of the value of a good/service in its next best alternative use. At a theoretical level, in markets that are 'perfectly competitive',[11] the supply price of goods and services will reflect the opportunity cost or the value of the resources used to produce the good/service in their next best use.

In other words, the concept of opportunity costs recognises that choices have to be made given that resources are scarce, and that there are competing uses to which resources could be put.

[11] The term 'perfectly competitive' is the theoretical basis of neo-classical economic theory and is characterised by certain key features:
1) there are a large number of buyers and sellers;
~~2)~~1) all firms aim to maximise profit;
~~3)~~1) there are no barriers to entry into the market; and
~~4)~~1) a homogenous good is traded in the market.

Consider an example of a risk reduction option that may cost £500 million to implement - this £500 million could be put to alternative uses that may result in a greater net gain to society (for example, by investing in a wide array of risk reduction options).

2.2.3 The Importance of Preferences

Neo-classical economics assumes that each individual is the best judge of what is best for him or herself, and is motivated by a desire to increase his/her own well-being. This means that the individual reacts ‘rationally’ and that these reactions can accurately be predicted. Individual choices are then driven by preferences, *i.e.* certain goods and services are preferred over others. It is these differing preferences that result in the prices of goods and services.

In moving from the individual to a societal level, there are three key assumptions:

- individual welfare can be measured (with this originally conceived in terms of units of utility [or ‘satisfaction’]), and conveniently reflected in the market prices paid for goods and services;

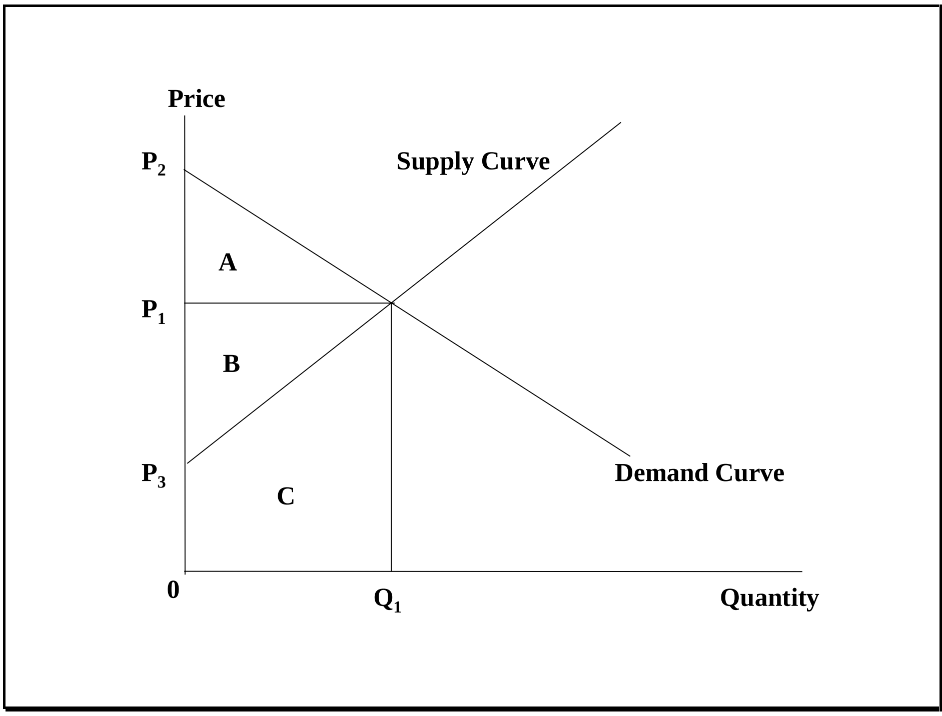
- individuals maximise their welfare by choosing the combination of goods, services and wealth that yields the greatest level of total utility; and

- societal welfare is the sum of individual welfare.

The first and second assumptions are based on the principle that utility (and thus welfare) can be obtained from goods and services (even if such items are provided free or at a minimum price). There are two components to the measurement of total utility. The first component is equal to an individual’s expenditure on a good, where this is given by the price and quantity of the product consumed. The second component is referred to as ‘consumer surplus’ and measures the maximum amount that an individual is actually willing to pay for that good or service (less actual expenditure).

Figure 2.1 provides a graphical representation of this concept based on a downward sloping demand curve (reflecting that as price falls consumers demand more of the good) and an upward sloping supply curve (reflecting suppliers’ desire to provide more of the good at higher prices). At the equilibrium point, supply equals demand (the intersection of equilibrium price, P_1, and equilibrium quantity, Q_1).

Figure 2.1: The Measurement of Total Utility and Consumer Surplus



Assuming that the actual price being paid for the good or service is P_1 and the quantity being supplied is Q_1, then total expenditure in this case is equal to the area B+C. Area B in isolation is referred to as the producer surplus, being defined as the difference between the price that the consumer actually pays for the good and the minimum price that the supplier is willing to sell at. The consumer surplus, being the difference between the price being paid and the maximum price that the consumer is willing to pay is area A. Total utility, in this case, is equal to the area A+B+C, which can be referred to as an economic measure, while the area B+C is a purely financial measure (as it only reflects actual expenditure). Finally, total surplus is given by the area A + B. It is worth noting, however, that the taxes which are usually present in most markets often distort the market and hence modify these findings.

2.2.4 Direct and Indirect Costs

For all appraisals, the cost side of the equation considers changes in producers' surplus. The extent of any changes in producer surplus will depend on the nature of the demand for the product and whether this (and the characteristics of the associated market) will enable costs to be passed on, hence protecting the supplier's producer surplus. However, it is inevitable that there will be some degree of cost burden associated with regulations and it is the aim of appraisals to calculate what these will be.

Appraisals will usually focus on the direct costs associated with the implementation of a policy in one, or a few, markets. The total direct (economic) cost of any policy is the cost of resources, as measured by their opportunity cost (*i.e.* their value when used in the next best option), employed by the 'producer(s)' over the life of the policy. This includes all costs imposed on third parties, where any externalities result directly from the policy.

The term producer, as used here, refers to the economic agent whose behaviour is the primary subject of the proposed policy, for example:

– the 'operator' of a fossil fuel power station who is required to install a scrubber system;

- the ‘owner’ of a vehicle who is required to meet a more stringent inspection and maintenance programme; or

- the ‘industrialist’ who is required to switch from the use of one chemical agent to another, *etc.*

Next comes the question of whether or not other related markets are affected and whether these impacts should be considered in the assessment. In general, most appraisals do not consider such effects as they are limited in scope due to data, time and resource constraints. The significance of such an omission, however, will vary over policies. The general consensus of the economics literature is that.[12]

- markets that are undisturbed by a policy intervention in another (related) market, *i.e.* in the sense that their supply and demand curves do not shift, do not have to be analysed;

- related markets that are affected by a policy intervention in the directly targeted market - in the sense that their demands shift in response to changes in the primary market - can also be ignored if:

  - prices in these related markets do *not* change (the supply curve is perfectly elastic); and

  - social and private costs[13] of the activities involved are *equal*;

- however, if prices in related markets do change (the consequence of an upward sloping, or less than perfectly elastic supply curve) in response to changes in the primary market, or if social and private costs in these markets diverge and quantities change, these markets ought to be examined.

On a practical level, it will be unrealistic in most cases to examine all related markets in order to identify less than perfectly elastic supply curves, or situations in which the social and private costs in these markets are unequal. At the same time, it is not acceptable to simply assume that social and private opportunity costs are equal, and that prices in related markets do not change.

2.2.5 Market Failure and Externalities

So-called ‘intangible’ impacts (including environmental and health effects) are special cases within the above framework. The failure of such goods to be valued stems very often from their ‘public good’ nature.[14] They fall into a category of assets for which either no, or only limited, markets exist in which they can be bought or sold. The absence of efficient markets means that there are no prices that can be relied upon to indicate the value attached to the good or service in question. As a result, they effectively become ‘free’ goods and tend to be treated as such, even though they may actually be highly valued.

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12 It is not possible within the scope of this study to present detailed arguments for the conclusions that follow: the reader is referred to Sugden and Williams (1978, p.134) or Arnold (1995, p.84).

13 Private costs reflect costs of a good or activity to the consumer or producer, whilst social costs are costs imposed on society as a whole (and usually include environmental externalities).

14 A commodity or service which if supplied to one person is available to others at no extra cost. A ‘pure’ public good has two key characteristics: non-rivalry and non-excludability. Examples of pubic goods include street lighting, clean air, clean water and so on.

In market economies, as stated above, prices are relied upon to provide signals regarding resource scarcity. They provide a true measure of economic value, and of an individual's willingness to pay when the market is characterised by a large number of buyers and sellers and where private property rights are enforceable and transferable. The choice to consume one good is made at the expense of consuming some other good. For environmental and health goods and services, such markets either do not exist or fail to perform efficiently, resulting in a so-called 'market failure'.

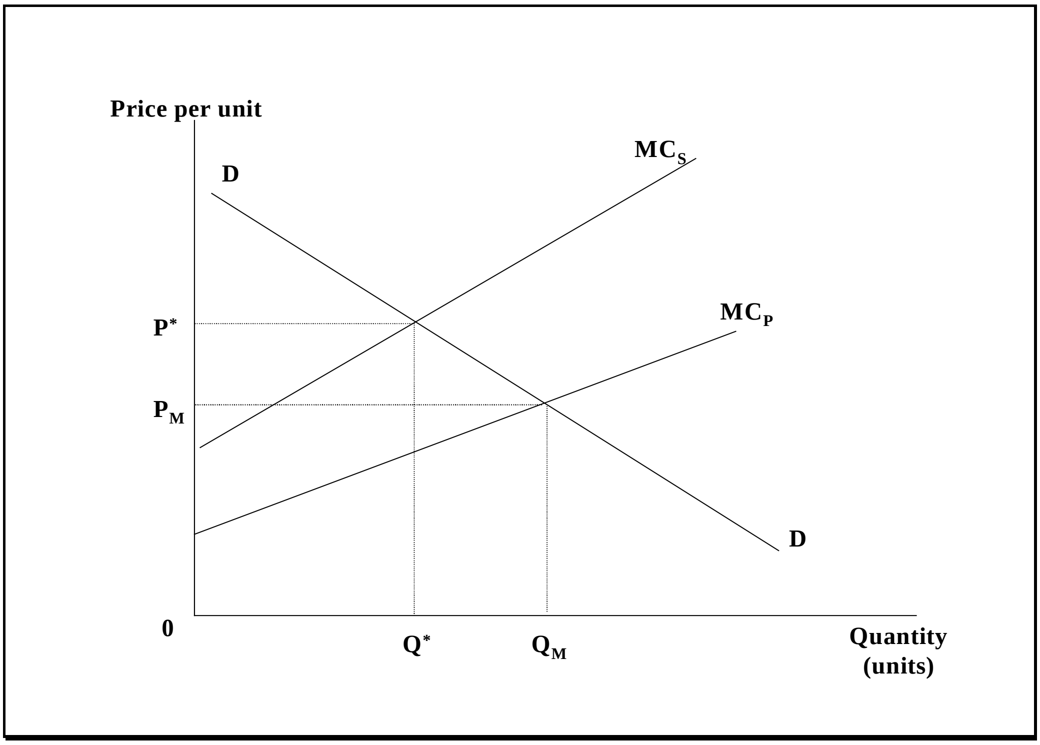
When a market failure results in human health and environmental effects remaining unpriced (or incorrectly priced), those effects are referred to as 'externalities'. Externalities may relate to positive impacts or negative impacts, although the more classical examples are of negative types. Externalities can be generally broken into two types, 'flow' and 'stock'. The difference is easier to explain by way of examples. Consider the emission of greenhouse gases; regulation tends to focus on the annual flow of greenhouse gases linked to climate change as opposed to the level of total atmospheric concentrations (*i.e.* the 'stock'). In a chemical risk management context, stock pollutants will tend to be those that are persistent, bioaccumulative and toxic. Regulation will generally focus on the flow of pollutants rather than their total stock in the environment.

Effluent discharges from industrial plants or municipal sewage treatment works provide another example of negative (flow) externalities, as such discharges may have negative impacts on water quality and thus on an adjacent nature reserve, or on recreational activities in the area, or on the health of downstream populations.

In such cases, because the full economic costs of the discharger producing a good and customers consuming that good are not properly reflected in its price, there is an inefficient allocation of resources. To correct this problem, the external social costs should be added to the discharger's private costs of production. Figure 2.2 provides such an example based on the modification of a firm's marginal cost curve (their supply curve).

The demand for the product in this particular industry is shown by the demand curve DD and the (private) marginal cost of production is depicted by MC_p. This firm also produces pollution and hence creates an additional cost to society - depicted by the curve MC_s. The market equilibrium price and quantity excluding the impact of pollution is P_mQ_m. However, the socially optimum price and quantity is P*Q*. The difference between the two marginal cost curves is referred to as the 'marginal external cost' of output in this case. This means that, in order to produce goods at the socially optimum level, the consumer price will be higher and the quantity of goods supplied will be lower.

Figure 2.2: External Costs at Firm Level



From the above, the following conclusions can be drawn:

- the output of the good is currently too great;
- the consumer prices of the products responsible for pollution are too low; and
- as long as the external costs lie outside the firm's decision making process, no incentives to search for ways to reduce pollution per unit of output will be introduced by the market.

As noted above, because many environmental goods and services are unpriced (in other words may be free to all), the full value of what consumers would be willing to pay for their consumption (use or protection) of these goods or services provides the measure of consumer surplus. To derive an estimate of willingness to pay or consumer surplus associated with such goods, however, requires the application of specialist economic valuation techniques (with the individual techniques discussed further in Part 3).

2.2.6 Total Economic Value

Given that environmental and health goods and services are special cases, environmental economists have developed a holistic valuation concept referred to as 'total economic value', with the aim of ensuring that all aspects of the value held towards such goods and services are taken into account.

The 'total economic value' (TEV) of an environmental asset (or a specific health state) is the sum of 'use values' and 'non-use values'. This split is, however, most often applied to environmental assets rather than health states. Use values are those associated with the benefits gained from actual use (now or at some point in the future) of the environment and may include private sector uses, recreational uses, education, science, amenity, and indirect uses. Non-use values (also known as 'passive use values') are not associated with actual or potential use of the environment, but solely with the knowledge that the asset is being protected and can be separated into two key types: 'bequest' and 'existence' values:[15]:

- **bequest values** relate to the desire of an individual to preserve environmental (and other) assets so that future generations may have use of them. For example, future generations' recreational or nature conservation needs; and

- **existence values** are defined as those values which result from an individual's (personal or altruistic) desire that an environmental asset is preserved and continues to exist into the future. For example, preserving biodiversity.

It is also possible to extend such principles to the health of individuals and society as a whole. For example, an individual may want good health care service in a country out of a desire to ensure that others in society will have access to such care. Society, therefore, increases its total welfare through knowledge of the provision of such services, assuming that individuals derive utility from good health.

When dealing with use and non-use values, care must be taken with regard to double counting. For example, it may not be appropriate to simply add the component parts to derive an overall value. Certain values may be embedded in others and analysts should examine the base data before such actions are undertaken.

2.2.7 Valuation of External Effects in the Context of Chemical Risks

The direct aim of chemical risk management is to reduce the risk of damage to ecosystems, resources or people more generally arising over the life-cycle of a chemical's use. However, the degree to which such direct and any associated indirect environmental and health benefits (or costs) can be valued in money terms and, hence, directly incorporated into a CBA will depend on the following:

- the existence of data linking exposure to effect;
- the availability of data on the stock or population at risk; and
- the existence of relevant valuation data where this addresses similar types of environmental quality changes and similar policy issues.

Should the above data not exist, then valuation will not be possible unless time and resources are spent on additional data collection and creation activities (such as the commissioning of policy specific studies). In such cases, the analyst is constrained to undertaking either a CEA or a CBA that includes non-monetary indicators of some of the costs and benefits. Either of these approaches may fail, however, to indicate whether a policy is actually justified from an economic efficiency perspective. Similarly, they may fail to indicate whether there are significant variations in the 'value' of the environmental and health gains stemming from different policy options (this being a particular issue where a policy concerns multi-pollutant effects and alternative options would result in varying impacts).

Where the above data exist, however, valuation may be possible, either through the use of policy specific willingness to pay values or through the use of benefit transfer techniques (see Part 3 for further discussion on these). The major constraints here are likely to be related to time and resources, as valuation exercises are themselves costly and take several months in elapsed time to complete.

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15 Two useful references for the interested reader are Lazo (1997) and Bateman & Langford (1997).

2.3 Cost-Effectiveness Analysis

2.3.1 General Approach

The aim of cost-effectiveness analysis (CEA) is to develop a ratio that indicates the costs of achieving a per unit change in a specified physical outcome. However, this definition assumes constant returns to scale (which is often false); to take this into account, CEA is sometimes defined in terms of finding the minimum cost of meeting a specified physical outcome. The numerator or 'cost' element represents the estimated resource costs of meeting the target or of adopting a particular option, while the denominator reflects the relevant physical outcome. Examples of such ratios are given by calculations of the cost per statistical life saved or the cost per unit of contaminant reduced.

There are two different approaches that can be adopted when using CEA:

- the first involves determining which option out of a set of competing options provides the least-cost approach to achieving a desired and pre-specified outcome; while
- the second involves calculation of the implicit economic value which would have to be placed on an end-outcome for a decision to be justified and comparison of this to other decisions or a maximum value.

2.3.2 Determining the Least-Cost Option

Under the first approach, the CEA can vary from being a simple comparison of two alternative options to a sophisticated analysis of alternative strategies involving different packages of risk reduction options (drawing on linear programming and other mathematical optimisation techniques). In either case, targets or goals (*e.g.* acceptable risk criteria or environmental critical load values) are set, with the CEA then aimed at finding the least cost abatement strategy. In general, the targets may be related to the minimisation of impacts associated with any stage in the life-cycle of a chemical's use that gives rise to unacceptable risks. For example, they may be specified in terms of reducing environmental concentrations below a particular level or reducing the number of illnesses or deaths to an acceptable number. In the more sophisticated analyses, a range of targets or constraints may be set in relation to time, costs, pollutant concentrations, *etc.*

Having set such targets, the CEA is then focused on determining the costs associated with the different technologies or strategies for meeting them. These results, when applied across a range of competing options, can be used to rank the different options in terms of the relative cost per unit of benefit.

The setting of strict targets in this manner negates the need for the analysis to also estimate the social benefits stemming from risk reduction (even though the benefits may vary between options and the differences in such variations may be important). However, underlying the target setting process is the assumption that the social benefits of meeting that target will outweigh the social costs.

2.3.3 Comparative Assessment of Options

Where risk reduction criteria are not pre-specified, CEA can be used to derive the implicit value that would have to be placed on a level of risk reduction for an action to be justified. CEA is used extensively in the field of health care for evaluating the outcomes and costs associated with different medical interventions. For example, CEA is used in such cases to determine the cost per life year gained. A variation of CEA based on assessment of the cost per quality adjusted life year (QALY) is also commonly used by health services (see Part 3, Section 3.4).

This type of approach can also be applied in chemical risk management, with the aim of indicating whether or not a policy is as cost-effective as other, previous policies aimed at achieving the same types of environmental or human health risk reductions. In particular, this type of comparative assessment has been applied within the context of a proposed policy's ability to deliver human health benefits. So for example, an appraisal might compare whether a policy delivers the same or a greater level of health improvement per unit of expenditure as previous policies. If a proposal does not generate similar (or greater) benefits per unit expenditure, then this may lead to questions over its relative efficiency compared to other policies.

Although the above example relates to the use of this type of assessment for new policy proposals, such assessments have also been undertaken in a retrospective manner, with the aim of highlighting potential mis-allocations of resources. A prime example of this type of application is given by the work undertaken by Tengs *et al* (1995) involving the comparison of some 500 'life-saving interventions' in terms of the cost per statistical life year saved associated with a range of different interventions, including health care, domestic/life style, transportation, occupational health and safety and environmental interventions.

2.3.4 Methodological Issues in Applying CEA

Care is required when applying CEA in either of the above manners. As pointed out by several authors (Tengs *et al*, 1995; Ramsberg & Sjöberg, 1997), such calculations can suffer from a number of problems, for example:

- when the cost estimates do not reflect the full social costs of the intervention as would be measured by opportunity costs, then alternative options may not be compared on an equal basis;

- where the proposed option would not achieve a continuous level of effectiveness per unit of expenditure (*e.g.* there is a limited number of individuals who can benefit from the proposed option), then comparing this option against others on an equal basis becomes difficult;

- when different options would lead to varying levels of risk reduction, with some options meeting targets and others falling short but involving significantly lower costs, conflicts may arise between strictly adhering to the target and finding an economically efficient solution; and

- when the proposed option has more than one target objective, for example, achieving health benefits in addition to saving lives, or environmental benefits across more than one environmental end-point, then options may vary in their cost-effectiveness with regard to different targets.

The latter factor, in particular, has often been highlighted as an issue with regard to the use of CEA. Where a number of benefits may arise from proposals, analysis of the relative efficiency of options in terms of just one indicator of performance may result in the selection of an option that performs well on that indicator but would generally be considered inferior across all other benefit categories. The alternatives may not perform as well on the key indicator but provide a greater level of benefits across all other indicators. This issue is likely to arise in many analyses concerning chemical risk management. Ways of dealing with this range from ignoring benefits other than the key target indicator, to supplementing the analysis with information on the other benefits (with this moving more towards a CBA), or adopting a multi-impact indicator of effectiveness using cost-utility techniques (see Section 3 of this Part for further discussion). The adoption of multi-criteria techniques (as discussed in below) allows the analyst to take into account these trade-offs between benefit categories.

There is an underlying assumption that the benefits of setting targets outweigh the costs of achieving them. This assumption gives rise to one of the key limitations concerning the use of CEA for regulatory options appraisal: it does not explicitly address the question of whether the benefits of regulation outweigh the costs (as does CBA). In other words, it does not consider how much society is willing or able to pay to obtain improvements in health or the environment. A second limitation is that CEA on its own provides no means of addressing equity and distributional issues which may be central to public concerns (although this applies equally to other techniques).

Other problems have arisen in the healthcare field over the failure of CEAs to adopt a common or standardised approach that would allow for the results of different studies to be compared. In particular, a panel on cost-effectiveness analysis stressed the importance of adopting a societal perspective when undertaking such analyses to ensure that estimates reflect the full resource costs of adopting a given option (Russell *et al,* 1996).

2.4 Cost-Benefit Analysis

2.4.1 The Aim of CBA

As indicated above, economic appraisal and, in particular, cost-benefit analysis provides the methodology underlying most of the current guidelines concerning the analysis of regulatory proposals where this includes chemical risk reduction strategies.

The aim of CBA is to determine whether an investment is worthwhile from an economic efficiency perspective, with this driving the requirement to place a monetary value on as many of the impacts of a proposed option as possible. The underlying assumption is that by valuing all of an option's effects in economic opportunity cost terms, one can determine the trade-offs that society is willing to make in the allocation of resources among competing demands.[16] As a result, CBA indicates whether or not a particular option is 'justified' in that the benefits to society outweigh the costs to society, and allows a comparison of alternative options to be made on this basis.

As the analysis of potential risk management options is only concerned with the additional or incremental costs and benefits arising from a proposed alternative, the aim in CBA is to calculate the monetary value of the changes that would occur under a particular regulatory option compared with the current situation (or base case). These may include changes in:

- costs (or savings) stemming from changes in production, use and consumption of the hazardous substance under examination; for example, in the case of metals this may include costs arising from mining, smelting, formulation of a metal-containing substance, use of the substance, purchase of the end product and its disposal;

- human health effects, where these include acute and chronic effects, from non-severe illnesses to increased levels of mortality, and impacts on different segments of the population (workers, the general public, specific sensitive groups); and

- environmental effects, where these include direct effects on ecological systems and on society's direct use of the environment (whether now or in the future) and those effects which are of a concern from conservation and preservation perspectives.

[16] The preferred measure of economic impact is the opportunity cost of the resources used or the benefits forgone as a result of the regulatory action.

For CBA, the same approach is taken to estimating the costs of introducing an option as for CEA. In valuing human health and environmental effects, analysts will need to also call upon the range of valuation techniques used to derive monetary valuations for human health and environmental effects, with value being defined by individuals' preferences. These preferences are obtained by measuring either individuals' (and hence society's) *willingness to pay* for a benefit (or to avoid damages) or individuals' *willingness to accept* compensation for a loss. The techniques which are used to derive these measures involve analysis of the market value of gains and losses, of individuals' *revealed* preferences or of individuals' *stated* preferences. A brief outline of these techniques are set out below with further detail provided in Part 3.

Market Effects

For some types of effects, willingness to pay can be derived directly by estimating the value of gains or losses resulting from an environmental change by linking market and production data to dose-response functions. An example is the linking of changes in air quality to changes in crop yields and hence crop value.

Revealed Preferences

These preferences are revealed through actual choices made by individuals in the marketplace. In this case, a value is inferred from expenditure on goods or services that embody the health or environmental characteristic of concern; such goods may be substitutes for, or complements to, the health or environmental attribute (*e.g.* organic food and pesticides, or property prices and clean air). The techniques included under this heading are:

- avertive expenditure (which relies on the estimation of expenditure on substitute goods);
- hedonic pricing (which is based on the concept that the price paid for a complementary good implicitly reflects the buyer's willingness to pay); and
- travel cost method (which is based on the concept that people spend time and money travelling to a site and that these expenditures, or costs, can be treated as revealing the demand for the site).

Stated Preferences

These are the preferences stated by individuals when directly asked through surveys to value an environmental or health good within a hypothetical market setting. In these cases, questionnaires are used to construct an experimental or hypothetical market and to provide the basis for eliciting an individual's willingness to pay. The two key techniques are:

- contingent valuation; and
- attribute based methods such as contingent ranking/conjoint analysis.

More detailed information on valuation is provided in Part 3.

2.4.2 Key Issues

It is important to remember that it is unlikely that all of the costs and benefits arising from a risk reduction option can be quantified and valued in monetary terms, owing mainly to a lack of the necessary data (as highlighted earlier). This is particularly true with regard to impacts on human health and the environment, although it is not restricted to such effects and may also affect the assessment of impacts on the target sector, related markets, consumers, regulators and others.

With regard to chemical risk management, a CBA that involves monetary valuation of only the costs of risk management is sometimes termed a risk-benefit analysis. In this case, 'risk' relates to the environmental or human health damage arising from a chemical's use, while 'benefit' relates to the reduced costs (*e.g.* to industry and consumers) associated with continued use of the chemical of concern. This is the phrase adopted, for example, in the EC *Technical Guidance on Development of Risk Reduction Strategies* (EC, 1998) and results in part from the nature of the outputs of the preceding risk assessment which are in the form of the ratio of predicted environmental concentrations to predicted no effect concentrations. Where fuller probabilistic risk assessments are undertaken, the monetary valuation of benefits is more likely to take place (for example, as is often the case in Canada and the US).

2.5 Application to the ZOZ Case Study

2.5.1 Introduction

The manner in which CEA and CBA would be applied to assessing alternative risk management options for ZOZ is examined below. This discussion follows on from the introduction presented in Section 1.3, starting with a discussion on the types of activities that would be undertaken in Steps 3 and 4 of the SEA.

2.5.2 Step 3: Undertaking the SEA

The last task in the second step of SEA as set out in Part 1 is to use the knowledge gained from the preliminary assessment of impacts and options to refine the objectives for the more detailed analysis. Because CEA and CBA differ in terms of their requirements, so too do the objectives of the economic analysis. For example, the following objectives might be set for the SEA:

- Develop estimates of the costs associated with each risk reduction option and determine the most cost-effective strategy for reducing the risks associated with ZOZ to acceptable levels. This is to be interpreted as reaching environmental concentrations that are below the Predicted No Effect Concentration (PNEC) for both the aquatic and terrestrial compartments.

- Provide details of the likely benefits to the environment and human health for each of the risk reduction options under consideration. To the degree possible, quantitative estimates of the benefits should be provided; where benefits cannot be quantified, qualitative descriptions should be given of the benefits, highlighting the likely severity and scale of effect arising from individual options. Indicate the trade-offs in social cost/benefit terms of selecting one (set of) options over another. Determine the option or combination of options which delivers the greatest net social benefit, by balancing the social costs and social benefits.

For CEA, the overriding objective is to provide an indication of the risk reduction option that would be the most cost-effective in reducing risks to the environment. The key here is that decision makers have effectively set target levels of risk reduction, specifying that for options to be effective they must reduce environmental concentrations to below the PNEC. Given the setting of such strict targets, the use of a CEA-based approach is appropriate.

The objectives as specified with CBA in mind vary from those for the CEA in terms of the scope of the economic efficiency considerations. For the CEA, such considerations are restricted to examination of the relative costs of proposed options. For the CBA, efficiency is a key consideration, relating to whether or not a balance is provided between costs and benefits, with the aim being the identification of the option delivering the highest net social benefits (*i.e.* benefits minus or net of the costs).

2.5.3 Specification of Risk Management Options

A list of potential risk reduction options was identified earlier. Given that the objectives of the SEA differ, the options examined in detail may also differ. A CEA is likely to focus on the options that are likely to achieve the target levels of risk reduction (i.e. to levels equal to or below the PNEC and No Observable Effect Level [NOEL]). No such constraints are set on the options that would be examined by the CBA, with this then potentially providing a wider array of possible risk reduction options. The differences that could arise in terms of the variety of risk reduction options considered in the CEA could have implications not only for the efficiency of any end solution but also for the distribution of costs and benefits arising from risk management.

As part of this task, analysts would also have to establish the baseline to be used in the analysis. This baseline should be the same for both the CEA and CBA, taking into account the results of the market analysis and any legal or other regulatory options that may affect risk generating activities in the future. Similarly, the time period to be adopted in the analysis would need to be determined, with this most appropriately set in terms of the predicted commercial life for ZOZ, taking into account the on-going development of technologies and environmentally friendly chemicals for the sectors and applications of concern.

2.5.4 Data Collection

For both the CEA and the CBA, a range of data sources is likely to be called upon. However, the scope of the data collection activities will vary significantly between the two types of analysis. Essentially, the CEA will focus on gathering information relevant to estimating the costs of implementing each of the potential risk reduction options (for example, the changes in capital and operating costs), while data collection for the CBA will be extended to include gathering information on the benefits arising from each option. Table 2.1 illustrates different sources that may be called upon and the techniques that may be used as part of data collection.

Table 2.1: Possible Data Sources for ZOZ CEA and CBA

Data Source	**Data Collection Techniques**
Industry (small, medium and large operators)	Direct consultation, surveys/questionnaires, written requests for information.
Trade associations	
Consumer groups	
Environmental groups	
Representatives of populations at risk	
Regulatory agencies/departments	
Other Governments with an interest	
Statistical offices/databases on company size, output, imports and exports, employment, *etc.*	Data searches.
Databases, *etc.* holding scientific information	
Specialist sources in fields related to risk reduction	Direct consultation, literature review.
Specialist sources in fields related to benefit assessment	
General public	Contingent valuation surveys, revealed preferences, market data.

Of particular note is that data collection as part of a CBA is likely to involve using, or drawing on previous applications of various economic valuation techniques in order to derive estimates of the general public's willingness to pay for environmental/health benefits (or willingness to accept compensation for damages). As noted in the table, the sources of the data will vary depending on the technique used.

2.5.5 Data Analysis and Options Appraisal

As data are collected they will need to be analysed to identify any new data requirements and to ensure that the data being collected are as reliable as possible. The data will also have to be combined to make a comparative appraisal of the proposed options feasible.

For a CBA, these latter activities are likely to include:

– developing scenarios on how a proposed option would be implemented;

– modelling the various scenarios to estimate the costs to the target sectors associated with each of the scenarios; preferably, these models will be based on calculations of changes in producer surplus but often will rely on financial cost estimates;

– modelling the various scenarios to predict changes in environmental and human health effects stemming from a given option; these predictions would then be linked to models developed to estimate the economic value of the changes in effect;

– setting out the timing of the predicted costs (by cost item) and benefits (by benefit type) to enable discounting and allow calculation of present value figures for both positive and negative costs and positive and negative benefits (*i.e.* where the present value is given by the sum of discounted costs/benefits over the analysis timeframe);

– derivation of net present value estimates (equal to the present value costs minus present value benefits) for each of the scenarios;

– carrying out a sensitivity analysis of key scenario and model assumptions to better understand the key uncertainties underlying the results; and

– based on the above, identifying the trade-offs associated with choosing one option over another, taking into account the implications of the sensitivity analysis.

Much of the above work would also be necessary in preparing a CEA. The key difference would, of course, relate to the need to predict the benefits of the expected changes in environmental and human health effects. Although analysts would need to predict the level of environmental and health improvements that would be achieved for each option, the economic value of these predicted benefits would not need to be estimated for a CEA.

The above list of activities is obviously a simplification, with there being a number of issues to be addressed within each of the above steps. For example, for our hypothetical chemical group ZOZ, separate scenarios would have to be developed for each target sector based on the options which would apply to their applications and how the options would be implemented by that sector. In some cases, this may require the development of separate scenarios for small, medium and large companies as their likely responses may vary considerably. The analysis would also involve tracing the impacts through the chain of trade for ZOZ to ensure that the cost implications to all the various parties were included in the analysis.

The type of analysis that might be undertaken in this regard is illustrated in Figure 2.3. For this figure it is assumed that the introduction of a risk reduction option would increase the costs of production in a manner that would restrict the supply of ZOZ. This would lead to a shift in the supply curve for ZOZ from S to S_1 (in the graph presented in the upper left quadrant of the figure), thereby increasing the price at which ZOZ would be supplied from P to P_1. This increase in price would then reduce the level of demand for ZOZ by those using it as an input to production, who in turn would supply fewer ZOZ-containing products to consumers at an increased price (as illustrated by movement from point 0Q to $0Q_1$ in the graph given in the lower right quadrant). From the above interactions, the aggregate demand curve AD, illustrated in the lower left quadrant, would be derived leading to the following results for this example:

– the change in consumer surplus is P_1BP_2 minus PAP_2;

– the change in producer surplus is P_1BP minus PAC;

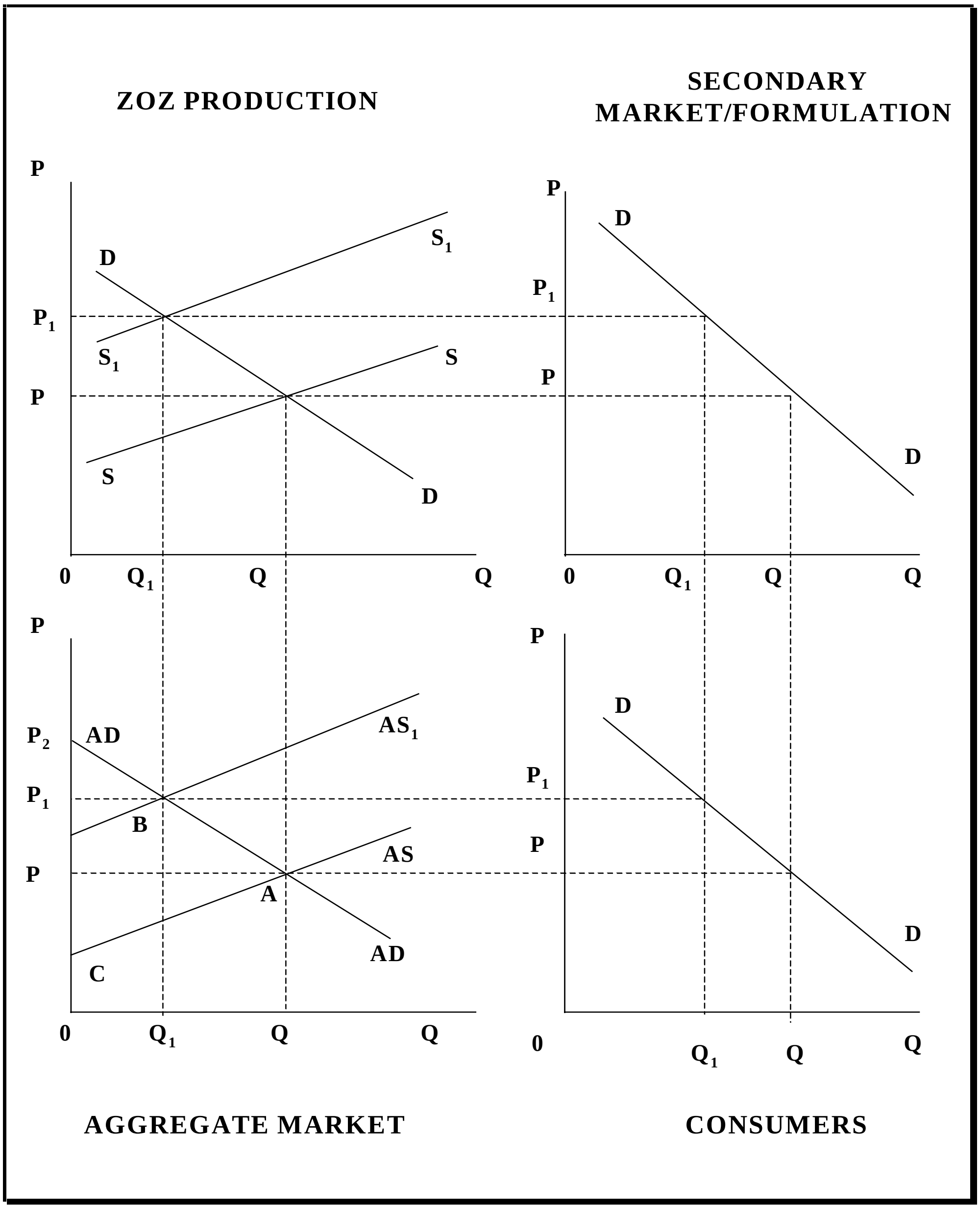
– the change in producer surplus is much greater than the change in consumer surplus (given the existence of substitutes and the fact that ZOZ is only one factor of production in the end product);[17] and

– the change in total welfare is P_2BP minus P_2AC.

In this case, the area indicating total welfare losses is large, with much of the incidence falling on the producers of ZOZ. The economic impacts to this sector may therefore be significant.

17 Note that although this conclusion holds for this example, it may not in other cases. The magnitude of the changes in producer and consumer surplus depend on the relative price elasticities of demand and supply.

Figure 2.3: Supply and Demand Model Used in ZOZ CBA[18]



However, such effects may be displaced by the development and production of substitutes for ZOZ, efficiency gains brought about by technical innovation *etc.*, which could have major impacts on the actual economic losses arising from the proposed options.

In addition to any displacement of efficiency gains, consideration would have to be given to any impacts that the adoption of substitute chemicals for ZOZ or alternative production technologies might have on the environment or human health. Similarly, the potential for technical innovation to lead to new technologies or to new products (replacing the need for the ZOZ-based products) should be considered.

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[18] Where: P = price, Q = quantity, D = demand, S = supply, AD = aggregate demand, AS = aggregate supply.

Given that the options (and their cost and benefit implications) are likely to vary across sectors and applications, it is likely that for both the CEA and CBA separate assessments would be prepared for each option and sector/application combination. This type of approach may be important to ensuring that individual industry sectors would not be required to undertake a disproportionate level of risk reduction. In addition, information would be sought on the distribution of costs across different company sizes (*i.e.* small, medium and large).

As indicated above, in order to compare costs and benefits, discounting would be used to convert the streams of future costs and future benefits to current values. Within the CEA, the present value of costs would act as the numerator in calculation of the cost-effectiveness ratios, while the denominator would relate to whether or not the target PNEC value was realised as indicated by the tonnage per annum of ZOZ no longer discharged to the environment or the number of health effects (*e.g.* deaths or cases of illness) avoided. The lower the cost per tonne reduced (or cost per health effect avoided), the more cost-effective the option and the higher the ranking it would be assigned relative to the alternative options.

The main decision criterion used in the CBA is the net present value (NPV) of each option, which is found by summing the streams of discounted costs and discounted benefits over the time horizon of the analysis. The NPV provides the basic measure of the economic efficiency gains (or losses) of a proposed option: a positive NPV indicating that the benefits outweigh the costs and thus that action is economically justified and a negative NPV indicating the reverse (for formulae and more details on discounting see Part 4). Within the ZOZ risk management context, a positive NPV indicates that the environmental and health benefits outweigh the social costs and that risk reduction is justified.

However, relying on the NPV alone to determine whether or not a proposed option is justified is unlikely to be possible in most cases, owing to the inability to place a money value on all environmental and health effects. Consideration should also be given to any non-valued effects to ensure that they are taken into account in the comparison of options. One way of doing this is through the calculation of implicit values which, in the case of ZOZ, would indicate how high an economic value would have to be assigned to the non-valued benefits (for example related to impacts on the terrestrial environment and related food chain) for these to be equal to, or greater than, the costs to industry and consumers.

Problems may also be introduced if decision makers are faced with a number of differing policies with positive NPVs. In this case, some sort of ranking is required in order to choose between the (often) competing policies. In a public policy context, it is likely that funds are limited and another decision criterion may be introduced, that of the benefit-cost ratio. Given a range of polices and limited funds, the benefit-cost ratio gives a better indication as to the ‘return’ generated by spending public funds, *i.e.* the policy with the highest benefit-cost ratio will usually be the one that will receive the funds first (unless it is economically justifiable to move to a more costly, but more beneficial, policy).

The options appraisal phase of both the CEA and CBA is also likely to address other issues of relevance to decision makers. This might include an assessment of distributional effects and impacts on trade and competitiveness. Such an assessment could be either qualitative in nature or be quantitative (see Part 3 for a more detailed discussion) and is likely to cover indicators such as:

- income per person;
- balance of trade;
- profitability;
- changes in sectoral growth; and

– trade intensity share of exports.

2.5.6 Step 4: Making Recommendations

The final step involves preparing a summary of the findings of the assessment. For CEA and CBA this is likely to include:

– highlighting the trade-offs involved in adopting the alternative options, where this includes consideration of both valued and non-valued effects (where the latter may relate to the achievement of different target levels in the case of CEA);

– indicating what options perform the ‘best’ in terms of cost-effectiveness ratios or NPVs;

– setting out the implications of uncertainty with regard to the calculated CE ratios and NPV estimates and, hence, to the ranking of the different options; this would probably be accompanied by calculations indicating under what assumptions a second-best option would become preferred;

– making recommendations as to any further analyses (*e.g.* further work on the monetary valuation of environmental benefits) that should be carried out prior to making a decision; and

– suggesting actions that could be taken to reduce the impacts of risk reduction on key sectors or types of companies.

3. TOOLS FOR ASSESSING MACROECONOMIC IMPACTS

3.1 Introduction

The previous section reviewed the tools aimed at assessing the economic efficiency of proposed risk management options, with the two key forms of analysis being CEA and CBA. For practical reasons, such analyses are generally focused on the target sectors and, potentially, a few of the key related markets. They are 'partial equilibrium' analyses in that they assume all other sectors will remain unaffected (or that the effects will be marginal).

In some cases, however, the actual effects of a policy on the structure and functioning of the economy as a whole may be significant, and should therefore be taken into account. In such cases, the use of so-called 'general equilibrium' methods are required, in which a more sectoral or macroeconomic approach to modelling is adopted, whereby the interactions between economic agents in the economy are explicitly taken into account. It should be noted, however, that these methods do not aim to assess costs and benefits as in a partial equilibrium sense, but they can capture those impacts that may not be seen as net costs or benefits. As such, they can be seen as additional or wider approaches to assessing the effects of a particular risk reduction option.

Input-output (I-O) and computational general equilibrium (CGE) models are the most commonly used forms of macroeconomic models. Both recognise that the implementation of new regulations by individual companies affects their behaviour as 'buyers' and 'sellers', which in turn affects their interactions with other companies in the same sector. In other words, the implementation of risk management options at the company level will also have impacts at the sectoral-level, the next highest economic accounting level to which individual companies belong. Sectors also act as 'buyers' and 'sellers'; thus, the introduction of a new regulation that affects a sector as a whole can affect interactions between sectors, and ultimately the functioning of the entire economy. The intra-actions between companies in the same sector, and the inter-actions between different sectors, encompass the direct and indirect effects of implementing a risk reduction option (as discussed in Section 2 above). Where such intra and inter-actions are significant, then a post regulatory equilibrium may need to be determined for the economy generally (Cal & Holahan, 1983 as reported in Zerbe and Dively, 1994).

To illustrate the distinction between a partial equilibrium based approach and that taken by I-O and CGE models, assume a carbon tax is imposed on petrol. It is likely that the imposition of such a tax will have impacts beyond the petrol market.[19] First, the tax serves to raise the price of petrol, which in turn will induce shifts in demand curves in other markets. Second, the prices of other goods and services whose supply curves are upward sloping, will change, inducing second-round effects on the demand for petrol. Third, primary inputs will be reallocated across the economy as the production of goods and services changes. This, in turn, will affect the earnings attributable to different factors of production (*e.g.* labour, capital, *etc.*). Finally, since different agents in the

[19] These indirect effects arise from the complementary relationships between the demand and supply of petrol and that of other goods, *e.g.* automobiles, and the ability to substitute such goods with others (e.g. use of non-fossil fuel reliant transport or equipment). Two goods are complementary in demand if a reduction in the price of one good causes an increase in the demand for the other, and two goods are substitutes in demand if a reduction in the price of one good causes a decrease in the demand for the other.

economy may not have the same marginal propensity to save/consume (*i.e.* the use that an additional unit of income is put to), the government may change the pattern of relative demand for different goods and services when redistributing the proceeds of the tax across different economic agents,. These changes result in a new vector of consumption and product prices, which directly affect the rate of productive capital formation, technological innovation, labour supply, and the economy's dynamic growth path.

When these effects are significant, partial equilibrium calculations of the costs and benefits of the tax will give a very poor approximation of the overall impacts of the tax policy. In other words, even a complete and correct CBA will not necessarily tell decision makers everything they may need to know. As noted by Mishan (1994):

> "the context of (standard) cost-benefit analysis is that of partial equilibrium analysis, one in which we concentrate on the valuation of several items on the assumption that the effects of consequent changes in the prices of all but the most closely related goods or bads may be neglected as we vary the amounts or introduce any one of several items..."

In other words, the assumption of *ceteris paribus* ('other things being equal'), which underlies the use of a partial equilibrium approach such as those adopted in conventional CBAs and CEAs becomes invalid. Thus, when a policy induces significant changes, macroeconomic models - because they explicitly model the interactions between markets - give a relatively more accurate picture of the overall impact of a policy than would be obtained through conventional (partial equilibrium) appraisal. It must be made clear, however, that while the tools discussed in this section do not do a 'better' job than the efficiency-based methods, they do something entirely different.

It is important to recognise that the outputs of both I-O and CGE models are effectively quantitative descriptions of the changes that would take place within an economy and there is generally no widely accepted single criterion by which to judge the relative performance of different options. Concepts of welfare can be used at the aggregate level but a balance is usually required in term of level of resources used against relatively little added value. In order to determine whether the losses and gains predicted as occurring across an economy are justified, one would usually need to incorporate such data into a CBA based framework.

3.2 Input-Output Models

3.2.1 Introduction

Wassily Leontief developed the first set of basic input-output tables in 1936 for the US economy. Since then, these models have been developed for a number of different countries and regions, and to address not only economic linkages but also economic-environment and economic-employment linkages. The primary purpose of economic-environment models has been to forecast residual discharges, mainly air emissions, with examples listed in Table 3.1.

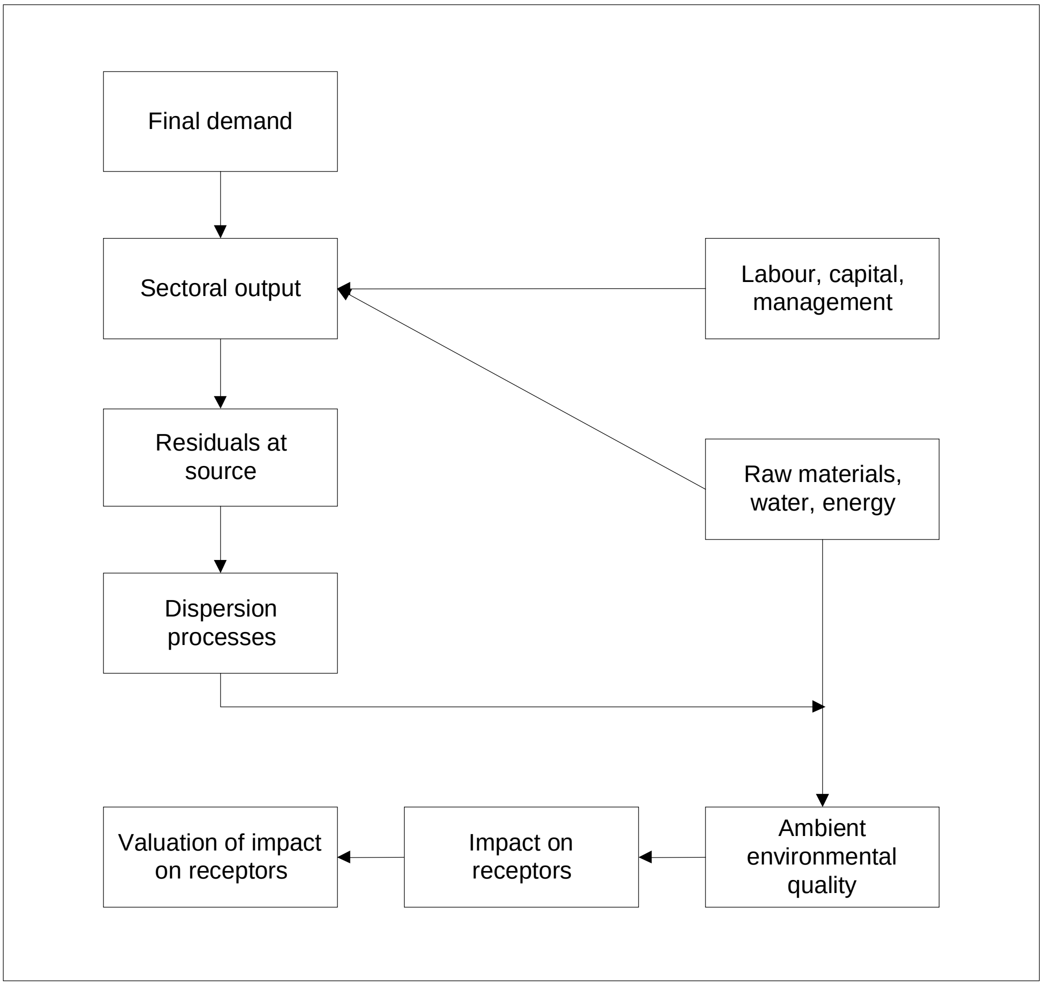
Table 3.1: Examples of Economic-Environmental Input-Output Models		
Author and Year	**Country/Region**	**Residual(s) Modelled**
Leontief & Ford (1972)	United States	Air emissions
Ayres & Gutmanis (1972)	United States	Air emissions/solid wastes
Cumberland & Stram (1974)	United States	Air emissions
Ridker (1972)	United States	Air emissions
US EPA SEAS model	United States	Air emissions
OECD (1978)	Japan	Air emissions
Jansen *et al* (1978)	The Netherlands	Air emissions
Victor (1972)	Canada	Air emissions/water discharges
Forsund and Strom (1976)	Norway	Air emissions
Shefer (1973)	Haifa Bay (US)	Air emissions
James (1982)	Australia	Air emissions
Proops *et al* (1996)	United Kingdom	Air emissions
De Haan (1996)	The Netherlands	Air pollution abatement
Hite & Laurent (1972)	Charleston (US)	Air emissions/water discharges
Miernyk & Sears (1974)	West Virginia (US)	Air pollution abatement
Koppl *et al* (1996)	Austria	Energy taxation
Source: EC, 1996		

I-O models provide a systematic description of the interdependencies that exist between sectors in the economy. They are essentially matrices that represent the flow of goods to and from each sector in the economy. For any particular sector, the vector of inputs from other sectors is recorded (*e.g.* the energy, raw materials, engineering and other goods to produce one unit of chemicals production), as are the outputs produced by that sector that are distributed either as final consumption or as inputs to the other sectors (*e.g.* the demand for chemicals in the production of energy, engineering and other goods). Because every output may be used (potentially at least) as an input to every other sector, as well as satisfying final demand, there is no single connection between changes in production and changes in consumption.

An input-output matrix can be used to estimate for any one sector how much input from other sectors (and in what proportions) is required to produce a unit of output. As a result, such a matrix can be used to examine how changes in the total output of one sector (or in final demand covering expenditure by households and government) are likely to impact on the demand for inputs from other sectors. I-O models are tools for projecting the magnitude of impacts, but they do not give an indication as to the change in net costs or benefits.

It is relatively straightforward to expand an economic I-O model to incorporate discharges of residuals and the consumption of environmental resources; they are simply treated as primary inputs. With the addition of natural-systems models, for example air dispersion models, it is possible to link residual discharges to ambient environmental quality, and subsequently to environmental damage. An economic-environmental quality model is shown in Figure 3.1.

Figure 3.1 Basic Input-Output Model for Economic-Environmental Quality



In addition to forecasting discharges of residuals (*i.e.* emissions), input-output models can be used to assess both the direct and indirect effects of controlling flows of residuals from economic activities. This is done by incorporating emissions into the model, by working out the contributions (direct and indirect) of each sector to total emissions. By manipulating levels of sectoral output, the structural adjustments required to meet target reductions in emissions (*e.g.* 20% decrease) can be determined (see Frankhauser & McCoy (1995) for further discussion). The impact on the whole economy can then be estimated with the use of a Leontief inverse matrix, including *direct* and *indirect effects* on final demands, sectoral outputs and primary inputs.

It is also possible to use I-O models to quantify (demand-side) direct and indirect employment effects. Once final sectoral output has been determined, these figures can be translated into employment. This is accomplished by constructing an industry-occupation matrix and corresponding employment/output coefficients from data on manpower requirements, man-hours, and productivity within each sector (OECD, 1997). The employment/output coefficients are treated in the same ways as the environmental quality coefficients discussed in the previous sub-section.

Input-output models may be used to compare two distinct states of the economy: pre-policy intervention versus post-policy intervention. The difference between the two ‘states’ represents the gains or losses to the economy (*i.e.* the positive and negative costs) of implementing the policy in question (normally expressed in terms of a change in GDP). The net cost to the economy can then be combined with estimates of the compliance costs falling on target sectors for inclusion in an expanded economic analysis (*e.g.* CBA-based framework); although care obviously needs to be taken to ensure that double counting of costs does not take place.

3.2.2 Limitations of Economic-Environmental Input-Output Models

While I-O models can be used to describe the interconnected nature of production in an economy, several key issues arise in their application to environmental issues. A realistic regional model may have 30 or 40 sectors, while a national model may have anywhere between 100 and 300 sectors. An obvious drawback of input-output models is the amount of time and effort required to collect the base data and to build the model. Constructing even a modest regional model requires several person-years of effort, not to mention a great deal of co-operation from industry and government (Hufschmidt *et al,* 1990). However, it may be possible to use ‘off the shelf’ models in order to conduct approximate analyses.

In addition, assumptions underlying the construction and operation of input-output models are subject to criticism; for example, the following comments have been made concerning economic-environment models (OECD, 1997):

- as prices are not incorporated into these models, it is not possible to determine how demand for one sector’s output may respond to changes in price;

- because of this failure to take into account the impact of price effects on the demand for different inputs (factors of production), I-O models ignore several relevant types of indirect effects that may impact on emission estimates (*e.g.* induced consumption effects, induced investment effects);

- there is no allowance for flexibility in production, as the assumptions concerning the inputs needed for a given unit of production are fixed; this raises further problems in economic-environment models, because the use of fixed coefficients will not accurately describe real production relationships (*i.e.* level of inputs per unit of output) or environmental quality effects, especially when non-marginal changes in output are anticipated;

- the use of fixed production relationships also implies that the average unit of output requires the same proportional mix of inputs as does a marginal unit of output; and

- the extension of the fixed coefficient assumptions into emissions modelling may not adequately handle background concentrations of residuals and threshold effects (Hufschmidt *et al,* 1990).

Economic-employment models have also been criticised for neglecting several relevant channels of indirect employment effects, including those stemming from price and wage adjustments, induced consumption effects of the incremental employment (*i.e.* the multiplier effects), and induced investment or accelerator effects (OECD, 1997).

3.3 General Equilibrium Models

3.3.1 Introduction

Applied, or computational, general equilibrium (CGE) models are the most sophisticated type of general equilibrium approach used to evaluate the macroeconomic impacts of implementing a proposed environmental policy. They are capable of quantifying direct and indirect effects of environmental policies on economic structure and product mix, economic growth, the allocation of resources and the distribution of income. Moreover, as CGE models consider both supply and demand interactions, (in contrast to 'neo-Keynesian' approaches which focus more on demand), they are capable of dealing with longer planning horizons. Consequently, the analyst can examine long-term movements in economic variables. It is worth noting that recent developments in CGE modelling have been in integrating top-down and bottom-up approaches which may have uses in chemical risk management (see, for example, Bohringer (1998)).

These models are based on the concept underlying I-O models (which may also provide useful information on equity or distributional effects), but the system is completed by including all relationships needed to represent the entire circular flow in the economy, and by endogenising choice by giving some of the relationships a flexible functional form based on individual optimising responses to prices. Thus production of each sector becomes a function of input prices and the output price, consumption becomes a function of income and prices, and those prices are determined endogenously within the model. The model is solved to find the level of prices, consumption and production such that quantity supplied is equal to quantity demanded in all markets.

Although there are many examples of CGE models, Zerbe and Dively (1994) suggest that the best 'thought-out' models will have the following elements.[20]

- a description of the utility functions and budget constraints of each household in the economy;
- a description of the production functions of each company in the economy;
- the government's budget constraint;
- a description of the resource constraints of the economy; and
- assumptions relating to the behaviour of households and companies in the economy.

There are also variations in how a CGE analysis is conducted, although most analyses involve the following basic steps (Gramlich, 1990):

1) the baseline, or pre-policy change world, is represented by a system of empirical equations describing demand and supply in all relevant markets. This model is subsequently solved, usually by computer, to yield a pre-policy vector of production and consumption prices;

2) the proposed policy change is then modelled by shifting the supply and demand curves appropriately;

3) the model is re-solved, yielding a new vector of production and consumption prices; and

[20] One could add international trade to this list. Factor mobility for the last two bullet points may also merit consideration.

4) finally, the overall effect of the proposed policy is determined by examining the difference between pre- and post-policy vectors of prices.

As indicated, CGE models compare two distinct states of the economy: pre-policy versus post-policy. The difference between the two 'states' represents the net (economic) effects of implementing the policy in question. As with I-O models, the outputs can provide an indication of economic impact that can be used as part of a CBA to determine whether the benefits of a proposed option outweigh the costs.

Various applied CGE models have been used to assess the implementation of environmental policies, with several of these designed specifically to assess the overall economic impact of addressing the greenhouse effect. Some of the leading models are reviewed in Cline (1992) and a review of economic models for abatement cost assessment can be found in Boero *et al* (1991).

In addition to applied CGE models, there are other related modelling approaches for assessing the direct and indirect effects of environmental policies, although these are not as sophisticated as the CGE models. A number of traditional econometric models used to assess the medium-term economic effects of environmental policies are also reviewed in OECD (1997).

3.3.2 Limitations of CGE Models

CGE models start at the 'top', giving an indication of what should happen in response to a policy or other economic change assuming that the economy in question conforms to the assumptions of the model. This has led to some commentators arguing that these models are too abstract for the real world, calling instead for use of the more traditional partial equilibrium approaches that take a set of observations relating to what is actually happening (EC, 1996). Additionally, these models may also suffer from the problem of 'losing' relatively small changes in a model of a large economy.

As with I-O models, the inherent complexity of CGE models means that the amount of time and effort required to collect the basic data and build a suitable model is often prohibitive. As a result, no model can actually include all possible markets. In practice, many markets are aggregated together and other simplifications are made to create a useable and practical model. Models are generally tailored to particular needs and functional forms are chosen with an eye to reducing the number of elasticity-related parameters that must be estimated. This usually means that most CGE models have an I-O model core that defines production relationships in terms of intermediate inputs, and the only flexibility in production is usually some substitution between capital and labour as inputs.

Most CGE models also make the simplifying assumption that there is no unemployment, *i.e.* the labour market is in equilibrium. Any change in employment levels is assumed to result from voluntary decisions on the part of the workforce. The outcome is that different models can reach different conclusions regarding the impact on employment of implementing environmental policies. As a result, OECD (1997) indicates that the findings of studies using CGE models should be considered with reservations.

3.4 Integration with Other Analyses

Clearly from the above, the use of macroeconomic modelling techniques provides a different level of information on the impacts of a risk reduction option than does the use of a partial equilibrium or efficiency-based approach (*i.e.* CBA and CEA). However, these models alone will not provide the decision maker with an indication of whether or not the benefits of a proposed policy would outweigh the costs. This requires that the outputs of an I-O or CGE model are combined with other data on environmental exposure, cause and effect relationships and valuations of environmental and health effects. In essence, the outputs need to be brought into an efficiency-based framework (such as CBA) in order to determine whether or not a proposed option would be justified in economic resource terms.

This is not to say that such approaches are not useful. On the contrary, it is often the case with partial equilibrium approaches that impacts on related markets are, for pragmatic reasons but also as a matter of course, assumed to be minimal. They are left out of any analysis of a policy's impacts, potentially resulting in the adoption of an option which would not be efficient if all effects were considered. In addition, tools that provide a better sense of the economy-wide, indirect effects may also serve to better examine the distributional or equity effects of an option.

However, given the usual complexity of modelling an economy (or even just specific sectors), the resource implications of such studies may restrict their use to only the risk reduction options that could be far reaching (for example, the phase-out of a widely-used substance). Hence, the application of macroeconomic modelling in SEAs of chemical risk reduction will be fairly limited.

3.5 Application to the ZOZ Case Study

3.5.1 Introduction

The manner in which an input-output model might be developed and applied to assessing alternative risk management options for ZOZ is examined below. This discussion basically focuses on the development and application of an I-O model as part of Stage 3 in the overall SEA process.

3.5.2 Specifying the Objectives

The objectives of the analysis, as defined with the application of a macroeconomic model, would vary from those for a CEA and CBA. For the latter, the objectives would relate to determining the economic efficiency of alternative risk reduction options. In applying an I-O model to ZOZ, the objective is more likely to be:

- developing information on the impacts of the proposed options on the economy as a whole and, possibly, on specific sectors within the economy; and

- creating links between the sectoral changes in outputs and emissions of ZOZ, so that predictions of the changes in emissions associated with the proposed changes in sectoral activities can be made.

3.5.3 Specification of Risk Management Options

A list of potential risk reduction options was identified in Section 2.1. It is unlikely that one would want to apply I-O modelling to some of the listed options, for example to assessing the impacts of product labelling or employee health and safety based options. However, there could be value in using such models to examine the implications of a product tax assuming that ZOZ is used in a product that is consumed at a cross-sectoral level. The same might hold for marketing and use restrictions where these may impact several sectors' costs.

The baseline against which the proposed options would be analysed would essentially be the starting assumptions for the I-O model. In other words, it would relate to the fixed set assumptions input into the model when it was developed. As a result, it is impossible for such models to take into account trends in production techniques, technological innovation, *etc.* However, the time period adopted in the analysis could be set in a manner consistent with any CEA or CBA in terms of the remaining commercial life for ZOZ, with this then taking into account developments in the sectors and applications of concern.

3.5.4 Data Collection, Analysis and Options Appraisal

As indicated above, it is unlikely that an I-O Model (or for that matter a CGE model) would be created specifically for the SEA of ZOZ risk management. Instead, it is more likely that an existing model would be modified to provide the necessary economic-environment linkages. The starting point for the construction of a typical input-output model is a data set, showing annual sales between all productive sectors in the economy, and to end consumers for a given base year. The sales are normally recorded in monetary units, although they can be measured in physical units.

A highly simplified accounting framework for a three-sector system is shown in Table 3.2. The three sectors considered are energy, industrial processing (the sector that would account for use of ZOZ) and commercial services. Looking at row 2, for example, the table shows that the industrial processing sector produced £52,756 of total output, of which £3,208 was purchased by the energy sector, £7,897 by the commercial services sector, and £41,651 by final consumers (final demand). To produce this output (see column 3), the industrial processing sector required £5,608 and £21,137 worth of inputs from the energy and commercial services sectors respectively, and primary inputs costing £26,011.[21] The total cost of all inputs to a sector equals the total value of output from that sector.

Table 3.2: Hypothetical Three Sector 'Transactions' Accounting Framework (£)

Production Sector	**Energy**	**Processing**	**Services**	**Final Demand**	**Totals**
Energy	0	5,608	1,658	15,464	22,730
Processing	3,208	0	7,897	41,651	52,756
Services	1,543	21,137	0	21,288	43,968
Primary Inputs	17,979	26,011	34,413	78,403	
Totals	22,730	52,756	43,968		119,454

Assuming that each sector uses inputs strictly in fixed proportions,[22] production technologies remain constant, no economies of scale apply, marginal and average input ratios are the same, and no input substitution occurs, then an input-output model can be constructed from the data in Table 3.2 (Hufschmidt *et al*, 1990). By dividing each entry in Table 3.2 by the corresponding column total, the direct requirements per unit output for each sector can be determined. This yields a set of so-called input or direct requirement coefficients (a_{ij}). In general:

$$a_{ij} = \frac{w_{ij}}{q_j}, i = 1,...n; \; j = 1,...n$$

[21] Primary inputs are factor incomes generated in the production process, *i.e.* income from employment, self-employment and gross profits (CSO, 1995).

[22] That is, a 1% increase in energy output will result in a 1% increase in all inputs.

where w_{ij} is the total output of the ith sector purchased by the jth sector in the base year, q_j is the total output of the jth sector in the base year, a_{ij} is the amount of output from the ith sector used in the production of a unit of output from sector j and n is the number of sectors in the economy. Table 3.3 displays the direct requirement coefficients derived from the transactions data contained in Table 3.2. For example, \$0.1063 and \$0.4007 of energy and commercial services, respectively, are required directly to produce \$1 of processed goods.

Table 3.3: Direct Requirements Coefficient Matrix ($)

Production Sector	**Energy**	**Processing**	**Services**
Energy	-	0.1063	0.0377
Processing	0.1411	-	0.1796
Services	0.0679	0.4007	-

With knowledge of the direct requirement coefficients, it is possible to determine the total output (direct and indirect) of each sector for any assumed level of final demand. For example, if final demand for energy, processed products and commercial services in the year 2000 is assumed to be \$25,000, \$60,000 and \$45,000 respectively, the total output of each sector is determined by solving the following set of simultaneous equations:

$$q_1 = 0.0q_1 + 0.1063q_2 + 0.0377q_3 + 25{,}000$$
$$q_2 = 0.1411q_1 + 0.0q_2 + 0.1796q_3 + 60{,}000$$
$$q_3 = 0.0679q_1 + 0.4007q_2 + 0.0q_3 + 45{,}000$$

where the total output of each sector is denoted by q_1, q_2, and q_3.

The first structural equation states that the energy sector has to produce a sufficient amount of energy to meet the input demands of the processing and commercial services sector, and the final demands of consumers. The answer is q_1 = \$36,428, q_2 = \$79,380 and q_3 = \$79,277.

As inter-sector linkages are modelled, the total output of each sector greatly exceeds the final demands, *i.e.* \$25,000, \$60,000 and \$45,000. This illustrates the capacity of input-output models to account for indirect effects in addition to direct effects. As backward and forward production linkages are taken into account, the value of total (direct) output is less than the value of total (direct and indirect) output, as determined by the model. In order to examine the implications of options to reduce the risks associated with ZOZ, the above model could be used to compare two distinct states of the economy: before and after implementation of the option. The difference between the two situations would give an indication of the 'net benefits' of the proposed option (normally expressed in terms of GDP). As illustrated, both direct and indirect effects would be considered.

The above set of simultaneous equations can be solved by simple substitution and elimination methods. For a model that contains a realistic number of sectors, however, it is necessary to make use of Leontief inverse matrix coefficients (for an explanation of these, see, for example Bailey & Parikh [1990]). These coefficients derived for the above three-sector model are shown in Table 3.4 and can be interpreted as the amount of gross output from sector i required, both directly and indirectly, to produce one unit of output from sector j for final output.

Table 3.4: Leontief Inverse Matrix: Total Requirements Coefficients (direct and indirect)

Production Sector	**Energy**	**Processing**	**Services**
Energy	1.0231	0.1339	0.0626
Processing	0.169	1.0997	0.2039
Services	0.1372	0.4497	1.0859

Using these coefficients, sectoral outputs are given by:

$$q_1 = (1.0231)(25{,}000) + (0.1339)(60{,}000) + (0.0626)(45{,}000) = £36{,}428$$
$$q_2 = (0.1690)(25{,}000) + (1.0997)(60{,}000) + (0.2039)(45{,}000) = £79{,}380$$
$$q_3 = (0.1372)(25{,}000) + (0.4497)(60{,}000) + (1.0859)(45{,}000) = £79{,}277$$

Now, assuming that detailed base year information for each sector's interactions with the environment is available,[23] this can also be represented in coefficient form and used to assess environmental effects. Table 3.5 provides a hypothetical data set for the three-sector model described above.

Table 3.5: Base Year Environmental Quality Parameters

Production Sector	Energy	Processing	Services
Minerals/fuels (t/year)	25,000	60,000	120,000
Water (m^3/year)	50,000	80,000	20,000
ZOZ (t/year)	100,000	10,000	5,000

Direct coefficients are derived for each entry in Table 3.5 by the corresponding base year total output level for each sector. Coefficients for minerals/fuel, water and ZOZ emissions are shown in Table 3.6.

Table 3.6: Matrix of Direct Environmental Quality Coefficients

Production Sector	Energy	Processing	Services
Minerals/fuels (t/year)	1.0999	1.1373	2.7293
Water (m^3/year)	2.1997	1.5164	0.4549
ZOZ (t/year)	4.3995	0.1896	0.1137

Each environmental quality coefficient (denoted by e_{kj}) indicates the average level of environmental quality variable k per unit output of sector j. In general, the levels of environmental quality variables r_1, r_2,....., r_s accompanying any given set of sectoral output is computed by solving the following set of equations:

$$r_1 = e_{11}q_1 + e_{12}q_2 + e_{13}q_3 + ... + e_{1n}q_n$$
$$r_2 = e_{21}q_1 + e_{22}q_2 + e_{23}q_3 + ... + e_{2n}q_n$$
$$.$$
$$.$$
$$.$$
$$r_s = e_{s1}q_2 + e_{s2}q_2 + e_{s3}q_3 +... e_{sn}q_n.$$

Therefore, assuming final demand for energy, processed products and commercial services is the same as that given above, 346,712 tonnes of minerals/fuel and 236,565 m^3 of water would be consumed, and 184,324 tonnes of ZOZ would be emitted. Further analysis could be conducted to

23 For example, data on the demand for raw material and discharges of residuals into the environment.

translate these residuals into ambient environmental quality, and ultimately into estimates of environmental damage in monetary terms.

By applying the above approach to each proposed risk reduction option, estimates could be developed both of the change in GDP that would result across the economy (and of output for individual sectors) and of the change in environmental quality. These data could either be presented to decision makers in these forms, or as noted earlier incorporated into a CEA or CBA based analysis (see Section 2 for a discussion on what this would involve).

4. TOOLS FOR ASSESSING MULTIPLE CRITERIA

4.1 Introduction

The final category of tools represents those techniques that fall under the heading of multi-criteria analysis. There are two key differences between analyses using MCA and those using either efficiency-based or macroeconomic analysis:

- in contrast to the economics-based forms of analysis, MCA has been developed to enable analysts to explicitly examine the performance of different options against multiple objectives in the comparative assessment of options; and

- the aim of MCA is to provide a tool to aid decision makers[24] by directly incorporating their value judgements in the assessment of options; this contrasts to economic analysis (particularly the efficiency based approaches of CBA and CEA) which is aimed at providing an objective measure of the net value (or social worth) of a proposed option. In essence, MCA is a decision tool and economic tools are more often used to provide information that is input into the decision making process (and thus can be included in MCA).

MCA can be applied to decisions which concern either a few sectors (or firms), or can be applied to decisions which are likely to have impacts at the wider macroeconomic level. The key features of such analyses are the specification of the range of decision objectives, the identification of criteria to provide a means of measuring the degree to which the various objectives are met, and the relative weighting of the objectives to reflect the preferences of decision makers (who may be a single entity or representative of a group of stakeholders).

The theoretical foundation for multi-criteria analysis was established in the late 1940s (von Neumann & Morgenstern, 1947) and was later expanded to include multiple preferences and objectives (Keeney & Raiffa, 1976). It has since then been the subject of much academic and practical research. As a result, a range of different techniques for structuring and assessing complex decision problems has been developed. These include simple approaches for screening and ranking options followed by more sophisticated approaches for the comparative assessment of options.

The remainder of this section reviews:

- the key principles underlying MCA;

- an overview of steps involved in applying MCA and the techniques available; and

- an illustration of how both the simpler and more complex MCA methods could be applied to the ZOZ case study.

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[24] Note that the term decision maker may refer to a 'supra' decision maker who is responsible for public policy making and takes decisions on behalf of society, or it could refer to a group (e.g. stakeholders) that has been brought together to act as a decision maker on a particular issue.

4.2 Key Principles

4.2.1 Multi-Attribute Utility Theory

As indicated above, the foundations for multi-attribute utility theory (which provides the framework for multi-criteria analysis) lay in the work carried out by von Neumann and Morgenstern (1947) on decision making under uncertainty (as part of wider work on game theory). They developed a normative model that prescribes how a rational individual 'ought' to make decisions when faced with uncertain outcomes. To do this, they defined a series of axioms that set out what they meant by rational behaviour in the face of uncertainty. Through these axioms, they demonstrated that, by combining information on preferences and probabilities, a utility function could be assigned to different outcomes so that a decision maker should prefer the alternative with the highest expected utility[25] (the expected utility hypothesis) (Keeney, 1982). The implication of this finding is that probabilities and utilities can be combined to calculate expected utilities. Comparison of expected utilities allows determination of the relative desirability of different alternatives; the higher the expected utility, the more preferred an alternative should be.

The utility theory developed by von Neumann and Morgenstern was extended by Keeney and Raiffa (1976) to relate to multi-attribute outcomes. Their extension involved adding assumptions concerning the independence of utility measures across attributes (Keeney & Raiffa, 1993; Drummond *et al*, 1997). The most commonly relied on assumption is that of additive utility independence, which implies that there is no interaction among the attributes and the overall preference depends only on the quantity/quality of individual attributes and not on the manner in which these are combined.[26]

4.2.2 Measuring Preferences in Multi-Criteria Analysis

Practitioners generally distinguish between the concepts of 'utility' and 'value', with each referring to a different measure of preferences.[27] Whether preferences are being measured in terms of utilities or values depends on the way in which they are elicited: are the outcomes considered certain or uncertain? Is the respondent asked to make a choice between outcomes or perform some sort of scaling task?

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25 It should be noted that cardinal utilities under uncertainty, as defined by von Neumann and Morgenstern, are different from the ordinal concepts of utility underlying contemporary microeconomics (or the cardinal utilities underlying nineteenth-century economics). See also Drummond *et al*, 1997 or Allais, 1991.

26 The other forms of utility independence are first order and mutual utility independence, with these relating to the use of multi-linear and multiplicative utility functions; these functional forms are rarely used in practical applications to environmental issues although they may be used in health applications.

27 Von Neumann-Morgenstern utility theory indicates that utilities are appropriate for problems that involve certainty or uncertainty or both. However, values are only appropriate for problems that involve certainty.

Table 4.1 sets out a typology of methods for eliciting preferences that are either value-based or utility-based.

Table 4.1: Methods for Measuring Preferences		
Judgement Required	**Framing of Outcomes**	
	Certain/Riskless (Values)	**Uncertain/Gamble (Utilities)**
Numerical estimation techniques	Direct rating Category estimation Visual scales/curve drawing Ratio estimation Bisection scaling	None applicable
Choice or indifference-based methods	Differences standard sequence Equivalence-based methods Person trade-offs Paired comparisons	Gamble methods (variable probability method, variable certainty method)
Source: Based on von Winterfeldt & Edwards (1986) and Drummond *et al* (1997)		

As can be seen from the table, most of the methods result in the elicitation of values. Only gamble-based methods, which incorporate an individual's attitude to risk and uncertainty, enable the elicitation of utilities. Within the methods that can be used to elicit values, two further distinctions can then be made. Numerical estimation techniques are those which ask an individual to indicate his/her strength of preference in relation to a numerical scale (*e.g.* 1 to 5). This contrasts with the use of choice methods, which ask individuals to choose between two alternatives (thereby revealing their preferences). The various methods are discussed more in Section 4.3.3 below.

Theory would suggest that wherever a decision is characterised by uncertainty, as is likely to be the case with regard to chemical risk management, then standard gamble methods should apply. However, practical issues in terms of time and resources are likely to result in techniques being adopted that are based on the other choice methods or scaling methods (potentially with uncertainty taken into account through other means).

4.2.3 From Individual to Group Preferences

It is important to note that the theory presented above was developed to cover decision making by an individual. In general, it does not hold when an analysis involves the aggregation of preferences across individuals as part of societal decision making.

Proponents of multi-criteria analysis argue however that, particularly in public sector decision making, there is a decision maker (or a group number of individuals acting as the decision maker) who is responsible for making the end decision. It is the responsibility of the decision maker to incorporate the views of the individuals who will be affected into his/her decision making framework. In essence, the decision maker's utility function is dependent upon (or comprised of) the utility functions of the individuals or groups that are being taken into account in the decision making process. The decision maker, therefore, must consider how the decision will impact on the different individuals and the trade-offs involved.[28]

[28] In setting out a series of assumptions concerning aggregation of individuals' rankings to develop a group ranking, Arrow (1951, as quoted in Keeney & Raiffa, 1993) found that there was no procedure for combining individual rankings into group rankings that did not involve interpersonal comparisons of the preferences of the individuals. For further detailed discussion on Arrow's Impossibility Theorem and other assumptions relevant to the aggregation of individuals' preferences see Keeney & Raiffa, 1993.

This does not prevent of course the use of participatory processes for establishing the values or utilities used in any analysis. Instead, the process requires that either a consensus be developed among those partaking, or the use of sensitivity analysis to examine the implications of adopting alternative value systems or utility functions.

4.3 Applying Multi-Criteria Analysis in Practice

4.3.1 The Key Steps

Applying the theory set out above to the analysis of decision problems essentially involves the use of the same type of framework as set out in Part 1 of this Guidance. Because MCA is often argued as being more of a process-based approach than CBA and CEA (for example, see Watson, 1981), it is often introduced in terms of a series of steps. These are summarised in Box 4.2 below. As can be seen from the Box, a number of the steps which are formally considered to form part of MCA also form part of the framework suggested here for chemical risk management.

Box 4.2: The Steps in Applying MCA

The steps generally followed in applying MCA are listed below, with these preceded by problem identification activities and followed by reporting:

1. Screening of alternatives
2. Definition of alternatives
3. Selection and definition of criteria
4. Assessment of scores for each alternative
5. Standardisation of scores
6. Assessment of weights
7. Ranking of alternatives
8. Sensitivity analysis

The techniques falling under the general heading of MCA have been developed to assist with several of the steps listed above. However, one can also distinguish between the techniques that can assist at a simple level in screening and choosing among competing options and those that provide a more systematic and rigorous assessment, incorporating decision makers' preferences.

4.3.2 Simple Screening and Choice Methods

The Methods

A number of the methods that have been developed as components of the more sophisticated multi-criteria techniques are also commonly applied on their own to assist in option selection. These include the following:

- **screening methods**, which involve the use of pre-defined criteria (*e.g.* achieving target level of risk reduction, costs to industry and consumers, administrative implications, *etc.*) to eliminate options from further consideration; these methods can also be used to highlight the advantages and drawbacks of particular options;
- **ranking methods**, which involve the use of verbal, alphabetical or numerical scales to indicate the order of preference for different options (*e.g.* the option which is ranked highest achieves the greatest level of risk reduction);

- **pairwise comparison techniques**, which involve indicating a preference between two options for each of a series of criteria (or impacts); where there are more than two options, the procedure is repeated until comparisons have been made across all potential pairings. The information is then used to highlight trade-offs involved in selecting one option over another; and
- **scaling methods**, which provide a means of indicating how well an option performs (for example using + and - signs or the number of ticks) across a range of criteria; such methods when used on their own can provide an overview of the key impacts.

Key Issues for Simple Screening and Choice Methods

The above types of methods, because of their simplicity, are readily applied and easily understood. As a result, they have been applied to aid decision making on chemical risk management issues. An example of this is the use of trend analysis to highlight the advantages and drawbacks to different sectors associated with specific risk reduction measures (see EC, 1997). Essentially, such analyses have been used as a means of supplementing quantitative data (generally cost data) with a summarisation of more qualitative assessments of impacts.

Although these methods can be extremely valuable in the chemical risk context, it should be recognised that they do not meet the theoretical requirements underlying this category of tools. This is because these methods fail to explicitly incorporate decision makers' preferences into the comparative assessment in a manner that enables the aggregation across criteria (or impacts).

For example, with the use of screening methods, in the absence of any formal preference structure, an option may be screened out on the basis of one criterion, the importance of which may be minimal in comparison to other decision criteria. Similarly, where screening relies on the use of acceptable thresholds that are represented by specific values (*e.g.* a predicted no effects level), an alternative may be excluded which fails by a very small margin to meet the value.

Similarly, when used on their own, rankings and scalings provide little information on the magnitude of any differences in impact between options, or on the relative preference for one type of impact versus another. They therefore hide the extent to which such differences may be important, for example, whether the costs to industry and consumers are valued less than or more than human health and environmental benefits. In addition, when there are several options under consideration, there is a tendency for people to add ranks (and scales) together; this is mathematically invalid given that they fail to take into account the relative (proportional) differences in magnitudes and preferences for different impacts.

4.3.3 More Comprehensive Choice Methods

The second set of multi-criteria techniques comprises techniques that fulfil theoretical requirements by incorporating both the assessment of impacts and the establishment of preferences across impacts. The key steps involved in the application of these techniques are:

- the selection and definition of criteria;
- the assessment of scores;
- the transformation of scores to value or utility measurements;
- the assessment of preference weights; and
- the aggregation of weighted scores.

The precise manner in which the above steps are carried out for any particular analysis will depend upon the actual form of MCA being applied. The main types of techniques that are currently in use in the assessment of human health effects are discussed in more detail in Part 3, Section 3.4. Similarly, the techniques which have been applied to environmental issues (albeit rarely in a chemical risk context) are discussed in Section 4.3 of Part 3. It is of note that there are well developed methods in the health field, while these are still lacking with regard to the environment and chemical risks.

Key Issues in Multi-Criteria Analysis

With regard to the application of MCA, a number of key issues need to be borne in mind in order to ensure a systematic and reliable analysis. Starting with the first of the above steps, the choice of decision criteria is a very important step in the MCA process. The criteria are specifications of the impacts that will be taken into account in the decision. Ideally, the decision criteria should represent the important aspects of the decision. They must be relevant to the decision and the decision maker must be able to relate to them. They must be unambiguous, and it must be possible to measure their performance with a reasonable degree of accuracy. Furthermore, the criteria should be additive, thus contributing independently to the total performance of a decision alternative.

The criteria should also be linked to decision making objectives. For example, applying MCA to risk reduction might result in criteria such as ‘reduction of the number of people affected by a particular health outcome’, ‘cost minimisation’ and ‘minimisation of administrative requirements’. The links between criteria and objectives may not always be as obvious as this, but are ultimately the only justification for the formulation of criteria.

Most impact assessors favour quantifiable criteria. However, leaving aside the costs of measuring these criteria, quantitative methods of comparison are by no means always the best for describing impacts. Moreover, because relevance and quantifiability do not necessarily go hand in hand, qualitative criteria have to be used in many impact summaries (for example, expressed as pluses and minuses - the significance of which must, of course, be clearly indicated).

The number of criteria/impacts included in such analyses is often large. This generally arises from the desire to ensure that all of the criteria that stakeholders may be concerned with are incorporated in the analysis. The result of this may be a failure to construct a systematic evaluation framework, and it may lead to double-counting, confusion between means and ends in the criteria, dependencies among criteria, missing criteria and inconsistencies in spatial and temporal scales.

With regard to the measurement scales used in the assessment, if these vary across criteria/impacts (*e.g.* number of illnesses avoided, number of sites no longer experiencing unacceptable aquatic risks, number of companies affected by increased costs, *etc.*), then the scales must be standardised to a common dimension or dimensionless unit before the criteria can be weighted and aggregated. There are several ways to standardise the criteria. The simplest procedure involves scaling performance according to the relative distance between zero and the maximum performance. Alternatively, the performance can be scaled according to the relative position on the interval between the lowest and highest performance. There are also other methods of standardisation, such as division by the sum of the performances of each criterion, or division by ideal or target values.

Weights should reflect the subjective preferences of the decision-maker(s), but their development and use is a complex and controversial task. In a weighting process, one has to present clear trade-offs to the decision maker, asking questions such as how much of, say, criterion 2 they are willing to sacrifice to obtain an increase in, say, criterion 4. In particular, the attribution of weights is much criticised for the fact that it incorporates subjectivity into the assessment. However, it should be noted that assigning preferential weights before demonstrating trade-offs to decision makers is a significant flaw in the methodology.

Practitioners argue, however, that such subjectivity is inherent to all appraisal methods that involve making choices between competing options, and because the inclusion of subjective judgements is always undertaken explicitly.

It is important though for all stakeholders to know who did the weighting and understand the perspective this represents. For example, weights are sometimes assigned by experts on the relative importance of criteria/impacts under one general heading; an illustration of this is for experts to weight the relative importance of risks to different environmental receptors on the basis of reasoned argument backed-up by scientific knowledge.

However, deciding the relative importance of the different themes (ecology, health, costs, distributional effects, for example) is generally considered to be the remit of those who will actually take the decision. In principle, though, different sets of weights should always be used at this level; in other words, account should be taken of different perspectives. Working with different perspectives is an important element of MCA, as it can be used to demonstrate how different preference functions affect the ranking and hence choice of options.

4.4 Application to the ZOZ Case Study

4.4.1 Introduction

All multi-criteria methods transform the input, performance scores and weights to a ranking using a decision rule specific to that method. Of the various techniques available, the 'weighted summation' method provides a good candidate for use in chemical risk management. It is theoretically well established, can be easily explained, and is easy to use. There is, therefore, less chance that the method will be viewed as a 'black box'.

The steps involved in applying the weighted summation method are (see also Section 4.3, Part 3 for further discussion):

1. Select and define the decision criteria and associated impact measures to provide the basis for a comparative assessment of the options.

2. Standardise the scores for each criterion/impact.

3. Assign the relative preference weights.

4. Multiply the weights by the standardised scores.

5. Add up the resulting scores to obtain total weighted scores for each option.

6. Determine the ranking of each option given the total weighted scores.

The most difficult tasks in applying the weighted summation method are those of: choosing a good approach for scoring the impacts and then standardising these scores (in other words, converting the scores to a common numeraire); and in assigning the relative weight that should be given to one criterion/ impact versus another.

It can be a less suitable method for processing qualitative data than some of the other techniques available. In the context of chemical risk management, this disadvantage may not be important, as the qualitative data will often be derived from underlying quantitative data. By selecting an appropriate method of standardisation, the underlying quantitative data can be used in the scoring process.

As for the sections concerning the application of economic efficiency based approaches and input-output models, this discussion focuses on the application of a weighted summation method, starting with Stage 3 in the overall SEA process.

4.4.2 Refining the Objectives

In adopting MCA rather than CEA or CBA as the methodological framework for the assessment, the objectives of the analysis are likely to shift from being the provision of objective (to the degree possible) estimates of costs and benefits (as for CEA and CBA) to assisting the decision maker in bringing such information together with judgements on the importance of these when deciding upon the most appropriate form of risk reduction.

For example, the objectives for an MCA might be as follows:

– the aim should be to find the option that best meets the conflicting objectives of society in terms of achieving risk reductions while at the same time minimising costs to industry, consumers and regulators;

– when defining the criteria that will provide the basis for option selection, ensure that these reflect the key concerns of the main stakeholder groups, taking care to ensure that the analysis remains focused;

– both qualitative and quantitative measures of impact should be incorporated into the MCA;

– the weights used to reflect the relative importance placed by society on the conflicting criteria should be drawn from both stakeholders (to provide an indication of the degree to which they might vary across different interests) and the end decision maker; and

– the trade-offs involved in selecting one option over another should be made clear.

4.4.3 Specification of Risk Management Options

A short-list of potential risk reduction options was identified in Section 1.3, where these included voluntary approaches, worker training, use restrictions and best available technology requirements. All of these could be examined in the MCA. Care would need to be taken, however, to ensure that the criteria covered and impacts used to assess these were comprehensive enough to pick up differences in the key effects arising under the different options.

It will be recalled that as part of this task, analysts would also have to establish the baseline to be used in the analysis. This baseline should be the same as for any of the other forms of appraisal (*e.g.* CEA and CBA) in that it relates to the current and expected situation in the absence of any further regulatory action.

Perhaps a more difficult task is that of determining how variations in impacts over time across the different criteria are to be accounted for within the analysis. There is a range of different approaches that could be adopted in this regard, including consideration of the total net effect by a given year or incorporating a time dimension into the scoring system (for example, by discounting scores in a similar way as to the discounting of costs and benefits in economic analysis). A common approach is to separate short- and long-term effects into two separate criteria and ask the decision maker to explicitly trade-off the short against the long-term. In essence, this means that instead of adopting a discount rate based on society's time preferences, a more subjective time preference is adopted.

4.4.4 Data Collection

Selection and Definition of Criteria

Before data collection activities can proceed, analysts will need to define the criteria against which the performance of the different risk reduction options will be assessed. Given the objectives specified above, it is likely that some form of consultation exercise or group process should be undertaken to elicit the views of different stakeholders (including the decision maker) on the criteria that should provide the basis for the analysis.

It will be important that the criteria are defined in a systematic manner in order to ensure that they cover not only the key impacts of concern, but also are able to highlight differences in impact between the various risk reduction measures. In general, the criteria and impact measures selected for these purposes should be good indicators of the change that will occur in health or environmental risks, compliance costs, or other impacts in moving from the pre- to the post-risk reduction situations. In the choice of criteria, it is vital to ensure that they match the likely impacts of the range of risk management options being considered. In other words, the selection of criteria should result in a reliable and consistent measurement of impacts across the range of options. Some options may weigh heavily on specific criteria (such as the impact on consumers), but the analyst should ensure that major impacts are covered for all the options in the development of criteria.

For the purposes of this illustrative case study, it is assumed that five different criteria have been defined, with these being:

- worker health and safety: measured in terms of number of illnesses per annum;
- risks to the aquatic environment: measured in terms of predicted environmental concentrations;
- change in costs: measured in terms of total annualised costs to industry and regulators;
- impacts on consumers: measured in terms of changes in quality and availability of key products; and
- employment effects: measured in terms of changes in the number of those directly employed in the affected industry sectors.

In reality, it is likely that for a chemical such as ZOZ, public health considerations would also be considered a key decision making criterion. For example, there may be concern that exposure to ZOZ within products leads to health risks for consumers. In addition, given that ZOZ is distributed widely in the aquatic environment, there is obviously the potential for the general public to be exposed indirectly *via* the environment to concentrations of ZOZ which lead to unacceptable health risks. For example, such exposure may result from the consumption of contaminated fish or water. For simplification, these further effects are not examined in detail in the ZOZ case study (but this does not mean they may not be critical to identifying the preferred risk management option).

Collection of Data and Scoring Options

In order to assign scores to the above criteria, data need to be collected for each of the above criteria and associated measures of impact. A number of sources will need to be called upon for these purposes, including risk assessors (pre- and post-option risk levels), industry and regulators, and consumer groups at a minimum. Note that the techniques used in collecting these data are likely to be similar to those used when undertaking a CEA or CBA (*i.e.* direct consultation, questionnaires, literature reviews, market surveys - but excluding economic valuation surveys).

Note data collection is likely to involve some of the same scenario and related modelling activities as undertaken for economic analyses. In other words, scenarios will have to be generated which explain how each of the risk reduction measures would be implemented for each of the target sectors (and potentially characteristics of the companies within these). These would then provide the basis for determining the impacts that would arise under each of the five criteria.

Table 4.2 summarises the data collected for the purposes of scoring a selection of the ZOZ risk management options. The scoring of impacts can be undertaken in a number of different ways, ranging from the use of qualitative to quantitative scales of value (see also Table 4.1). Any model or technique for measuring values (or utilities) is simply a set of prerequisites and rules for assigning numbers to the impacts of concern. In practice, the techniques used may be complex systems or simple sets of instructions for constructing rating scales. In all cases, human judgement is a major ingredient.

Table 4.2: Impact Scores for MCA-Based Assessment of ZOZ Risk Management

Criterion or Impact	**Scoring Basis**	**Voluntary Approach**	**Worker Training**	**Marketing and Use Restrictions**	**Command and Control**
Worker health & safety	Number of illnesses	14	17	1	2
Aquatic environment	Predicted concentrations (Φg/litre)	4	10	0.1	1
Costs	Annualised costs over 15 years (mill $)	250	90	1000	700
Impacts on consumers	---/0	-	0	---	--
Employment	Number jobs lost (100s)	0.5	0	2	1

The main issues in value (or utility) measurement are concerned with the method used for constructing the scales and whether or not respondents are asked to consider a certain or uncertain outcome (note that these issues are those typically stressed when considering the psychology of scaling). The overall approach taken will have an impact on the reliability of the analysis and the degree to which any errors or biases enter into the measurement process.

When it makes sense to do so, a scale is selected or constructed that represents some natural quantitative attribute of the impacts to be assessed (*e.g.* for the various cost variables and predictions of changes in health and environmental risks such as the population subject to a given risk level or number of illnesses avoided). However, qualitative scales may need to be constructed for other impacts. As can be seen from Table 4.2, it has been possible to rely on natural quantitative scales for most of the criteria, with a qualitative scale only adopted for scoring impacts on consumers in terms of product quality and availability.

4.4.5 Data Analysis and Options Appraisal

The three key activities that would take place during this stage of the SEA are standardisation of the impact scores across the criteria, assignment of relative preference weights and aggregation of weighted scores to develop option rankings. Each of these activities is discussed in more detail below.

Standardisation of Scores

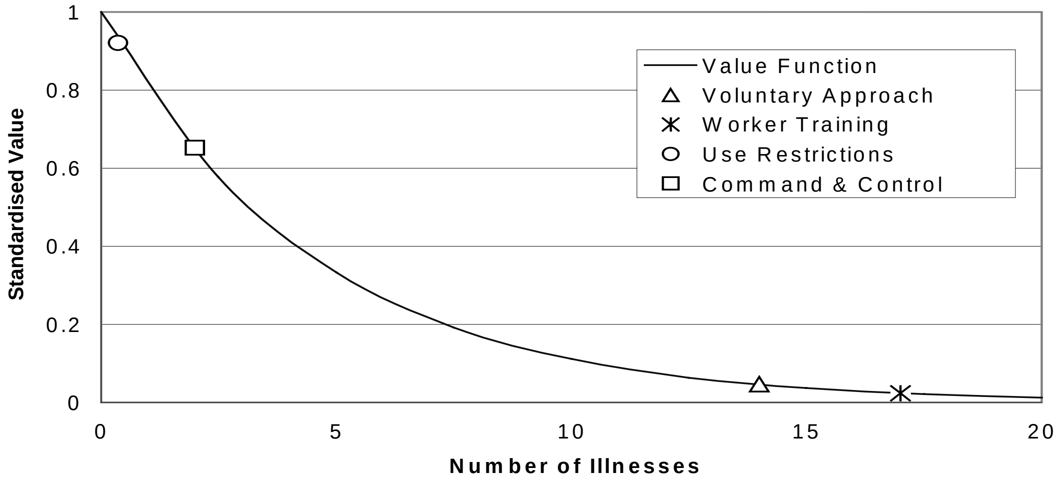
As the data are collected, they will be brought together to allow the comparative assessment of the various options. Within an MCA-based application the next step is then to standardise scores when they are measured using different scales, as is the case in Table 4.2 for ZOZ risk management.[29] This involves converting the 'natural' measurement scale to a value or utility scale by means of judgements of relative value or preference strength.

Table 4.1 set out the available assessment techniques, categorised according to whether they ask respondents questions that have a certain outcome (a riskless outcome) or an uncertain outcome (a gamble) and the type of response sought of respondents (see also Keeney & Raiffa, 1976; Keeney, 1992; and Beinat, 1997). For this case study, it has been assumed that the outcomes under each option are certain, resulting in the adoption of a value function-based approach. For each criterion, value functions are used to establish a relationship between the physical scores, for example, the remaining risk to the aquatic environment and the importance of that score on a scale between 0 and 1.

Figure 4.1 shows a hypothetical value function for worker safety (developed, for example, by a panel of experts). It is based on the assumption that changes in risks to workers are not valued highly until a certain level of reduction in the number of illnesses is achieved, at which point the value of the reductions rises quickly. The argument is that workers, regulators and others will not perceive the benefits of small changes and require some target reduction to be achieved before they will value the changes. This means that the scores should be standardised assuming a non-linear relationship. The relevant range specified for the possible number of illnesses is between 0 and 20. Reading from the curve presented in Figure 4.1 indicates that worker training as an option is very close to achieving the minimum value, while use restrictions achieve the score of almost 1.0.

[29] It should be noted that standardisation of scores is not necessary for all MCA methods.

Figure 4.1: Value Function for Worker Health and Safety



The same type of approach would be applied to the standardisation of risks to the aquatic environment, costs and employment given that they have all been scored using a ‘natural’ quantitative scale of effect. The standardisation of impacts on consumers, however, would be undertaken on a different basis. In this case, the scores assigned to impacts relate to four possible levels of effect. These are defined in Table 4.3.

Table 4.3: Scoring System for Impacts on Consumers

Score	Impacts
0	No change in product quality or availability.
-	Minimal impacts on product quality or availability in terms of the number of products affected and the period over which they are affected.
--	The impacts on quality and/or availability are either substantial but for a limited period, or fairly restricted but for the entire 15-year period.
---	Implies that there are significant changes in quality or in the availability of highly demanded goods.

Figure 4.2: Value Function for Impacts on Consumers

The level '-' is considered of little relevance, only a small difference from the status quo. Its normalised value is close to 1. Any further impacts on product quality or availability to the level '- -' is considered more important and is valued more highly. However, the greatest impacts on consumers take place if the disruption in product quality and availability becomes more widespread and of a longer duration, resulting in a score of '- - -'. The standardisation curve in this case is therefore convex. This is illustrated in Figure 4.2.

The standardised scores which result for each of the options following the above process are summarised in Table 4.4, where zero is the lowest possible score (represents the poorest performance) and 1 is the highest possible score (representing the best performance).

Table 4.4: Standardised Scores for ZOZ Risk Management Options

Criterion	Voluntary Approach	Worker Training	Marketing and Use Restrictions	Command and Control
Worker health and safety	0.05	0	1	0.65
Aquatic environment	0.15	0	1	0.8
Costs	0.35	1	0	0.05
Impacts on consumers	0.9	1	0	0.7
Employment	0.4	1	0	0.1

Assessment of Weights

The relative importance of the decision criteria is expressed by weights. Weights can be perceived as trade-offs: how much of attribute X is a decision maker willing to sacrifice for the benefit of attribute Y? In other words, they reflect the preferences of decision makers for one type of impact relative to another.

A number of different approaches can be used to elicit weights, including:

– methods based on trade-offs, such as swing weighting, which assign a value reflecting relative preferences to each attribute; however, this may prove difficult in the context of chemical risk management given the nature of the trade-offs that have to be considered (*e.g.* costs versus long-term health effects or long-term environmental damage) (see also von Winterfeldt & Edwards, 1986); or

– methods relying on qualitative information on preferences, such as pairwise comparison (Saaty, 1980[30]) and ranking methods (Janssen, 1992).

The use of direct methods for assessing trade-off weights requires respondents to answer a relatively large number of questions on their preferences that are often difficult to comprehend or to answer. In addition, the resulting preferences are only valid for the original scoring system. If the consideration of new options would result in changes to the scoring system, then all of the weights would need to be re-estimated. As a result, there are advantages to the use of methods, which rely on qualitative information. For example, the use of pairwise comparisons involves questions that are relatively easily answered and it provides sufficient information to control for errors. The drawback of such methods however is that they can require a large amount of information from the decision maker. There are also advantages and drawbacks to the use of ranking methods in this context (see Part 3, Section 4 for further details).

In this example, stakeholders have reached a consensus on the weights to be used in the appraisal. Together with the help of experts, they have considered the possible ranges in effect across all criteria, and evaluated the benefit that can be obtained by reducing all criteria scores from the highest to the lowest level. In their opinion, risks to the aquatic environment should be given the highest weight. They concluded that reducing predicted environmental concentrations in ZOZ from 10 Φg/litre to 0 Φg/litre over the 15-year period is more important than:

– reducing the number of illnesses experienced by workers from a maximum of 20 to 0;

– minimising impacts on consumers from very significant to no effects; and

– minimising the number of jobs lost from a maximum of 200 to none.

However, keeping costs low by reducing them from a maximum of $1,000 million to 90 million over the 15-year period was considered almost as important to reducing risks to the aquatic environment. Table 4.5 sets out the weights that were assigned to each of the criteria on the above basis.

Table 4.5: Criteria Weights Assigned in MCA of ZOZ Risk Management

Criterion	**Importance Ranking**	**Weight**
Worker health and safety	2	0.2
Risks to aquatic environment	1	0.4
Costs to industry and regulators	1	0.3
Impacts on consumers	3	0.05
Employment	3	0.05

30 A more up-to-date reference with a focus on decision makers is Saaty (1996). A full list of references in relation to the analytical hierarchy process (developed by Saaty) can be found at www.expertchoice.com/hierarchon/references/preamble.htm.

Aggregation and Option Ranking

The next step within this stage would involve standardising the criteria scores by applying the value functions to them and then combining these standardised scores with the weights listed in Table 4.5. The steps involved would be as follows:

- multiply the weights by the standardised scores;
- add up the resulting scores to obtain total weighted scores for each option; and
- rank the options according to the total weighted scores.

The results of applying these steps are shown in Table 4.6. As can be seen from the Table, the option that appears the best over all five criteria is the adoption of best available technologies across the various industry sectors. The next best option is the adoption of use restrictions, which would need to perform better in terms of costs, impacts on consumers and employment if it were to equal the adoption of best available technology.

Table 4.6: Total Weighted Scores for ZOZ Risk Management Options

Criterion	**Voluntary Approach**	**Worker Training**	**Marketing and Use Restrictions**	**Command and Control**
Worker health and safety	0.01	0	0.2	0.13
Aquatic environment	0.06	0	0.4	0.32
Costs	0.11	0.3	0	0.15
Consumers	0.045	0.05	0	0.035
Employment	0.02	0.05	0	0.005
Total	0.245	0.40	0.60	0.64
Rank	4	3	2	1

4.4.6 Sensitivity Analysis

The final task that would be undertaken is a sensitivity analysis of the results. For example, this might include calculating how much the costs of adopting use restrictions would have to decrease for this to become the preferred option. Or, the sensitivity analysis might examine how alternative weighting systems, for example reflecting more extreme views as to the importance that should be attached to the different criteria, may affect the ranking that would be assigned to each of the options.

4.4.7 Making Recommendations

The final stage of the process would be to prepare a summary of the analysis findings. For an MCA-based analysis such as that described above, this is likely to include:

- a discussion on the scoring systems, the underlying data and the approach taken to standardisation;
- a description of those who were involved in assigning the relative preference weights and the process adopted in eliciting the weights;

- a summary of the analysis results, highlighting the trade-offs involved in adopting one option versus the others and the end ranking of options;
- the findings of the sensitivity analysis and its implications for the ranking and choice of options; and
- recommendations as to any further analyses (*e.g.* widening the criteria considered in the analysis, incorporating other stakeholders' preferences into the weighting systems).

In addition, it is likely that any final report would address other issues that were raised during the analysis process but not explicitly incorporated in the MCA. For example, this might include options screened out of the analysis in the early stages, criteria put forward as the basis for decision making but not included in the end analysis, data on additional impacts that might arise from implementation of any of the options that were not considered in the MCA, and so on. In this regard, there is no reason why final reporting could not also include the same type of supplementary assessment of wider impacts as may be likely for economic analyses.

5. REFERENCES

Allais M (1991): Cardinal Utility History, Empirical Findings, and Applications: An Overview, Theory and Decision, Vol.31, pp.99-140.

Arnold F (1995): Economic Analysis of Environmental Policy and Regulation, New York: John Wiley & Sons.

Bailey D & Parikh A (1990): Techniques of Economic Analysis, London: Harvester Wheatsheaf.

Bateman IJ & Langford IH (1997): Non-Users Willingness to Pay for a National Park: An Application and Critique of the Contingent Valuation Method, Regional Studies, Vol. 13, No.6, pp.571-582.

Beinat E (1997): Value Functions for Environmental Management, Dordrecht: Kluwer Academic Publishers.

Boero G *et al* (1991): *The Macroeconomic Costs of Controlling Greenhouse Gases: A Survey* in Environmental Economics Research Series, London: Department of the Environment.

Bohringer C (1998): The Synthesis of Bottom-Up and Top-Down in Energy Policy Modelling, Energy Economics, 20, pp. 233-248.

Cal & Holahan (1983): as referenced in Zerbe R and Dively D (1994**):** Benefit Cost Analysis: In Theory and Practice, New York: Harper Collins College Publishers.

Cline WR (1992): The Economics of Global Warming, Washington DC: Institute for International Economics.

CSO (1995): Input-Output Tables for the United Kingdom 1990, Central Statistics Office, London: HMSO.

Currim I & Sarin R (1992): *Robustness of Expected Utility Model in Predicting Individual Choices,* Organizational Behavior and Human Decision Processes, vol.52, pp544-568, as reported in Drummond *et al*, 1997 (see below).

Drummond MF, *et al* (1997): Methods for the Economic Evaluation of Health Care Programmes, Oxford Medical Publications, Oxford: Oxford University Press.

EC (1996): Methodologies for Impact Assessment, EC – DG VII: Transport Research APAS Strategic Transport, Luxembourg: Office for Publications of the European Communities.

EC (1997): Working Paper on *Risk Management*, Document Reference: Doc.97/RiMa02.

EC (1998): Technical Guidance on Development of Risk Reduction Strategies, Directorate General XI, Brussels: European Commission.

Frankhauser S & McCoy D (1995): *Principles of Environmental and Resource Economics: A Guide for Students and Decision-Makers*, in Modelling the Economic Consequences of Environmental Policies, (Eds: Folmer H, Gabel HL & Opschoor H), Edward Elgar Publishing Ltd.

Gramlich E (1990): A Guide to Benefit Cost Analysis, 2nd Edition, New Jersey: Prentice-Hall Inc.

Hufschmidt M *et al* (1990): Environment, Natural Systems and Development: An Economic Valuation Guide, Baltimore: The John Hopkins University Press.

Janssen R (1992): Multiobjective Decision Support for Environmental Management, Dordrecht: Kluwer Academic Publishers.

Keeney RL (1982**):** Decision Analysis: An Overview, Operations Research.

Keeney RL (1992): Value-Focussed Thinking: A Path to Creative Decision making, Cambridge, Massachusetts: Harvard University Press.

Keeney, RL & Raiffa, H (1993): Decisions with Multiple Objectives: Preferences and Value Tradeoffs, (2nd Edition), Cambridge: Cambridge University Press.

Keeney, RL & Raiffa, H (1976): Decisions with Multiple Objectives: Preferences and Value Tradeoffs, New York: John Wiley & Sons.

Lazo JK, McClelland, GH & Schulze WD (1997): *Economic Theory and Psychology of Non-Use Values*, Land Economic, Vol.73, No.3, pp.358-371.

Mishan EJ (1994): Cost-Benefit Analysis, Fourth Edition, London: Routledge.

OECD (1997): Environmental Policies and Employment, Paris: OECD.

Ramsberg & Sjoberg (1997): *The Cost-Effectiveness of Lifesaving Interventions in Sweden*, Risk Analysis, Vol.17, No.4, pp.467-478.

Russell *et al* (1996): *The Role of Cost-Effectiveness Analysis in Health & Medicine*, Journal of the American Medical Association, Vol.276, No.14, pp.1172-1177.

Saaty TL (1980): The Analytical Hierarchy Process, New York: McGraw Hill.

Saaty TL (1996): Decision Making for Leaders, AHP Series, Pittsburgh Expert Choice.

Schoemaker, P (1991): *Choices Involving Uncertain Probabilities - Tests of Generalized Utility Models*, Journal of Economic Behavior and Organization, Vol.16, pp295-317.

Sugden R & Williams A (1978): The Principles of Practical Cost Benefit Analysis, Oxford Oxford University Press.

Tengs TO et al (1995): Five Hundred Life-Saving Interventions and their Cost-Effectiveness, Risk Analysis, Vol.13, No.3, pp.369-390.

Tversky, A & Kahneman, D (1992): *Advances in Prospect Theory: Cumulative Representation of Uncertainty*, Journal of Risk and Uncertainty, Vol.5, No.4, pp.297-323.

von Neumann J & Morgenstern O (1947): Theory of Games and Economic Behaviour, 2nd Edition, Princeton NJ Princeton University Press.

PART 3: THE ASSESSMENT OF COSTS AND BENEFITS

1. INTRODUCTION

To provide a more detailed discussion on the application of SEA, this Part examines how assessments are carried out in practice, presenting descriptions of the various techniques that are available for assessing costs and benefits.

In contrast to Part 2, which gives an overview of the main assessment tools, the discussion provided here has been organised by impact type: compliance costs, human health benefits, environmental benefits and equity considerations. In each of these cases, an overview is given of the types of impacts of concern, with this then followed by discussions on how such impacts are analysed using monetary and non-monetary assessment techniques and the types of issues that arise in such assessments.

More specifically, this Part has been organised as follows:

- Section 2 discusses the assessment of the costs (both positive and negative) of adopting a risk reduction measure, including a discussion of what should be considered when examining the economic costs of a proposed risk reduction option and when considering the financial implications of proposed options;
- Section 3 reviews the techniques for assessing human health and safety benefits (both positive and negative), with this including a review of both economic and other assessment techniques and some of the issues that their use raises;
- Section 4 examines the assessment of environmental benefits (both positive and negative), with this again examining the use of both economic and other assessment approaches and common analytical issues; and
- Section 5 discusses the assessment of equity effects, which can be viewed as a cross-cutting analytical issue, given the increasing need to consider how impacts are distributed and, thus, who the ‘winners’ and ‘losers’ are for a proposed option.

To help in understanding how assessments of costs and benefits might be carried out in practice, the ZOZ case studied is further developed in this Part. This has involved making up a range of data for the purposes of illustrating the assessment process. In some cases, problems have deliberately been created for the purposes of illustration, so it is important to remember that this is an imaginary case study.

2. IMPACTS ON PRODUCTION, USE AND CONSUMPTION

2.1 Introduction

This section focuses on the impacts of a proposed risk reduction option on the producers and consumers of the chemical, or of goods produced using it. Whatever methodology has been adopted as providing the basis for the SEA, decision makers will want to have at least some idea of the costs arising from the adoption of proposed risk reduction options. In general, they will want information on both the magnitude of these costs (whether they represent increases or decreases) and on their likely significance to those who would bear them.

For the proposals which may have significant effects on a particular sector or activity, decision makers are also likely to want information on the degree to which an option may affect any other sectors and on any potential employment effects. Again, such issues may be relevant regardless of the methodology selected; however, the manner in which they are treated will vary.

The remainder of this Section reviews the assessment of compliance costs. It:

- starts by setting out what we mean by compliance costs and the difference between economic and financial definitions of these costs;

- then reviews the theory underlying the assessment of compliance costs, with this being relevant to all methodologies;

- discusses applying the theory in practice with regard to the estimation of the direct effects of proposed options;

- reviews the assessment of the indirect or secondary effects on related markets and employment, with regard to the use of both monetary techniques and non-monetary measures; and

- finally, gives a summary of the potential sources of data for undertaking such assessments.

2.2 Costs of Compliance

2.2.1 What are Compliance Costs?

Often the introduction of risk reduction options, particularly where these relate to regulatory controls, will tend to raise the production costs faced by companies producing or using the substance of concern. Costs may increase owing to the direct need to respond to restrictions or through changes in monitoring, reporting and other administrative requirements. In some cases, companies may be prevented from passing any increases in costs on to their customers. In other cases, companies will raise the price of their output to other companies (with such increases occurring directly or indirectly depending on the nature of the interactions between sectors) or to consumers.

The significance of any changes in costs will depend, to a large degree, on the nature of the chemical uses that are under investigation. Most chemicals act as 'intermediate products', which are primarily used in the production of other goods. Their value will usually constitute only a small part of the total value or cost of the final product. As a result, any impacts on the cost to other companies or consumers is likely to be small (equivalent to the increase in costs multiplied by the proportion of those costs as part of total revenue). As an example, consider a regulation that leads to a 20% increase in the cost of a chemical input which constitutes only 5% of end product costs; this will lead to a potential price increase to the consumer of the product of around only 1% (assuming equal product performance and a stable market share).[31] In this rather simplistic situation, the likely impact on consumers would be small. The focus of assessment for this type of chemical therefore is likely to be on the impacts of the increase in costs to its producers (rather than to its customers).

In contrast, where a chemical accounts for a large proportion of end product costs, the impact on consumers (which include other related markets as well as end consumers) may be much more significant. Determining the significance of such impacts on consumers requires information on the relationship between changes in price and changes in demand (*i.e.* the price elasticity of demand). For most chemical products, this type of information is unavailable. However, where a regulation impacts upon widely consumed or highly integrated products (such as the costs of energy production), then data on the price elasticity of demand may be available.

In determining compliance costs, it will generally be helpful to view a chemical as part of a package providing a service to those using it. Although it should be possible to trace through the impacts of a proposed option over a chemical's life-cycle of use (in terms of value added), it may be more appropriate to identify key industries which are as 'near' to the consumer of the end-product as possible, and for which the change in chemical costs has a clearly identifiable impact. Not only will this help ensure that double-counting of effects is avoided, but it will also help focus data collection and analysis. Where there is more than one use of concern, separate analyses will need to be carried out for the different applications.

2.2.2 Economic versus Financial Costs

As a starting point to any discussion on the assessment of costs within a SEA, it is useful to distinguish between the concepts of economic costs and financial costs. As previously discussed, economic appraisal is aimed at the allocation of scarce resources, and the economic value of a good (or service) is the full value of the scarce resources that have been used in its production. The value of these resources are, in turn, measured in terms of individuals' willingness to pay for them (or willingness to accept compensation for their loss). This is the concept of opportunity costs, in other words the value of the foregone opportunities of using scarce resources on the next best alternative.

Within the context of compliance with risk reduction requirements, opportunity costs include:

- the real resource costs of complying with the requirements;
- transitional costs arising to industry and regulators associated with changing systems, training, *etc.*;
- the costs to government agencies in administering, monitoring and enforcing the requirements;

[31] This is of course an over-simplistic example of what happens in practice. The actual impact of an increase in production costs on a product's market price will depend on the price elasticity of demand and other market factors; see also Section 2.3.3.

– any welfare losses to producers and consumers, including gains or losses arising from change in product quality or availability; and

– indirect costs to other sectors supplying services to or buying goods from the affected industry sector.

Added together, these economic opportunity costs provide a measure of the social costs of a policy as measured in economic efficiency terms. Economic opportunity costs may differ significantly from financial estimates of compliance costs. First, the full financial costs of producing a good may not be reflected in the price charged for the good, with examples including the use of own labour or where prices are distorted as a result of market failure and/or government intervention (*e.g.* government subsidies, monopolies, externalities, *etc.*). This type of issue commonly arises when estimating transitional costs and regulatory costs to government agencies. Financial costs will also often include elements that do not relate to the consumption of scarce resources; examples include taxes which are effectively a transfer payment from the private sector to the public sector. In addition, financial cost estimates will fail to account for any welfare losses associated with changes in product quality or product availability.

Externalities (whether positive or negative) will also not, by definition, be reflected in the financial costs of producing a good (see also the discussion in Part 2). However, the key externalities in relation to chemical risks are likely to be related to human health and the environment. Within SEA, these are dealt with through the extension of the analysis to include consideration of the health and environmental benefits (and/or costs) of adopting proposed risk reduction options.

The first three reasons for differences arising between estimates of economic and financial costs are then those that are the most important to estimation of compliance costs. In many cases, the differences caused by such factors are unlikely to be significant and one can assume that financial cost estimates provide a good surrogate measure of economic costs. In other cases, this will not be true, with the need to make a distinction greatest when assessing compliance costs in a developing country situation (see for example ODA (1996) for further discussion).

For most appraisals, one will want information on financial costs for reasons other than their use as surrogate estimates of the social costs of a proposed measure. It is these private costs that are important to examining the distributional effects that a policy may have on different industry/business sectors and on small, medium and large companies.

2.3 Estimating the Impacts on Producers and Consumers

2.3.1 The Concepts of Producer and Consumer Surplus

Following on from the principles of economic theory outlined in Part 2, estimating the marginal costs (in opportunity cost terms) of adopting a risk reduction measure should be based on examination of changes in producer and consumer surplus. These measures can be used to provide an approximation of the changes in welfare associated with changes in prices. They are fundamental concepts within welfare economics, as they indicate the minimum and maximum price that either the producer is willing to accept or the consumer is willing to pay.

2.3.2 Measuring Changes in Producer and Consumer Surplus

In terms of producer surplus, a private sector company will face a variety of options in determining how to produce its outputs or services. The imposition of a restriction on a particular activity or the use of a particular substance will tend to raise production costs. This will in turn raise the price of its output to other firms and (directly or indirectly) to consumers. It will also tend to reduce the company's profits. Such a situation is shown in Figures 2.1 and 2.2.

In Figure 2.1 it is assumed that a risk reduction option will lead to an increase in production costs, with this shifting the supply curve upwards to the left (indicating that an increased price must be charged for a given quantity). Under the pre-policy supply curve, production is represented by curve S_1 and producer surplus by area A. The post-policy supply curve is then S_2 with producer surplus now represented by area B. The shift in the supply curve also has an impact on consumers. The loss in consumer surplus is illustrated in Figure 2.2, and is represented by area C (because as the amount that the consumer actually has to pay increases, his surplus willingness to pay decreases).

These effects provide the two measures of lost surplus. The loss in producer surplus is calculated by determining the difference between area A and area B. The loss in consumer surplus corresponds to what is referred to as the Harberger-Meade welfare loss triangle, which represents the net loss in consumer surplus and is given by the difference between area C and area B. The overall loss of surplus is, therefore, measured as A+(C-B).

Figure 2.1: Change in Producer Surplus

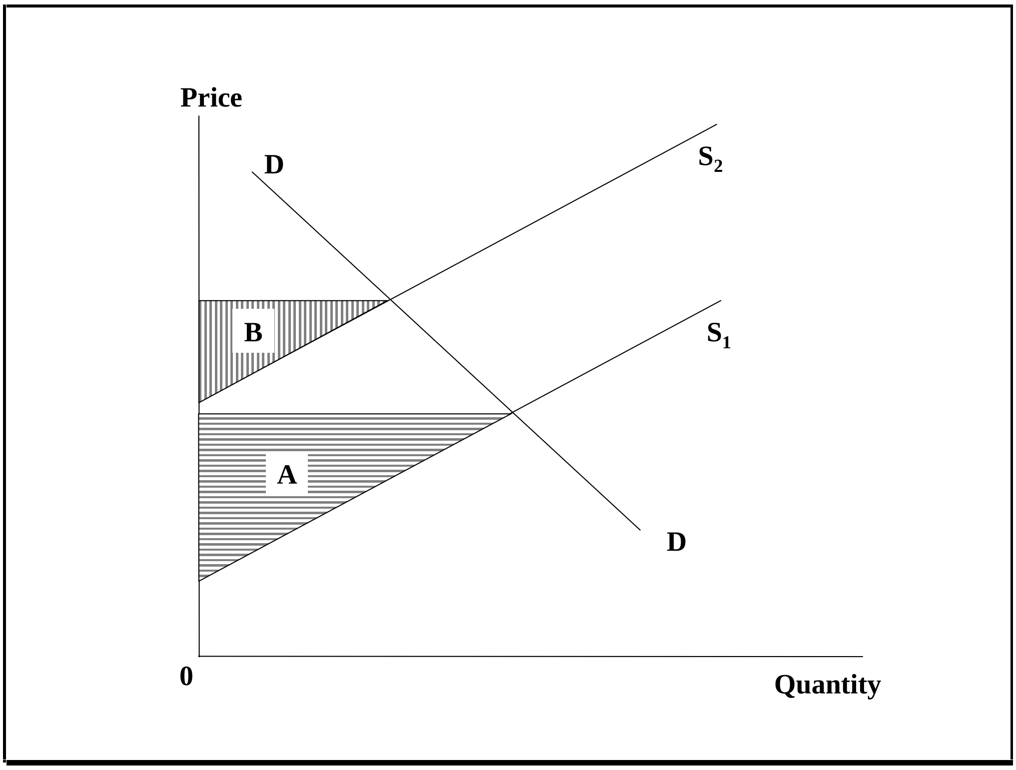
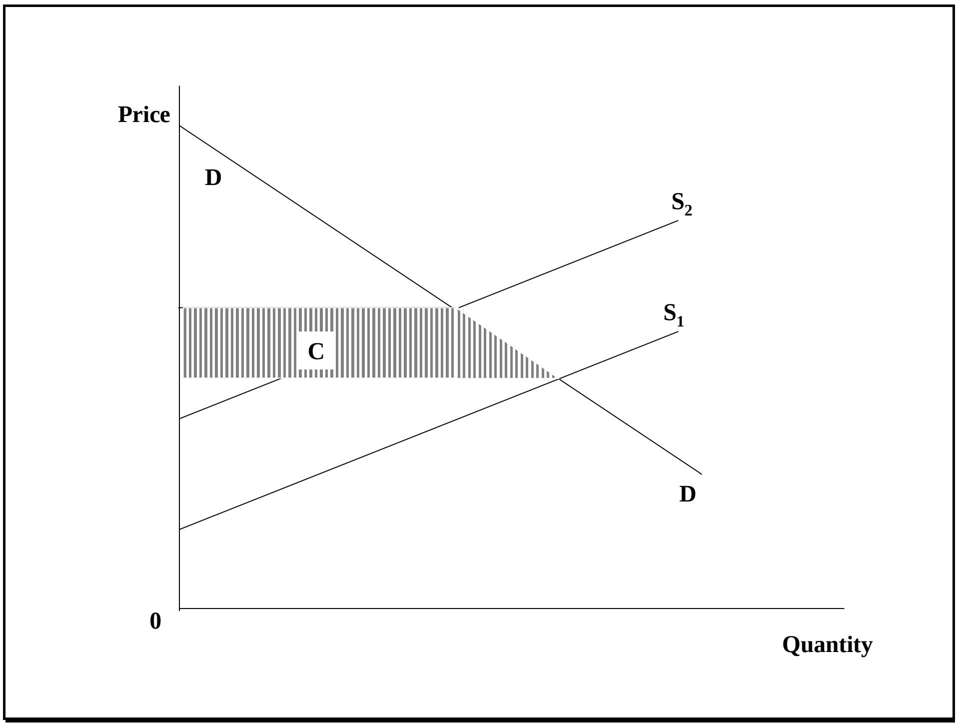


Figure 2.2: Change in Consumer Surplus



The relative importance of producer versus consumer surplus losses will depend upon the nature of the product under consideration:

- products that act as 'intermediate goods', and which are primarily used in producing other goods, will constitute only a small part of the total costs of the final good. As a result, it is likely that only changes in producer surplus will need to be considered; while
- products that account for a large proportion of end product costs may suffer
- from impacts on both producer surplus and consumer surplus.

In the latter case, the extent to which there will be impacts upon producer and consumer surplus very much depends on the product in question and the relative size of its price elasticity of demand and supply.

2.3.3 Price Elasticity of Demand

The price elasticity of demand of a product refers to the relative change in quantity demanded to a change in price; in other words, it is a measure of the sensitivity of demand to a price change (a useful discussion with regards to regulatory cost incidence is provided in Arnold (1995)):

$$e_p = \frac{\text{proportionate change in quantity demanded (q)}}{\text{proportionate change in price (p)}}$$

$$= \frac{Mq/q}{Mp/p}$$

$$= \frac{Mq\ p}{Mp\ q}$$

When the price elasticity of demand for a given product is below one, it is said to be price inelastic. In such cases, the quantity demanded will change proportionally less than the change in price. When the price elasticity of demand is greater than one, it is said to be elastic. In these cases, the quantity demanded will change proportionally more than the change in price.

Knowledge of the elasticity of the demand for products will provide an indication of how much of a price change can be passed on to consumers (or the next firm in the chain of production), hence determining the relative share of lost (or gained) producer and consumer surplus. A number of factors may influence the elasticity of demand for a product (or service):

- substitutability: if a product has many substitutes (with potentially differing risks) then demand will tend to be elastic;
- necessity: if the product in question is essential to a production process then its demand will likely be inelastic;
- time: it is likely that over time demand becomes more elastic. Time allows the development of substitutes and allows for changing consumer patterns.

The most important determinant of the price elasticity of demand for a chemical product is likely to be the availability of substitutes at prices that are affordable to consumers. There will of course be other related considerations such as the characteristics of the markets of concern, degree of competition within the sector, *etc*.

2.3.4 Substitutes

The above is based on the assumption that the regulation or restriction of a product does not lead to any deterioration in the quality of the final goods produced. It also assumes that there exists sufficient flexibility in technology for the companies involved to make alternative arrangements to substitute for the controlled product or the current production technology as appropriate. This may not be a valid assumption in all cases, particularly with regard to drop-in substitutes, as:

- drop-in substitutes may not be available;
- drop-in substitutes may have a different efficacy; and
- drop-in substitutes may require regulatory approval, with impacts on cost and schedule ranging from minimal to unacceptable.

In such cases, estimates of lost consumer surplus (where likely to be significant) may need to be augmented to take quality loss into account. Theoretically, this value of quality loss could be estimated using standard valuation techniques (for example, through the elicitation of willingness to pay for the retention of quality using survey techniques).

2.4 Assessing Direct Effects

2.4.1 From Theory to Practice

Given the above theoretical background, how then are the social costs of compliance assessed in practice? The answer is that this relies on estimating the predicted magnitude of the actual costs that will be incurred by producers when adopting a given risk reduction option.

It is important to note that it is the change in costs stemming from the introduction of a policy that should be assessed within a SEA, although this change can be measured at different levels of disaggregation. In order to determine this change in costs, the baseline for the analysis must be defined (see also the discussion in Part 1, Section 5). It is the changes in activity compared with this baseline that should be costed when estimating compliance costs. In determining what these changes would be, it is important therefore that trends in usage of the chemical under consideration, technological innovation and product development are taken into account if the assessment is to properly reflect the true costs that will be faced by industry (and benefits resulting from any risk reduction activity).

Related to the definition of the baseline is setting the time horizon for the SEA itself as this will also affect what costs (and benefits) will be taken into account (see also Section 5 of Part 1). As previously discussed, it is generally preferable to set the baseline according to how long the chemical is expected to remain in use in the absence of risk reduction requirements; that is, for the remaining economic life of the chemical, taking into account the market for the associated products. Where this is unknown, it may be appropriate to set the time horizon in terms of the expected 'life' of the various risk reduction options (for example, where measures are based on the installation of treatment plant). Linked to decisions concerning the baseline, is the issue of discounting and how costs and benefits occurring in the future will be converted so as to form a common basis with costs and benefits occurring immediately.

It is also important that care is taken to identify the full range of potential costs that are expected to arise under a proposed policy, where these may include:

- non-recurring costs (or one-off costs);
- recurring costs (or revenue/annual costs);
- hidden costs; and
- indirect or secondary costs.

The derivation of such data for a SEA will usually make use of consultation and survey techniques in order to elicit cost data either directly from the companies affected or through the appropriate trade association.

Both the different levels of disaggregation that can be adopted and the various types of costs that may arise are discussed further below.

2.4.2 Total, Average and Marginal Costs

The costs of complying with a proposed risk reduction option are the incremental (or additional) costs that would arise over the time horizon of the analysis. In estimating compliance costs, a further distinction is made regarding the measure of costs that is being estimated:

- total costs (TC) are the sum of all cost components over time (*i.e.* the sum of discounted one-off and recurring over the analysis time horizon);
- average costs (AC) are the costs per unit of good (or service) produced (or provided) as is derived via:

  $AC = TC/Q$ *i.e.* total cost divided by the quantity produced

- the marginal cost is defined as the change in costs associated with producing one more unit of the good (or service) in question and is derived via:

MC = dTC/dQ — *i.e.* the change in total cost divided by the change in quantity produced or the derivative of total cost with respect to quantity.

An example of these different types of costs is given in Table 2.1.

Table 2.1: Example of Average, Marginal and Total Costs

Product	**Level of Production (Q)**	**Non-recurring Costs (K)**	**Recurring Costs (R)**	**Total Costs (TC = K+R)**	**Average Costs (TC/Q)**	**Marginal Cost (dTC/dQ)**
A	100	200	50	250	2.5	-
B	150	225	60	285	1.9	0.7
C	200	250	70	320	1.6	0.7
D	250	275	80	355	1.4	0.7

The concept of marginal costs is central to economic appraisal and determination of the most economically efficient outcome. Within the context of chemical risk management (and environmental regulation more generally), marginal costs can be viewed as the additional (or incremental) costs of introducing a measure that would reduce risks by a specified amount. In this sense, they can be defined as the rate of change of total costs with respect to the level of damage avoidance. In economic efficiency terms, action should be taken up to the point where the marginal benefits of risk reduction are equal to the marginal costs of achieving that reduction. The concept therefore is central to determining what level of risk reduction is justified when using CBA, for example. If the marginal social costs of a proposed option are greater than the marginal social benefits then the option is not economically efficient; if the marginal social benefits are greater than the marginal social costs, then the option is justified on the basis of economic efficiency.

2.4.3 Non-recurring Costs

The types of costs that may need to be considered within an analysis can be classified into two types: non-recurring and recurring. Non-recurring costs will relate mainly to capital costs, but more generally are the additional one-off costs generated by the new or amended regulation. They include the purchase of any necessary equipment, the costs associated with its installation, and any other one-off requirements (transitional costs) arising from the implementation of an option.

Examples of typical non-recurring cost items are:

- the purchase and installation of plant and machinery;
- the purchase or construction of buildings and infrastructure;
- legal and other experts’ fees;
- training and other associated start-up costs;
- machinery or production down-time arising from the installation of new equipment;
- the purchase or modification of computer systems;
- product research, development and marketing costs; and
- loss of inventory and productive capital if an adequate transition period is not allowed.

2.4.4 Recurring Costs

The second type of costs is recurring costs. These are the additional on-going costs generated by a proposed option and may occur on an annual basis or be more periodic in nature (*e.g.* arising every five years).

Examples of typical recurring cost items include:

- staff costs or time;
- raw material costs and other consumables (energy, utility costs, chemical inputs);
- waste treatment and disposal;
- maintenance activities and replacement parts;
- regulatory sampling, testing and monitoring costs;
- reporting and other on-going regulatory compliance activities.

Some of the recurring costs arising from the need to make changes in production processes, in order to accommodate a switch to an alternative chemical or as part of more fundamental changes in process technology, may not be obvious at first. One example of a hidden, and potentially unquantifiable cost, is the impact which regulations may have on innovation by reducing the options open to companies as part of new product or process development

2.4.5 'Hidden' Costs

In addition to the more traditional non-recurrent and recurrent cost considerations, it may also be important to identify what are sometimes referred to as 'hidden' costs. These relate to the direct costs of implementing an option that are not immediately related to the change in technology or chemical inputs. These hidden costs may be borne by the private sector or by the public sector.

Hidden costs may arise from:

- institutional and human resource changes needed to implement the option, such as increasing monitoring capacity and undertaking skills development;
- changes in information requirements, for example associated with the need for compliance monitoring and enforcement activities;
- changes in market size and structure and, hence, in opportunities for innovation and long-term cost reduction; and/or
- any changes in the incentives provided to companies to encourage compliance.

The above may be categorised as either transitional administration costs or barrier removal costs. The former are the costs of activities that are related (and limited) to short-term implementation of the risk reduction option, *e.g.* planning, training, monitoring, and so on. The latter refers to the costs associated with reducing or minimising any transactions costs (the costs incurred in facilitating the exchange of goods and services).

2.4.6 Productive Capital and Residual Value

The previously described costs effectively relate to costs stemming from changes in current production activities. These additional costs should be distinguished from the 'sunk' costs associated with the purchase of the original productive capital (equipment and/or facilities). The changes in production required by risk reduction may, however, result in the loss of the original productive capital, either directly as a result of controls on the process itself or through restrictions on the marketing and use of the end-product(s) from the process. Because these 'sunk' costs have been irrevocably committed they should be ignored within an economic analysis.

Industry has often argued in the past though that the residual value of such lost capital assets should be included as a cost of the regulatory option. From an economic perspective this argument is incorrect; indeed, any residual or re-sale value should be off-set against the additional costs of risk reduction.

2.4.7 Product Innovation and Development

It may well be the case that the introduction of a risk reduction option - be it in terms of a complete phase-out, restricted use of a hazardous substance or even labelling or handling requirements - will encourage (and increase) product innovation and development. Such developments may enable industry to meet risk reduction requirements at a lower cost and with more effective or improved products.

Innovation is, of course, a long-term objective of industry and risk reduction requirements, therefore, may only have the effect of bringing forward future developments or providing an increased incentive for investment in innovative research to produce solutions more quickly. For example, Bucht (1998) states that "...a ban may be an incentive for innovative and profitable development of less hazardous alternatives injecting dynamics into a dead-lock situation...". Two situations where this has occurred are highlighted: in the field of pesticides and following the Montreal Protocol on the phase-out of the production and use of CFCs.

Major international agreements (such as the Montreal Protocol, the Kyoto principles, and regulations that span many countries) may have the effect of speeding up the innovative cycle, so that new developments occur more quickly than if an option were introduced gradually at a regional or national level. The globalisation of the world market also means that firms who react the fastest to the potential for new products and markets will have the possibility of increasing market share and profit margins.

Innovation may not only bring benefits to particular firms, but may also benefit a sector as a whole by reducing costs, increasing product performance or reliability, or creating new spin-off products. Consumers may also benefit from reduced prices, increased choice, the purchase of more environmentally friendly goods, and so on. The degree of any price change or of any change in the range of options available will be product dependent, being a function of both the nature of the innovation and of the end-product. Where consumers are aware that goods have been produced with less damaging substances, there may be additional public relations benefits for the firms involved. Although such benefits are likely to be unquantifiable within any given assessment, they may be seen in terms of strong consumer loyalty, increases in the share price and so on.

In general, assessing the degree to which innovation may result from any particular risk reduction proposal will be difficult. Unfortunately, a review of the available literature does not provide rough and ready 'rules of thumb' that can be used to estimate the degree to which innovation may reduce the costs of risk reduction. Discussions with industry will highlight whether an option is likely to force the development of new technologies or whether any technological developments are at a 'near' market stage. In this regard, it may be valuable to consult:

- producers of the chemical of concern;
- producers of substitutes to the chemical; and
- other actors within the chain of trade who may be aware of research and development activities aimed at reducing reliance on the chemical of concern.

It is important to recognise that once a chemical has been identified as a potential priority for chemical risk reduction, particularly at the international level, then industry will begin research into and development of alternatives.

In addition, where restrictions on the use of an existing chemical would result in the increased use of a substitute chemical, then the price of the substitute may decrease in the medium to long-term. As demand increases for a good (or service), the marginal costs of producing the next unit of that good will usually decrease. As a result, over the longer-term the market price demanded by suppliers of the good will fall (although in the short-term prices may increase or remain constant). The implication is that price increases associated with the switch to a substitute chemical are likely to be reduced over time as the market for the substitute increases or as new suppliers enter the market. The same arguments hold for the costs of adopting new technologies.

2.4.8 Timing of Costs

Finally, it is worth noting that the magnitude of any compliance costs may be highly sensitive to the timing of the proposed risk reduction option. The costs arising from an option that allows new technology to be introduced at the design stage of regular programmes of plan renewal may be significantly lower than those arising from an option which requires immediate implementation of an end-of-pipe solution or output reduction. As part of any cost analysis, it may thus be important to consider different scenarios concerning how an option would be implemented.

2.5 Indirect Effects and Employment

2.5.1 Introduction

Indirect or secondary effects are essentially the side effects or policy spill-over effects that arise from adopting a particular policy option. They concern the changes in output in related sectors of the economy through backward and forward production linkages. They are the second or third round responses associated with the inter-industry demand for different goods and services (in other words the changes in demand for factor inputs and related services induced by the first round of expenditure on environmental risk reduction).

As discussed above, because most hazardous chemicals act as intermediary goods and will comprise only a small proportion of end-product costs, the potential for secondary effects to arise from risk reduction is likely to be minimal; in such cases, they can be ignored. Only in cases where regulations would have a significant effect on highly integrated sectors of the economy, or on widely used primary or intermediary products, will an assessment of the impacts on related markets be necessary (for further discussion see Koppl *et al*, 1995; Hahn, 1996; and Gray, 1997).

The remainder of this section discusses how such effects can be taken into account using both economic and non-economic appraisal methods. The former is aimed at allowing them to be included in a CBA, while the latter is aimed at aiding their inclusion in the assessment of wider considerations as part of a CEA or including them more directly in a MCA. These types of analyses are often performed as complements.

2.5.2 Economic Assessment of Indirect Effects

If a CBA or CEA of a proposed environmental policy is based on appropriate opportunity costs (or shadow prices) for changes in compliance costs, then the estimated net present value of a policy will include all direct positive and negative costs (and potentially some indirect costs). In other words, the measure of net costs will account for all changes in costs arising to those affected by the policy, as well as for any changes in the surpluses associated with changes in economic rent and prices.

Such partial equilibrium analyses, however, may fail to take into account impacts on related sectors (*i.e.* those sectors supplying or buying from the directly affected sector), which as described above may be significant for some cases of chemical risk management. Where this is the case, the CBA (or CEA) will need to be expanded or supplemented. There are three approaches to how this can be achieved.

The first approach is to expand the sectors to which the CBA is applied; in other words, undertake assessments of the changes in producer and consumer surplus that would arise in related sectors. This type of approach would involve expanding the scope of the CBA to a point where the resulting impacts are unlikely to be significant. The positive and negative effects on the costs faced by these related sectors can then be added to those estimated for the directly affected sectors to derive estimates of net costs.

A second approach is to draw on the use of employment multipliers to provide an indication of how changes in expenditure will result in additional changes in output and direct and indirect employment (Abelson, 1996). It is important to note, however, that multipliers only provide an indication of the positive gains in output and employment; no account is taken of any reductions in output or employment that may result from changes in supply and demand resulting from changes in the costs faced by different sectors.

Although multipliers provide 'order of magnitude' estimates of the growth in income and/or output resulting from the capital expenditures stemming from a new policy, they tend not to be included in CBAs.[32] The argument for not including multiplier effect is that secondary benefits are generally viewed as transfers within an economy rather than net additions to the productivity or income of that economy (Abelson, 1996). This argument for ignoring secondary effects presumes, however, that unemployed resources are completely mobile and are distributed evenly throughout the economy. Differences in secondary benefits may nonetheless occur, and may affect total output and incomes if these assumptions do not hold.[33]

The third approach is to call upon the use of macroeconomic modelling techniques, as discussed in Section 3 of Part 2. There is a growing amount of literature surrounding the use of macroeconomic models as part of the process to estimate pollution abatement costs and the impacts that regulations have had on productivity. Much of this research has been undertaken in the US in response to claims that environmental policy damages competitiveness and reduces economic growth. The purpose of the studies has varied from providing more accurate means of predicting compliance

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[32] In the above example, the effects of taxes and imports have been ignored. Taxes and imports serve to lessen the increases in demand for domestic output induced by secondary increases in income, and thereby lower the multiplier. In fact, in an economy where resources are fully employed, the effect of the multiplier is completely neutralised, as all additional demand must be met by imports. In such a case, there are no secondary economic benefits. This is one of the underpinning reasons for employment not being taken into account in cost-benefit analyses.

[33] For further discussions on the use of 'multipliers' and 'multiplier effects', see: Armstrong & Taylor (1985); Black (1981). An illustration of how to derive multipliers from input-output tables is provided in Schofield (1987), as well as an explanation of how to calculate Keynesian and Economic Base multipliers.

costs to determining the net value of direct and indirect costs to the economy more generally. In general, this research has reached varying conclusions with some authors suggesting that the economic effects have been significant, but others concluding that the effects have been minimal.[34]

See also the discussion on competitiveness provided in Section 2.6.2, which addresses some of the issues examined by this research.

2.5.3 Economic Assessment of Employment Effects

Following on from the above discussion, it is important to note that impacts on employment are rarely considered in economic appraisals. The reasons for this are directly related to the arguments presented above for ignoring secondary effects. The assumptions are that:

- the economy is effectively fully employed;
- any measured unemployment is the result of the need to match changing demand for labour to a changing supply;
- because of the need to match skills with changes in demand, there will be periods when individuals can anticipate that they will be out of work as they move from job to job; and
- since such periods are reflected in employment contracts and in unemployment benefits, there is no cost to society from the existence of a pool of unemployed workers.

In reality, the above assumptions do not hold in many countries. This suggests that the assessment of employment effects may be important for chemical risk management proposals that are likely to have significant impacts on the size and structure of one or more sectors, or on employment at the regional level given that the chemicals industry often tends to be geographically concentrated.

How then can changes in employment be valued in economic terms? The first step is obviously to develop an indication of the actual physical change in the number of people employed, where this may include indicators of short-, medium-, and long-term employment, and of whether those jobs are full- or part-time.

Once a physical measure of the net employment effects arising from a policy (whether gains or losses) has been determined, this can then be valued in money terms. The economic gain arising from the creation (or loss) of employment is traditionally defined in terms of:

- the gain (loss) of net income as a result of the new job after allowing for any unemployment benefit, informal employment and work-related expenses;
- the value of the lost (additional) time that the person has at his or her disposal as a result of being employed (unemployed); and
- the value of any health related consequences of being unemployed (employed) that did not exist prior to unemployment (employment).

The economic value of the changes in employment resulting from a policy is then calculated by summing the value of the above and multiplying them by the period over which employment will be created (lost).

[34] See, for example: Robinson, 1995; Gray & Shadbegian, 1994, Morgenstern *et al*, 1997; Morgenstern *et al*, 2000; and Porter & Van der Linde, 1995.

2.5.4 Non-Monetary Assessment of Indirect Effects and Employment

Indirect Effects

As noted earlier, decision makers are likely to want some indication of whether such effects may arise and how serious they may be. Depending on the nature of the proposed risk reduction options, it may or may not be important to incorporate some measure of indirect effects. Although the methods described above may help in quantifying and understanding indirect effects on the output of different sectors and of the economy more generally, their use may not be practical or appropriate for many appraisals.

It may therefore be of value to present qualitative/semi-quantitative indicators of the likely magnitude of output effects. For example, indicators may include:

- the number of related sectors likely to be affected and the activities undertaken by these sectors;
- the profitability and output/turnover of the sector compared to the magnitude of change; and
- the number of companies operating in these sectors and their characteristics (*e.g.* mainly small to medium-sized or larger-sized operations).

The manner in which such impacts are assessed within an MCA will obviously depend on the specific approach that has been adopted for the assessment. For example, impacts could then be rated across policies in relative terms using simple stepped scoring systems.

Employment Effects

As indicated above, prior to undertaking an economic assessment some physical measure of the changes in employment that would result from a policy is required. The types of measures that are used in other decision contexts (*e.g.* urban regeneration) include the following:

- number of long-term full-time job equivalents;
- number of jobs by occupational skill category;
- change in employment relative to regional or national average rates.

One could always define a range of lower level, more specific indicators depending on the nature of the proposed options under consideration and their likely impacts. Whatever indicators are selected, it will be important to ensure that they are capable of distinguishing between proposed options in terms of the end impacts and that they are appropriate to the assessment approach.

2.6 Financial Considerations

2.6.1 Indicators of Financial Impacts

Many decision makers will want information not only on the net economic costs arising from the adoption of a proposed option but also on the financial implications for individual companies within those sectors and their ability to bear the implied costs. Three key indicators in this regard are likely to include impacts on (see also Helfert, 1997 for further indicators):

- changes in the demand for key outputs and thus impacts on cash flow forecasts;
- the profitability of companies within the sector; and
- the ability of affected companies to service any debt requirements that will arise from the option.

In most cases, it will be important to consider whether the implications of an option are likely to vary by size of operation; in other words, will small firms be affected to a greater degree than medium and larger-sized operations, or will the financial impacts be similar regardless of size.

Related to the question of financial impacts are implications for the competitiveness of the affected sectors. Where an option may have significant effects on competitiveness, this will obviously impact on the financial performance and viability of the companies within the sector.

2.6.2 Competitiveness

Competitiveness basically denotes the ability of a productive sector to sell its goods and services in domestic and world markets. There are many possible indicators of competitiveness, some of which are policy targets in their own right. Underlying these indicators is the assumption that being competitive is important because it enables goods and services to be produced and sold, contributing to increases in sectoral output and incomes. These indicators include:

- income per person;
- balance of trade;
- unit labour costs;
- generation of employment;
- labour productivity;
- market share;
- profitability;
- firm growth; and
- trade intensity share of exports.

As a general rule, at the level of the individual firm, a chemical risk option may have implications for competitiveness if it imposes costs on some firms that are not imposed on their competitors. It may not always be the case, however, that risk reduction would impose costs on firms. It may also generate benefits for companies to set against the costs (*e.g.* income from the sale of recovered by-products, savings in resource inputs, reductions in waste disposal requirements, *etc.*). Moreover, even if an option would impose costs on a company, they may not be substantial enough to impact on its competitiveness, or the firm may be able to pass the cost burden on to customers, even in the short-term.

Christainsen & Tietenberg (1985) identify five reasons why environmental regulations in general may limit growth in productivity, income and, where the policy is not applied to all competing firms, competitiveness:

- investments in more pollution control may crowd out other investment;
- more stringent abatement requirements for new plants may prolong the life of older, less productive, plants;
- pollution control equipment requires labour to operate and maintain with no contribution to saleable output;
- compliance with regulations absorbs managerial and administrative resources with no contribution to saleable output; and
- uncertainty about present and possible future regulations may inhibit investment.

A survey conducted by Dean (1992) found that "…Plants with high compliance costs have significantly lower productivity levels and slower productivity growth rates then less regulated plants. The impact of compliance costs is stronger for total factor productivity than for labour productivity, and stronger for productivity growth rates than levels…". However, Jaffe *et al* (1995, p. 152) state that subsequent work has shown these effects on productivity to be "…largely an artefact of measurement error in output…".

The degree to which the costs of any risk reduction requirements will affect the competitiveness of a firm also depends on the incidence of the compliance costs. In general, the incidence of compliance costs (*i.e.* whether the burden is borne by producers, passed on to consumers in the form of higher prices, or shared by both) varies over the type of regulation imposed and a range of other factors, and it may well change over time. The ability to pass some portion of the cost burden on to customers, however, will help minimise any impacts on competitiveness.

To assess cost incidence, it is first necessary to consider whether the proposed policy options are related to the ongoing economic activities of firms in the affected sector, or whether they impose costs on some firms which are not imposed on their competitors. If the regulatory costs are not related to the ongoing economic activity of the affected firm, then these costs cannot be shifted on to its customers (see Arnold, 1996). Even if compliance costs are related to ongoing economic activity, if they are not imposed on all competing firms then the affected firm cannot raise prices to recover the costs as unaffected firms will be able to undercut the higher prices.

Second, it is necessary to distinguish between costs arising over the short-term and those that are longer-term in nature. "…In the long-run compliance costs will be borne mostly by consumers, and therefore will be widely dispersed and involve few serious economic impacts..." (Arnold, 1996). In the short-run, it is necessary to distinguish between two cases:

- incidence of variable-cost increasing regulations; and
- incidence of capital-cost increasing regulation.

In the case of the latter, "…short-run marginal costs do not appreciably rise, so the short-run industry supply curve does not shift upward, and prices in the short-run will be unaffected…" (Arnold, 1996). In this case, the compliance cost burden is borne by the producers, and they become less profitable. The effect is the opposite when the regulatory costs primarily impact upon variable costs. Of course, any actual regulation is likely to impose a mix of capital and variable costs, so that a hybrid of these two cases is probably appropriate for assessing short-run cost incidence.

As noted earlier, investment to meet risk reduction requirements may also yield net financial gains, and so can be justified in terms of financial returns irrespective of human health or environmental considerations. Smart (1992) gives five reasons why firms can benefit by moving 'beyond compliance':

- reducing risks (*e.g.* through pollution prevention) at source can save money in materials and in end-of-pipe remediation;
- voluntary action in the present can minimise future risks and liabilities and make costly retrofits unnecessary;
- firms staying ahead of regulations can have a competitive edge over those struggling to keep up;
- new 'green' products and processes can increase consumer appeal and open up new business opportunities; and
- an environmentally progressive reputation can improve recruitment, employee morale, investor support, acceptance by the host community and management's self-respect.

All of the above may be translated into monetary returns.

Where the competitiveness of a sector is negatively affected, this may be marked by bankruptcies and job losses. If the affected sectors are significant players in the national or regional economy, for example the affected firms are major export earners, then exchange rate depreciation may occur. This, in turn, may introduce import inflation into the economy, which may result in further negative macroeconomic effects.

If a proposed option would make some sectors uncompetitive, the economy would tend to restructure over time to replace the uncompetitive sectors. This would be at a cost, however, as new firms may not be as productive as the displaced ones. In addition, if economically important sectors lose their competitiveness, this could lead to substantial transaction costs for the economy and, in some instances, a higher equilibrium rate of unemployment. It may also be the case that affected firms may move to countries which have less stringent environmental polices. However, actual evidence of such moves to 'pollution havens' is relatively weak and this may not be a major issue.

Overall, effects on competitiveness are only likely to arise if different countries impose different levels of compliance costs on competing firms. Hence, even though an environmental policy may reduce labour productivity, or reduce rates of economic growth, these effects will only affect competitiveness if they are borne disproportionately by competing firms. The degree to which any chemical risk reduction options would give rise to impacts on competitiveness will depend on the extent to which its implementation is harmonised across all companies operating in the same 'global' market. It should nevertheless be recognised that there will always be differences between countries (such as the appropriateness of some technologies or waste assimilation capacity) which should be taken into account.

In general, most researchers have concluded that the effects of environmental policy on economic growth and employment have been relatively small (OECD, 1985; OECD, 1996).

2.7 Sources of Data

2.7.1 Types of Data

Collection of the data required to carry out the above types of assessment may not be straightforward. At a minimum, it is likely to include data on:

- the linkages within the chemical's life-cycle, moving from production to intermediate use to consumption to disposal;

- levels of production and details of the main manufacturers (both domestic and importation sources) and the basic characteristics of the sector;

- current levels of consumption, both overall and in different specific uses;

- trends in both production and consumption, taking into account past, current and expected legislation; and

- details of potential substitutes, their availability, efficacy and associated risks.

Where a hazardous substance is a minor input to the overall production of a range of end-products (within a given category), few data may be available on the level of consumption in relation to a specific use. At a broader level, end-product statistics may not be available in the form needed for the analysis. Although these problems are more likely to arise in the case of newer chemicals, they may also affect the analysis of many of the more established chemicals. This is particularly true where a chemical has a widely varying range of uses.

The data provided by different sources may also highlight differences in viewpoints and in how changes in the market for a given product are perceived. This is particularly likely to be the case as one moves down the chain of trade for the chemical of concern (*e.g.* from producers to formulators to consumers). It is therefore important to consult a series of different industry sources and research organisations that have a knowledge of different aspects of the chemical's life-cycle across the range of its uses, and of the chain of trade for both the chemical and the associated end-products.

Industry data may be of a commercially confidential or proprietary nature. For example, if figures on production and intermediate use are unpublished, industry may be wary of releasing such information, particularly if the affected market sector is highly competitive. Details on possible substitutes (both drop-in replacements and technologies) are in particular likely to be of a sensitive nature if they are only at a near-market stage of development.

2.7.2 The Range of Sources

In collecting the types of data required for assessing the impacts on producers and consumers, it will be important to consider not only published and grey (unpublished) literature sources but also to consult directly with relevant organisations. Some suggestions on potential sources of the types of data required are listed below, although it must be recognised that across assessments the key sources are likely to vary considerably and require an extension to the suggestions provided here.

Published and Unpublished ('Grey') Literature

- Trade association web sites and trade publications

- Sector and/or product specific journals (*e.g.* on the textile industry and wool manufacturing)
- Research Institute publications
- Market research surveys
- Outputs from other risk reduction work (*e.g. OECD monographs*, work carried out by other organisations, governmental research reports)
- Research reports prepared for individual companies, trade associations and other interest groups
- National and international statistics on trade and levels of economic activity

Consultation

- Direct consultation via face-to-face interviews or by telephone with trade associations and individual companies
- Use of sectoral surveys sent to individual companies
- Direct consultation with relevant regulatory agencies (national and international)
- Direct consultation with relevant non-governmental bodies

As indicated above, it is likely that a range of different sources will have to be tapped, where these include individual companies, trade associations, research institutes and national/international trade statistics. Co-operation from industry is likely to be essential for this process as most cost data will need to come from them. Trade associations can be particularly important as they are likely to have an understanding of the issues already, have contacts with relevant experts and, through their direct links to the affected companies, are likely to be able to generate some, if not much, of the data required.

Although industry is likely to be willing to help (especially if the proposal is to ban a much relied upon chemical), setting unrealistic timescales for completion of a study or making unrealistic time demands may reduce its willingness to co-operate. Both those commissioning assessments and those undertaking them should be aware of the need to avoid this problem.

2.8 The ZOZ Case Study

In Section 2 of Part 2, a brief overview was given of how a CEA and CBA would be carried out for the ZOZ case study. This also drew on the discussion in Section 1 of that Part which set out a series of potential risk reduction options. In the remainder of this section, we examine some of the costing issues in more detail.

The cost analysis will draw upon any market analysis undertaken in the first stage of the risk management process. As will be recalled, this fictitious case study assumes that the hypothetical chemical ZOZ is used across a wide range of different applications, with total annual production and consumption volumes being large (see also Section 1.3 of Part 2).

In total, it is assumed that some 100,000 tonnes (t) of ZOZ were produced in 2000, at a value of around $200 million. Production levels are expected to vary only slightly from this level over the next 10 years. Because of the versatility of the ZOZ group of chemicals, ZOZ are used in a wide range of industry sectors as intermediaries in the production of other chemicals and directly as softeners, emulsifiers and dispersive agents. Risks to the aquatic environment have been identified from the production of ZOZ, as well as for three (out of 12) different applications. These three applications of concern account for around three-quarters or some 75,000 tonnes of ZOZ use. The detailed risk assessment found that the other uses of ZOZ do not pose unacceptable risks to the environment, so they are not considered further. Use in the production of chemical intermediaries also raises some concern for worker health and safety owing to acute and potential chronic respiratory effects. As noted earlier, they may also be public health concerns owing to the presence of ZOZ in the aquatic environment.

Further data would be collected on the use of ZOZ in the three sectors of concern as part of the cost analysis, with Table 2.2 setting out how ZOZ is used in each sector, the life-cycle of concern and data on the quantities used.

Table 2.2: Market for ZOZ in 2000

Sector	**Description of Use**	**Activity Giving Rise to Aquatic Risks**	**Tonnage**
Production of ZOZ	Processing of raw materials into final product	Effluent emissions	100,000
Chemical Intermediaries	Production of a range of different resins and stabilisers	Effluent emissions	35,000
Textile Industry	Used in cleaning and dying processes	Effluent emissions	13,000
Industrial Cleaning Products	Used as an emulsifier and dispersant in products	Use of products containing ZOZ	27,000

From the table it can be seen that ZOZ poses a risk to the aquatic environment (and thus potentially to the general public) through effluent emissions from its production and from two of the use sectors, and through its use in industrial cleaning products. In addition, although the production of chemical intermediaries containing ZOZ (which occurs at only a few sites) poses unacceptable health and environmental risks, use of the intermediaries does not.

With regard to the likely trends in use, producers expect the demand for ZOZ to remain at similar levels in the absence of any risk management action being taken. To date, there has been little effort into the development of drop-in substitutes for ZOZ, although they do exist for most of the relevant cleaning products. The concern surrounding the environmental impacts of ZOZ and the threat of future regulation has already led to a reduction in demand for these applications. Alternatives also exist for use in the textile industry but ZOZ was adopted, in part, as a result of environmental concerns over these alternatives. New technologies are being developed to eliminate the need for ZOZ or its alternatives, but these are not expected to come to market for at least three years, probably five.

It is difficult to predict what will happen in the future, owing to changes in production processes, technological innovation and other factors. As a result, an assumption has been made that costs should be examined over a 15-year period, with this reflecting the remaining life-time of ZOZ in quantities of concern for these sectors. This time period provides the basis for the cost analysis.

A number of options were short-listed as providing a means of controlling the risks arising from the use of ZOZ, with these including voluntary approaches, employee health and safety measures, information programmes, market instruments and command and control measures (see

Section 1.3 and Table 1.2 of Part 2). Based on initial discussions with trade associations and industry experts (*e.g.* from research facilities), the options were defined in more detail.

In part with the assistance of the trade associations, companies in each of the industry sectors were surveyed to gather data on:

- details on the size of the company (levels of production, turnover and employees) to enable companies to be identified as being small, medium or large;

- their current use of ZOZ in tonnes and per unit of production *(e.g.* concentration);

- their likely response to each of the proposed policies and why they would respond in this manner (in each case they were provided with a scenario of how the measure would be introduced, for example, by levying a 30% product tax);

- the costs associated with the proposed response and how effective they believed it would be (they were asked to estimate costs in terms of any necessary one-off investments in capital and research and development, and in terms of any changes in annual revenue or in on-going production costs; in addition, companies were asked about any potential hidden regulatory or non-regulatory costs that might arise);

- if the response involved the use of alternatives, details of what these alternatives would be now and in the near future (chemicals or processes); (in particular, there were questions about the potential for technological development; any quality, efficacy or other issues that might arise in such cases); and

- any other issues that might arise for them as a business, for example concerning customer requirements, competitiveness, employment, *etc*.

In addition, follow-up consultations were held with a selected number of companies from each sector to make sure that as accurate a picture as possible was developed of current practice and likely responses to the different proposals. These consultations and the survey data were supplemented with information collected through a review of the relevant trade and wider literature, and government of statistics covering production, turnover, number of companies, export, import, and employment data. In addition, the potential manufacturers of alternatives to ZOZ were contacted to gather their views on the availability and performance of alternatives (chemicals and technologies). Through this work a model was developed for each of the sectors concerning their likely response to the different proposed policy scenarios.

All together, the work took several months to complete, particularly as trade associations and individual companies had to be allowed adequate time to consider the issues, collect and synthesise the necessary data. Tables 2.3 and 2.4 summarise the data collected for just one sector, the industrial cleaning products industry. For this sector, the risk reduction options that are being considered include a voluntary agreement to reduce the concentration of ZOZ in products to 1% or less through substitution, product labelling (providing the basis for an information programme), a product tax which would be levied on the concentration of ZOZ in the cleaning product, and full marketing and use restrictions (*e.g.* a ban).

Table 2.3: Overview of Market and Use	
Characteristic	**Data Found**
Industry Structure	Over 150 producers of cleaning products, with 80% of these being small to medium-sized companies.
Response to Survey	Only 30% of those surveyed responded, with the majority of respondents being medium- to large-sized companies; poor response from smaller companies. Those responding account for roughly 50% of ZOZ use by this sector.
Importance of ZOZ	Not all companies use ZOZ in their products, with roughly 40% of those who responded doing so.
	ZOZ-based products tend to be specialist products, and account for roughly 10% of the market in industrial cleaning products.
	Concentration of ZOZ in products varies from <1% to around 15% maximum, with ZOZ accounting for up to 30% of production costs at roughly $3,000 per tonne.
Substitutes	Replacement emulsifiers and dispersants would be required in most cases (as opposed to re-formulations not requiring a ZOZ replacement).
	Replacement chemicals are not available for all applications. Considerable uncertainty as to the degree of replacement possible, with estimates varying from it being possible for between 30% to 80% of products.
	Replacements cost on average from between 20% and 40% more than ZOZ, with some research and reformulation costs involved.

As can be seen from Table 2.3, the response rate to the survey was fairly low. In general, there are inadequate data (from the survey and other sources) on the market for the various products containing ZOZ to model changes in producer and consumer surplus. As a result, the marginal changes in production costs arising under each of the scenarios provides the basis for estimating the impact of each of the proposed measures. The result is that considerable uncertainty remains as to the actual impact that the measure will have on both individual companies and the sector as a whole. This is also highlighted by the ranges given in Table 2.4 and the uncertainty attached to the predictions under the various column headings.

In addition to the financial costs indicated in the table, producers of the relevant industrial cleaning products indicate that there may be some additional costs associated with loss of cleaning efficacy and general product performance, with this leading to a loss in sales. The small- and medium-sized companies that did respond also suggest that this could have a disproportionate impact on the more specialist companies, affecting the market size and structure overall; they argue that this would also deter innovation within this sector. Both of these factors would, in turn, lead to the loss of jobs in the sector, particularly in smaller enterprises. The analysis suggests, however, that any such impacts that do arise are unlikely to be significant outside the sector as other companies will fill any voids in the market. In addition, the responses of other suppliers of emulsifiers and dispersants suggest that controls on the use of ZOZ would have a positive effect on innovation.

The focus so far has been on the costs of the different policies to producers and consumers of industrial cleaning products containing ZOZ. There are also likely to be other cost impacts arising under each of the proposed measures that need to be assessed, for example:

- the impact on the producers of ZOZ and the degree to which the measures would have implications for the non-regulated sectors using ZOZ (*e.g.* if the per unit costs of ZOZ increase with lower production volumes);

- similarly, the gains arising to producers of substitutes of ZOZ, who will see production volumes increase potentially enabling the price of these substitutes to be reduced to increase their competitiveness more generally;

- the costs faced by regulators in implementing and monitoring each of the options, with these likely to be highest with the introduction of a product tax; and

- the costs which would arise to industry and other sectors in implementing the options; for example, industry trade bodies would need to oversee, monitor and report on the success of any voluntary agreement.

It is clear that not all of the cost issues that are likely to arise under the various options can be assessed in quantitative terms owing to a lack of data and other uncertainties. As a result, it will be important that the cost estimates presented in Table 2.4 are complemented by a more qualitative assessment in the final comparison of options. Given the concerns over the loss of competitiveness and export markets, the distributional effects may also need to be considered in more detail (see Section 5).

It is also clear that the level of uncertainty affecting the above estimates may need to be examined in more detail (see Part 4, Section 3). In addition, as the costs arise over different time periods, they would normally be discounted to convert them to a common unit of measure (see Part 4, Section 2). Discounting will also be important so that the costs can be combined with the benefits, which may arise over a different period of time (see also Sections 3 and 4, which illustrate the assessment of health and environmental benefits as part of the ZOZ case study).

Table 2.4: Cost Analysis for Industrial Cleaning Products Sector Under Alternative Risk Reduction Options

Measure	Response	% of Those Surveyed	Estimated Costs			Effectiveness of Option	Other
			Non-Recurring	Recurring	Time Period		
Voluntary Agreement	Reduce concentration of ZOZ to <1%	55% of those using ZOZ indicated they would sign up for the Voluntary Agreement, with the remaining 45% saying they would not agree	$17 million in research and reformulation costs	Estimated average 15% increase in raw material costs across 55% of use gives annual costs of $1.8 million (or $18.6 million after discounting)	Non-recurring costs would be spread over 1 to 2 years. Initial increase in raw materials costs may be later reduced with increased production of substitutes	Not all companies would agree to restriction so some level of risk will remain at local and regional levels	Companies have indicated that quality issues would arise in reducing concentrations to <1%. This would probably lead to a loss of sales and production of certain products would cease
Product Labelling	Labelling of ZOZ as dangerous for the environment	All respondents indicated that they could meet this requirement	$3.2 million	Not significant	Non-recurring associated with re-design of labels and would occur in first year	Industry is sceptical that this would be effective, with no guarantee that reductions in use would occur	There is concern that this might single out ZOZ-based products while other equally damaging products remained unlabelled
Product Tax	Tax on the use of ZOZ to increase the costs of using it by 30%	75% of respondents indicated that they would continue using ZOZ and would only increase end-product in line with competitors	$12 million in research and reformulation for a relatively small number of firms	Increase in annual production costs in line with product tax; 30% increase translates to additional $300 per unit of production, leading to increase in annual costs of $8.1 million	Non-recurring costs would be spread over 1 to 2 years. Unclear how costs of substitutes would be affected as this depends on changes in demand for ZOZ and related products in response to tax	Effectiveness depends on the degree to which increases in production costs are passed on to consumers; this cannot be predicted owing to the lack of price-elasticity demand data for this sector. Best estimate is that demand may decrease by 40% at this tax rate	Production of certain products may cease with a loss of sales. Concern over impact on highly competitive export markets
Marketing and Use Restriction	Ban on the use of ZOZ	Only 35% said they could find replacements	$25 million research, reformulation and change in production lines	Between 2% and 4% increase in materials costs, giving increased annual costs of $5 to $10 million	As for Voluntary Agreement	Would be effective	Considerable concern over loss of specialist products and efficacious cleaners. Loss of sales would occur with impacts on sector employment

3. HUMAN HEALTH AND SAFETY BENEFITS

3.1 Introduction

In assessing human health and safety effects, the analysis should consider both changes in the risk of fatality and changes in the risk of morbidity, where the latter can be further divided into acute effects and the incidence of chronic disease. Table 3.1 sets out the types of risk end-points which may need to be considered within a risk assessment for different risk groups under EU risk assessment requirements.[35] It also provides an indication of other related impacts that may need to be examined in the associated SEA.

Table 3.1: Summary of Human Health Risks Considered in EU Risk Assessments

Risk Group	Risk End-Points	Associated and Indirect Impacts
Workers Consumers Humans indirectly exposed via the environment	Acute toxicity Irritation and corrosivity Sensitisation Repeated dose toxicity Mutagenicity Carcinogenicity Reproductive toxicity	Fatalities/deaths brought forward Various morbidity effects Lost working days and non-working day opportunities Health care costs Changes in quality of life Stress effects related to pain and suffering

Source: European Commission Regulation 1488/94 and associated documentation.

Within any SEA, it should be recognised that more than one type of health effect may be relevant, as impacts may vary over different population groups. For example, the nature of worker exposure and the risks of concern to this group may vary significantly from the risks of concern for the general public. Risks to the general public may relate to long-term indirect exposure to a chemical, while worker risks may relate to these as well as potential for direct exposure. As a result, options to reduce one type of risk (*e.g.* to reduce direct worker exposure to immediate risks) may not address all of the risks of concern. The benefits provided by alternative risk reduction options may therefore vary significantly where more than one population is at risk and where the activities giving rise to the risks vary.

The actual assessment of fatality-related (mortality) and morbidity effects can be carried out in a number of ways. At the simplest level, the assessment may be aimed only at indicating that the risk of fatalities or cases of a non-fatal health outcome occurring are reduced to an acceptable level on the basis of acceptable/tolerable dose information. At a more detailed level, the aim may be to determine the change in the likely number of fatalities/non-fatal cases occurring per year for a particular population group or for the public more generally. The first type of information limits the discussion of benefits to one that is qualitative in nature, while the second allows the use of semi-quantitative techniques, such as CEA and MCA. Where the data exist and are considered acceptable,

[35] Following the *Technical Guidance Document in support of Commission Directive 93/67/EEC on Risk Assessment for New Notified Substances* and *Commission Regulation (EC) No. 1488/94 on Risk Assessment for Existing Substances*.

it may be possible to attach a monetary value to changes in the expected number of fatalities/non-fatal cases drawing on the valuation techniques used within CBA.

It is worth emphasising, however, that any effects that remain qualitative or semi-quantified should not be excluded from the analysis. Every effort should be made to present this data alongside the fully quantified data in order to ensure that decision makers are fully informed about the impact of various risk reduction options. A useful tool in this case is the derivation of implicit values that attempts to derive what the value of certain effects will have to be to influence the final decision. Such calculations should also be presented in the SEA to give decision makers all possible decision criteria.

This section sets out the current approaches to assessing impacts on human health using both economic and non-economic approaches:

- the assessment of fatality-related effects using economic valuation techniques is reviewed first, with this including a discussion of some of the key issues which arise in preparing such assessments (Section 3.2);

- a discussion on the use of economic valuation techniques in the assessment of morbidity effects is then provided (Section 3.3);

- the non-monetary techniques used in health economics for assessing changes in health outcomes are then examined (Section 3.4); and

- the section ends with the ZOZ case study and a discussion on data sources for use in human health benefits assessment (Section 3.5).

Table 3.2 highlights the main differences that exist between the application of monetary and non-monetary methods for assessing health effects. It also highlights some of the factors that analysts will want to take into account when deciding upon the appropriate approach for any particular SEA.

3.2 Economic Valuation of Fatality-Related Effects

3.2.1 Introduction

The valuation of fatality-related effects (sometimes referred to as mortality or longevity effects) is based on determining what individuals would be willing to pay (WTP) for either a reduction in the risk of a fatality or for extending life by a year. Such valuations are not concerned with determining the value attached to a particular individual's life but with the value attached to reducing the risk of premature death of all of those who might be affected more generally, even though the probability of death is still far below one (with one signalling certainty). By concentrating on the total sum that all of those who might be affected would be willing to pay to reduce the risk to them, it is possible to value the societal benefit of small changes in risk.

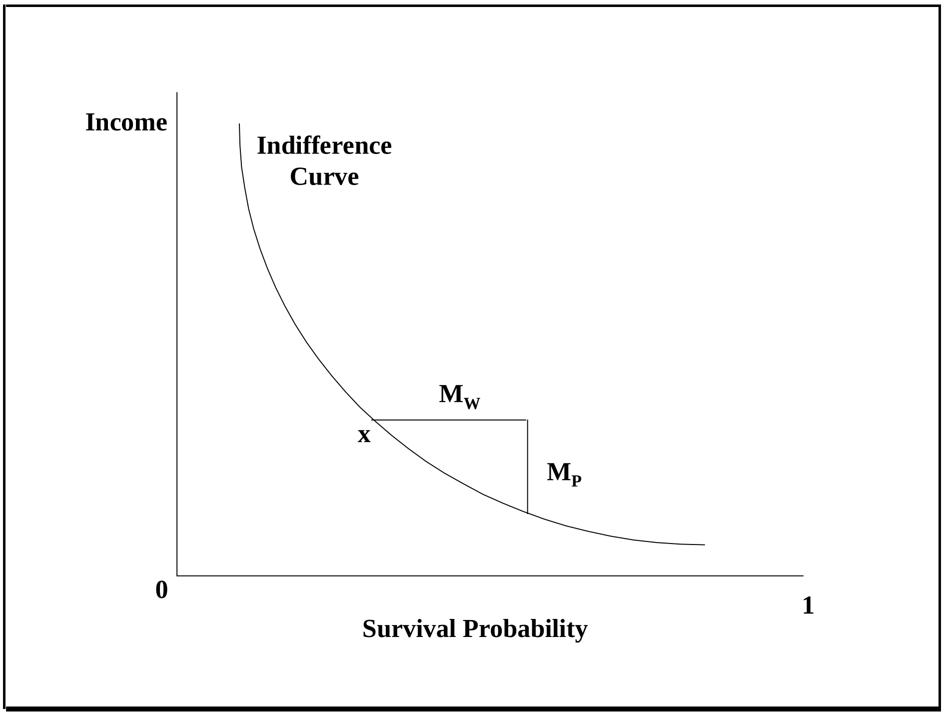
The aim of economic valuation is to develop a monetary estimate for mortality that provides a measure of an individual's preference for a reduction in risk in terms of the amount that he or she would be willing to pay for it. By so doing, an indication is given not only as to how a person values changes in risk relative to other potential goods/services, but also with regard to his or her ability to pay (with this linked to the ability of society more generally to pay for safety).

Table 3.2: Decision Factors for Selecting the Approach Towards Health Impact Assessment

Decision Factors		**Monetary Valuation**	**Multi-criteria Analysis**
Type of Output		The output is a money value that reflects the economic (or social) value of the predicted health impacts. In some cases, these relate to the value placed by individuals on a small reduction in the risk of either death or illness, while in others they reflect actual medical and other costs incurred as a result of illness, or in order to avoid illness.	Health effects are generally assessed in terms of Quality Adjusted Life Years (QALYs) or other similar expressions of health related quality of life. These can be applied to both death and illness related impacts to provide an overall measure of the number of QALYs gained (or lost) as a result of a change in risk.
Acceptability		Monetary valuation of health effects may not be acceptable to all stakeholders or within some cultures. In particular, the use of survey methods to derive valuations for reducing the risk of death can be controversial.	Use of QALYs may be more acceptable to non-economists than use of monetary valuation (although less acceptable to economists). Has widespread application in other fields such as medical intervention decision making.
Process Issues		Few chemicals specific studies exist to provide readily adopted data. Several issues arise in the transfer of data from other risk contexts. New studies may be required to ensure reliable valuations are being applied.	Few empirical studies that have derived QALYs specific to chemicals related health effects may exist. Work may be required to develop QALYs for health effects of concern.
Data Requirements		Data requirements vary considerably across the different valuation techniques that are available. In some cases, market data can be used, while in others surveys would need to be carried out. In all cases, data are required on cause and effect relationships, and on the population affected for valuation to take place.	Either surveys of those suffering from the illness of concern or medical judgement is required to generate QALYs. Requires consideration of many of the same factors that must be taken into account in monetary valuation (age, initial state of health, *etc.*).
Resource Issues	Time	Transfer of valuations from previous studies can be carried out quickly; new valuation studies would take many months.	Use of existing QALY data can be carried out quickly as could use of medical judgement; new surveys would take many months.
	Expertise	Requires specialist economic valuation expertise and input from health experts.	Requires specialist input from health experts combined with health economists.
Overall Advantages		Allows costs and benefits to be compared in same unit of measure within a cost-benefit analysis framework.	Allows different health effects (death and other illnesses) to be assessed in same unit of measure; provides a unit of measure appropriate for use in cost-effectiveness analysis or wider multi-criteria analyses.
Overall Disadvantages		Valuation may be controversial, can require that a number of different assumptions are made in transferring values from different studies, and may not be able to cover all health effects in the same manner owing to resource/time constraints.	May require that a number of assumptions are made in transferring QALYs across specific illnesses or in deriving QALYs through the use of expert judgement. Suffers from some of same problems of valuation in that aggregate figures may 'hide' the fact that risk reduction delivers small changes in QALYs across a large population (*e.g.* for acute morbidity effects).

The derivation WTP with regard to fatality effects is based on establishing what those who could be affected by a risk would be willing to pay for small reductions in risk (or improvements in safety). Figure 3.1 sets this out graphically (from Hammitt, 2000). In this case, the most the individual would be willing to pay (Mw) for a small increase in the probability of survival or risk reduction (Mp) is the value of a statistical life (*i.e.* the slope of the indifference curve) multiplied by the change in risk (Mp).

Figure 3.1: Willingness to Pay for Reductions in Mortality Risk



These amounts are then aggregated over all affected individuals to derive a total value for the risk reduction option or safety improvement under consideration. The resultant figure indicates what the risk reduction option is worth to the affected group. In order to standardise the values that emerge, the concept of preventing a ‘statistical’ death or injury is used.

For example, assume that a group of Y individuals (where Y is a large number) is willing to pay an average of X monetary units each to reduce the probability of death of one of that group. They will altogether then be willing to pay XY monetary units to avoid one statistical death. Thus, XY monetary units provides an estimate of the ‘value of a statistical life’ (VSL). So, for example, if there are 1 million individuals at risk and the mean value expressed by respondents to reduce the risk of death by 1 in 1 million is $1, then the value of a statistical life is estimated at $1 million. These calculations reflect the amount of money that is worth spending (from a societal perspective) to achieve a reduction in risk across a whole population.

Criticisms have arisen over the use of VSLs in the context of chemical risk management. These are based on arguments that within the context of chemical risks, there is a need to adjust the approach taken to valuation in order to account for age, health state, and the latency of effects. The criticisms related to latency, in particular, have led to the concept of the ‘value of statistical life-years lost’ (VLYL) as an alternative method for expressing reductions in fatality risks. This approach allows distinctions in risk-reduction options to be based on their effects on longevity (Graham, 1995);

the VLYL represents the impact of premature death on an average individual's life span (for example, those who die of cancer at the age of 65 may lose 15 or so years of life expectancy).

Finally, in addition to willingness to pay for a reduction in risk, the concept of the willingness to accept (WTA) compensation for an increase in risk is also highly relevant in the valuation of fatality risks. Theoretically, WTA is the more correct measure of economic impact in cases where an action would result in an increase in the level of risk faced by an individual.

3.2.2 The Valuation Techniques

There are essentially three approaches for estimating individuals' WTP for a reduction in risk or their WTA compensation for an increase in risk:[36]

- by examining the actual voluntary expenditures made by households on items that reduce the risk of death from certain activities (such as purchasing air bags for cars), or by examining the costs associated with any avertive behaviour aimed at reducing risks;

- by examining the increased compensation individuals need ('other things being equal') to work in occupations where the risk of death at work is higher (an estimate of the WTA compensation); and

- through the use of experimental markets and survey techniques to directly elicit individuals' WTP for a reduction in the risk of death.

The basic principles underlying each of these approaches are discussed in more detail below.

3.2.3 Avertive Expenditure or Avertive Behaviour Approach

The Approach

The avertive expenditure or avertive behaviour approach[37] relies on data from conventional markets to derive direct estimates of individuals' willingness to pay for a reduction in risk (for a more detailed discussion see: Hanley & Spash, 1993; Freeman, 1993). The approach is based on the premise that an individual's expenditure on reducing or mitigating an environmental risk can be viewed as a surrogate demand function for the end level of risk. In other words, an individual's perception of the costs to him associated with a given risk can be assumed to be at least as high as the amount he/she is willing to pay to reduce the risk. Because the resulting estimate captures only actual expenditure, it does not include any measure of consumer surplus. The results, therefore, provide a minimum valuation of the benefits of reducing a risk.

Applying this technique in practice involves the following:

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[36] An additional method has been used in the past, referred to as the human capital approach (effectively, earnings over a lifetime). However, this approach is infrequently used as it suffers from a number a problems:

1) it ignores the intrinsic value of life;

~~2)~~1) it discriminates against those outside the labour force; and

~~3)~~1) it is not based on consumer welfare theory.

[37] This technique is also referred to in the literature as the consumer expenditure, preventative expenditure or defensive expenditure approaches, with the names used interchangeably.

1. Identification of the risk of concern and the markets where expenditure for reducing the risk would be made.

2. Estimation of the actual expenditure (related to the risk of concern) that is made by an individual in the surrogate market.

3. Identification of any secondary benefits that may arise from expenditure on the surrogate goods; if possible deduction of the value of these secondary benefits from the estimate of actual expenditure.

4. Determining the number of units of the surrogate good sold to risk averting individuals.

5. Calculating the total value of avertive expenditure where this is equal to the number of units sold multiplied by the cost per unit net of any secondary benefits.

By way of example, a consumer's voluntary expenditure on protective equipment – gloves, masks, *etc.* – prior to the use of a hazardous chemical (*e.g.* pesticides, wood treatment products) can be viewed as an indication of his/her willingness to pay to reduce the risks associated with the use of those products.

In applying this method, it may be important to make a distinction between action taken to avoid a risk and action taken to mitigate against a risk. It may be easier to determine the reduction in risk associated with actions taken to avoid risks than for actions aimed at mitigating risks (as risk reduction may only be partial). In contrast, the welfare costs of actions taken to avoid risks may be more difficult to estimate as they may not all relate to readily identifiable expenditures; mitigating behaviour, however, does usually involve expenditure making the calculation of welfare costs easier.

Key Issues

There are two main advantages to using this approach in the valuation of human health. The first is that the data may be readily available where the avertive expenditures relate to a commonly marketed good. Second, because the approach relies on the use of observable market data, the results are likely to be more widely accepted than those stemming from the use of hypothetical markets.

However, the consumer expenditure approach also has several drawbacks with regard to its application in the context of chemical risk reduction. A key problem relates to the fact that individuals' subjective views of the probability of a risk outcome occurring have been found to be highly different from scientific estimates of those probabilities. Their views are also likely to include not only the risk of death, but also the risk of illness (for example, associated with an accident or exposure which results in acute effects). As a result, it may be difficult to separate out the two impacts (or the joint products provided by the averting action), with the figures then over estimating the value of reducing the risk of a fatality. However, if the aim is to estimate willingness to pay for health gains overall, then the two impacts need not be separated. This means that care should be taken to recognise the potential for double counting; it may also limit the degree to which valuations that are not chemical-specific can be used through benefit transfer approaches.

Furthermore, it may be difficult to quantify the full costs associated with the purchase of a surrogate good where these include the time costs of purchasing, installing and maintaining the good (US EPA, 1999).

3.2.4 Wage Risk Premia

The Approach

Thaler and Rosen (1976) were the first to point out the positive and statistically significant relationship between the risk of death on a job and the wage rate for that job.[38] Since then, a large number of studies have been undertaken to model the relationship between the wage premium paid by employers to those working in higher risk occupations, where there is a risk of death. Essentially these 'hedonic' wage models[39] estimate the wage-risk tradeoffs as revealed by worker's preferences.

From a worker's perspective, a job can be viewed as a differentiated good with a bundle of characteristics such as wages, skill requirements, location, working conditions, prestige, as well as the risks of accidental injury and exposure to hazardous substances. Assuming that workers are free to move from one location to another, then hedonic wage models can be used to estimate the implicit marginal value placed on the various job characteristics. If it is further assumed that the worker is maximising utility, then the implicit marginal value placed on each characteristic reflect an individual's marginal willingness to pay for that characteristic. It therefore reflects the change in income necessary to compensate for a small change in a given characteristic, *i.e.* for a small change in risk (for further details on the underlying theory see Freeman, 1993).

The hedonic wage function underlying the use of these models is:

$$W = f(Q, X, R)$$

where W = wage rate in each occupation
Q = qualifications/skills of the worker
X = other job attributes such as location, working conditions, *etc.*
R = the level of workplace risk, *i.e.* risk of death or of injury.

Application of these models relies on specification of the different characteristics of the relevant jobs and the use of regression analysis to determine the marginal value associated with each characteristic.[40] Because jobs are rarely similar in all aspects except risk, regression analysis is required to determine the importance of different job characteristics, deriving a valuation for each characteristic, including the associated risk of fatality.

Key Issues

The wage-risk method relies on the assumption that there is enough labour mobility to permit individuals to choose their occupations so as to reflect all of their preferences, one of which is the preference for a level of risk and, thus, the level of compensation required to accept that risk. In economies suffering from long-standing structural imbalances in the labour markets, this is at best a

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38 Although such models are a formalisation of the concept of compensating wage differentials as first postulated by Adam Smith (see Freeman, 1993).

39 Where the hedonic relates to the different characteristics of a good that affect its quality, with the hedonic price being the implicit price attached to that characteristic as part of the overall price for the good.

40 The type of analysis used in these studies is related to the use of hedonic pricing techniques for the valuation of differences in the environmental attributes associated with property, for example. See Section 4.3.3 for further discussion.

questionable assumption. In addition, labour markets are often segmented on the basis of geography because of the costs of relocation and a lack of information on job opportunities. Markets can also be segmented on the basis of education and skill requirements. In both cases, this can lead to different implicit marginal valuations occurring across regions or across occupations.

Other key issues that arise with the application of this method are:

- in using these methods, it is difficult to distinguish between an individual's implied WTA for fatality risks as opposed to morbidity risks;

- there may be demographic bias, *e.g.* when choices made by young males in high-risk jobs are used to generalise for the entire population;

- the estimates of WTA will depend on workers' perceptions of the probability of death, while the studies usually adopt a statistical measure of the long-run frequency of death;

- where workers' perceptions of the probability of death are inaccurate, then estimates of the WTP for a reduction in risk will be biased,[41] and

- the probabilities for which the risks are measured are generally higher than those faced in most other situations of interest; and related to this is that high risk occupations generally involve individuals who are 'risk takers' and thus whose WTA for an increase in the risk of death is not typical of the population at large (*e.g.* steeplejacks).[42]

In addition, as discussed by Viscusi (1992), the results appear to be sensitive to the risk data used and the form of the model providing the basis for the regression analysis. Several of the more recent studies have corrected for some of the above issues [such as Gegax *et al* (1991), Marin & Psacharopoulos (1982) and Viscusi & Moore (1989) as reported in Freeman (1993)], although none has addressed all of the above issues simultaneously.

3.2.5 Hypothetical Markets - the Contingent Valuation Method

The Approach

An approach that is increasingly being used to derive the value of a statistical life is the contingent valuation method (CVM).[43] It contrasts with the two methods discussed above which rely on the use of observed data (preferences as revealed in the market place), and instead relies upon the direct questioning of individuals concerning their WTP for improvements which are contingent upon the existence of a hypothetical market for such goods. People are asked directly, through the use of a survey either:

- the maximum amount they would be willing to pay for the reduction of risks; or

41 This issue has been explored for the chemicals industry by Viscusi & O'Connor (1984), albeit with regard to injury risks and not fatality risks. The study found that workers in the industry perceived the risk of injury on the job to be 50 % higher than labour statistic estimates.

42 This is probably one reason that the estimated value of life declines as the mean risk level in a group increases. From a theoretical perspective one would expect the opposite if the populations were homogeneous.

43 For a detailed review of the use of contingent valuation methods in general, see Mitchell & Carson (1989).

– the minimum that they would be willing to accept in compensation to forgo the improvement.

In these surveys, individuals are typically provided with a detailed description of the good or service that is being valued. Within the context of the valuation of a life, this involves providing information on the nature and level of a risk that exists before a proposed risk reduction option is introduced and the nature and level of the risk after it has been reduced. In this respect, it is critical to the reliability of the results that respondents have an accurate understanding of the risks and level or risk reduction.

Details are also provided on the proposed intervention (risk reduction option) and on the method by which the respondent would pay for it, with this being referred to as the payment vehicle. The payment vehicle should be appropriate for the good or service being valued and the proposed intervention. Potential types of payment vehicles include increases in taxes, one-off payments to special funds, user fees, and increased product prices. In all cases, care should be taken to ensure that the payment vehicle is realistic and emotionally neutral.

Within the survey process itself, different procedures can be used to elicit respondents' WTP (or WTA). These include the use of open-ended questions, bidding games, and referendum questions (dichotomous choice). For example, respondents can be asked:

– to name an amount that best reflects their maximum willingness to pay, with no amounts suggested within the survey itself; or

– whether they would be willing to pay a specific amount, with this usually followed by questions as to whether they would pay higher (or lower) amounts;

– to choose the amount from a selection of figures that best reflects their willingness to pay.

These surveys can also instruct individuals to ignore certain factors, such as risks to friends and family, or the direct costs (such as medical costs or lost earnings) associated with an accident, when deciding on their WTP.

Several other questions are also asked in order to gather information across a range of respondent characteristics that may affect WTP. These include, for example, income, education, age, environmental attitudes, *etc.* Statistical analysis is then applied to the responses to estimate the mean willingness to pay across all respondents and to determine the significance of the various respondent characteristics to individual's WTP (and whether the results conform to what would be expected) and to identify any potential biases in the results.

Key Issues

The main advantage of CVM (and related techniques) is that it can be used to value any change in risk or environmental quality. Furthermore, it can be used to elicit valuations for highly specific changes, which could not be valued through the use of market data (whether for direct or surrogate markets). In addition, survey methods can provide a valuation that incorporates not only benefits to the individual him/herself but also related to bequest (*i.e.* future generations) and existence (related to knowing that others can benefit from a service) values.

The CVM method is subject to a range of criticisms within the context of the valuation of a statistical life and environmental policy making. These include the following (Ball *et al*, 1998):

- people may not have clear pre-formed preferences for non-market goods and survey responses may not, therefore, be an accurate measure of true economic preferences; in such cases, people may have to 'construct' their preferences using the information provided in the survey;

- the contingent valuation task may be too complex in that individuals are asked to make complex judgements about human costs at the same time as considering questions concerning small changes in the probability of often unfamiliar risks. It is no surprise that there have been serious difficulties in conveying information on the impact of small changes in risk and that some respondents have provided inconsistent answers to valuation questions;

- the potential for biases to arise in the survey methodology itself which affect the answers given to the valuation questions, where these can relate to the hypothetical nature of the market being proposed and the design of the survey in terms of the ordering of questions, the choice of payment vehicle and the amount of information provided; and

- an insensitivity by respondents to the good being valued; for example, in the context of safety, research has found that respondents are often insensitive to the magnitude of the reduction in risk and the severity of different injury states (Jones-Lee *et al*, 1995).

At this point in time, CVM remains a somewhat controversial approach. It is, however, subject to rapid change and development and is a preferred approach for many researchers, who argue that most of the problems raised above can be dealt with through better questionnaire design.

3.2.6 Hypothetical Markets - Attribute Based Stated Preference Methods

Attribute based stated preference methods (ABSPM) have been developed as an alternative to the use of contingent valuation methods. Instead of asking individuals about their willingness to pay for a given environmental change, they collect data from individuals on their preferences by asking them to make choices (within an experimental or hypothetical context) between bundles of attributes, which include the health or environmental good of concern. For example, in a health care context, a respondent would be presented with choices among different treatment options where each option provides different health outcomes (impact on health state, side effects, *etc.*) and cost outcomes. The choice made from these options would reveal the respondent's preferences for these outcomes (or attributes) and the implicit trade-offs between them. Similarly, a market good (*e.g.* sun tanning cream) can be broken down into its attributes (price, quality, *etc.*) including health risk reduction attributes (potential to prevent skin cancer) in order to determine a respondent's willingness to trade money for the health risk reduction attributes in the product.

These methods are based on the idea (Lancaster, 1966) that individuals receive utility from consuming the characteristics or bundles of attributes of goods. Their advantages over other valuation techniques include that they can be designed so as to eliminate some of the problems that affect the use of revealed preference or actual market data,[44] they can also be designed to include attributes or bundles of attributes that do not currently exist in the market or "real world"; and they can be combined with market data where there is a desire to ensure that the valuation exercise reflects actual choices while avoiding some of the problems that can arise in relying on such data alone.

In these methods, descriptions of the alternatives are commonly generated using experimental design techniques, often with the objective of minimising the number of combinations of the attributes that are provided to respondents to enable the statistical identification of the

[44] For example, they can be designed so as to eliminate the type of collinearity problems that can confound the use of revealed preference or market data.

underlying utility functions. It is common practise to ask respondents to consider multiple scenarios in order to make the most efficient use of survey resources. An individual's response can simply involve a choice of an attribute bundle from a set of alternative bundles (or simply "alternatives"). However, a ranking of alternatives or a selection of some quantity of specific alternatives has also been used.[45]

The seven steps outlined below typically characterise the process of developing and designing an attribute based approach to a policy problem (based on Adamowicz *et al*, 1998b).

1. **Characterisation of the decision problem**: This involves the identification of the problem at hand (choice of policy options defined by different levels of environmental risk, choice from alternative products that are defined by characteristics including health considerations, *etc.*).

2. **Attribute level selection**: The number of attributes and value of the levels for each attribute is defined in this stage, as appropriate for the decision problem at hand.

3. **Experimental design development:** Once attributes and levels have been determined, experimental design procedures are used to construct the choice tasks, alternatives or profiles that will be presented to the respondents.

4. **Questionnaire development**: The questionnaire can vary from paper and pencil tasks to computer-aided surveys. As in any survey-based research, pre-testing of the questionnaire is a necessary component of the research programme.

5. **Sample sizing and data collection**: The usual considerations of desired accuracy levels versus data collection costs must guide definition of sample sizes.

6. **Model estimation**: The most common approach has been the use of Mulitnomial Logit (MNL), and the most common estimation method has been maximum likelihood, although the most appropriate method will depend on the issues being examined (see also the discussion on random utility methods provided in Section 4.2.5).

7. **Policy analysis and decision support system development**: Most ABSPM applications are targeted to generating welfare measures, or predictions of behaviour (probability of choice of an alternative from the choice set), or both. Thus, the models are used to simulate outcomes that can be used in policy analysis or as components of decision support tools.

The model that underpins ABSPM is based on random utility theory. Under this theory, individuals hold unobservable utilities for different choice alternatives, with these utilities being comprised of both a systematic component and a random or unexplainable component. The systematic portion includes those attributes of choice alternatives that can be identified and measured, and that explain differences in individuals' choices.

These fundamental relationships are typically expressed as:

[45] Since the method focuses on attributes, it is often suggested that the choice experiment method is nothing more than conjoint analysis. However, conjoint analysis relies on the use of rating scales to identify intensity of preference for a single attribute bundle (*e.g.* rating likelihood to buy on a scale from 1 to 7 representing very unlikely to very likely). The rating is not elicited in a choice context, and thus has only weak links at best to utility or demand theory (Adamowicz *et al*, 1998b; Freeman, 1991; Luce, 2000; Adamowicz *et al*, 1998b). Thus, this type of contingent ranking is not generally consistent with economic theory and cannot be used to assess economic values for policy analysis.

$$U_{in} = V_{in} + \varepsilon_{in}, \qquad (1)$$

Where U_{in} = the utility that individual *n* associates with choice option *i*,
V_{in} = the systematic component of utility that individual *n* associates with , and
ε_{in} = the random component associated with *n* and *i*.

The presence of the random component implies that utilities are stochastic when viewed by a researcher. Consequently, researchers can predict only the probability that *n* will choose *i*, and not the exact option that will be chosen. The systematic component of utility is typically specified as $V_{in} = X_{in}\exists$, where the X_{in} contains the attributes for alternative i (and in some cases information about the individual n) and ∃ is a vector of parameters (at times referred to as 'part-worths').

One can specify the probability that *n* chooses *i* from a set of choice alternatives (C_n) as follows:

$$P(i|C_n) = P[(V_{in} + \varepsilon_{in}) > (V_{in} + \varepsilon_{in})], \text{ for all } j \text{ alternatives in } C_{n.} \qquad (2)$$

In other words, (2) states that the probability that *n* chooses *i* from the choice set C_n is equal to the probability that the systematic and random components of *i* for individual *n* are larger than the systematic and random components of all other options that compete with alternative *i*. Based on the above, probabilistic relationships can be developed to describe the probability that individuals will choose a given option in response to changes in attributes and/or the factors reflecting differences in individual decision makers.[46]

Welfare measures for ABSPM (and Random Utility models in general) are discussed in Small & Rosen (1981) and Hanemann (1982). Since the approach provides estimates of the parameters of a (conditional indirect) utility function, the welfare measures are relatively easily constructed. The parameter estimates for the attributes can be used to create marginal values. Assuming a linear relationship, for example, the ratio of the parameter from a single attribute over the parameter on a cost or price attribute provides the marginal value of that attribute. Measures based on expected utilities provide the basis for estimating welfare changes when the value of changes to a single alternative from a set of many is being considered (see Hanemann (1982) and Freeman (1993) for further discussion).

Two options for choice experiment approaches for valuing health and safety risks are a "state-of-the-world" approach and a behavioural approach. In the state-of-the-world approach, respondents are presented with alternative states with some of these states reflecting lower health risks or better health states, while other alternatives reflect poorer conditions. A price or cost is included in the description of alternatives. This is essentially the approach of Johnson *et al* (1998) who provide respondents with a description of an initial condition and two alternative health states in which the attributes are symptoms, duration of symptoms, daily activities (reduced or not) and costs to the household associated with each state. Respondents are asked to choose from the alternatives, thereby reflecting the trade-offs between health outcomes and money.

The behavioural approach attempts to capture behavioural responses to risks (perceived risks) and/or health outcomes by examining trade-offs for goods or services with related health effects. For example, the benefits of air quality improvements can be examined using a choice experiment that asks respondents to choose between alternative rental accommodation/apartments

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46 The most common choice model used to provide the basis for the econometric analysis of the choices is the conditional logit model, which requires the stochastic term to be distributed extreme value - see McFadden, 1974 for the basis of this model.

where the apartment conditions vary by rent, air quality, and other factors. The ability to experimentally design the attributes of the alternatives is a key element in this type of analysis. The experimental design allows one to separate out the effects of the attributes, for example, potentially separating visibility from health impacts. Actual observed choices of apartment cannot provide such information since the actual market situation typically suffers from collinearity and from a limited range of attributes. Similar examinations can be performed using any consumer product (sunscreen, herbal remedies, automobiles and safety features) to identify the respondent's willingness to trade-off health and safety risks and money. For further reading on the use of ABSPM in assessing health risk trade-offs see Viscusi *et al* (1991), Johnson *et al* (1998), Johnson & Desvousges (1997), Desvousges *et al* (1997).

There is increasing use of choice experiments in the health economics literature as a guide to health care service provision (see Ryan & Farrar, 2000). In these cases respondents are presented with choices of alternative treatment options (and at times the cost of these options reflected in various ways including time, travel cost *etc.*). The choice between treatment options reflects the trade-offs between attributes of the treatments (discomfort, side effects, *etc.*) and monetary factors. This approach also provides an alternative to QALY studies that rely on ratings of health outcomes.

Key Issues

The key issues with regard to the application of this method in the context of chemical risk management relate to the econometric modelling that is used and to the design of the ABSPM experiment.

With regard to econometrics, the issues relate to:

- the functional form assumed for the utility function, with a linear relationship generally assumed even though this may not accurately reflect the underlying preference structure (see Adamowicz *et al* (1998a) for a comparison of different functional forms);

- the level of serial correlation that may arise from the fact that respondents are asked to complete a series of choice tasks (*e.g.* 8 or 16 different tasks), with this aspect not being taken into account in some of the econometric models used in data analysis (Revelt & Train (1998) identify methods to correct for serial correlation, while Ouwersloot & Rietveld (1996) provide a comparison of panel and non-panel estimation approaches);

- the inclusion of individual specific variables which are generally not incorporated directly in the utility function because these characteristics are invariant among choices. However, heterogeneity in preferences can be accounted for through a range of approaches if it is important to classify individuals into different groups in order to estimate the utility function specific to these classes (*e.g.* because choice patterns are expected to vary across classes) For further reading see Revelt & Train (1998) and Boxall & Adamowicz (2000); and

- whether or not to combine revealed and stated preference data to estimate a joint model, to test the degree to which the different methods lead to convergent estimates. Louviere, Hensher & Swait (2000) show how this can be done to enable identification of the relative error variance in the two sets of data.

There are also a number of design issues which require consideration, with these being related to: the choice context, the presentation of attributes and alternatives, the experimental design, and the hypothetical nature of the choices.

A key element in constructing an ABSPM experiment is to design the choice scenario as closely as possible to the individual's actual choice context (thereby avoiding some of the criticisms often made of survey based techniques and in particular contingent valuation methods). This choice context must also be able to provide the information necessary for the policy question. If the policy question is to assess the value of improved health condition, then the instrument must be designed within the context of an individual choosing among alternative health options and their implications for health condition and other factors. If the policy issue is a more general question of eliciting the value of health risk reduction through public policy (*e.g.* more stringent air quality regulation) then the choice context may be defined as a referendum or as some other form of social choice mechanism. The individual would then be presented with alternative policy options (*e.g.* alternative regulatory levels with the resulting health risk levels and associated costs presented as attributes) within a referendum or social choice context. Attempting to mimic the choice context requires that care is taken in framing the choice question (referendum, market product choice, *etc*.), and in the selection and presentation of the attributes (descriptions, number of attributes and levels, *etc.*), and the determination and presentation of the alternative bundles of these (generic alternatives, "labelled" alternatives, inclusion of a status quo alternative, determination of the number of alternatives, *etc.*).

Some form of experimental design usually provides the basis for the combinations of attribute levels presented to respondents in ABSPM. In particular, by ensuring that attributes are not correlated to one another, a better understanding is developed on the impact that each attribute has on utility. However, when using experimental designs care is required to ensure that respondents are not being asked to make too many choice decisions, or that those they are being asked to make have not become over-simplified. For a discussion on the types of design procedures typically used and an overview of the issues that should be considered in experimental design, see Louviere, Hensher & Swait (2000).

Finally, it should be noted that ABSPM are still hypothetical methods. That is, these methods generate responses to questions about what the respondent would choose, and not information on what actual choices have been made. While there is still considerable scepticism regarding the ability of hypothetical methods to generate models that are consistent with actual choice behaviour, there is mounting evidence that well designed stated preference (SP) methods, and in particular ABSPM, can perform at least as well as models based on actual choices (see List, 2000; Haener *et al*, 2001). There is the added value that they are flexible enough in the design to include new goods or policies and clearly identify attribute effects.

3.2.7 Issues in the Valuation of a Statistical Life

The main issues that arise in estimation of the value of a statistical life (VSL) are:

- the transfer of risk estimates from different probability ranges;
- the decision context and characteristics of the risk;
- the treatment of acute versus chronic mortality;
- the treatment of age dependent mortality and whether an approach based on VSL or on the value of life years lost is more appropriate; and
- the conflict of current methods with altruistic and egalitarian views.

The Transfer of Estimates from Different Probability Ranges

Because of a lack of resources or of the time required to undertake a valuation study specific to a particular chemical risk management issue, analysts may seek to transfer estimates of VSL developed for one policy issue to other policy issues.

Table 3.3 below provides a summary of central estimates of VSL (and range if available) for European studies using market-based approaches, wage risk studies and contingent valuation methods (from the ExternE project - EC, 1995). All of these values have been converted into 1990 prices, with the resulting range for the VSL of between £0.5 and £4.3 million. The mean of the range is approximately £3.5 million. It is worth noting that, on average, the highest values come from the CVM studies and the lowest from the consumer market studies (averting behaviour studies), reflecting actual expenditures.

By comparison, Table 3.4 gives a summary of results from US studies. The range for these studies is $0.7 - $16.3 million (£0.4 - £10.2 million). These results suggest that, as a result of the higher per capita income in the US and other cultural factors, WTP is higher in the US compared with that found for the European countries where the studies were carried out. It also results from the higher values found in the US labour market (wage price) studies, which produce the higher end of the valuations. Within the European studies, the earlier CVM studies undertaken by Jones-Lee study (1976) and Frankel (1979) are the only ones that approach similar valuations to the upper tier of US studies.

The US EPA (EPA, 1999) recommends that a figure of $5.8 million (representing the central estimate of the figures presented in Table 3.4) be used in EPA related policy analyses. In general, the use of a mean value should be treated with care, as the mean may not be the best or most appropriate for any given context.

For example, one of the key concerns in undertaking such transfers relates to the probability range over which estimation of the VSL was originally carried out and over that to which it is to be transferred. Typically one is dealing with much lower probabilities of death from most chemical risk issues (of the order of 10^{-6} and lower), than those for the issues for which the estimated value of a statistical life has been derived.

Traditional approaches to deriving willingness to pay estimates generally assume a 'linearity' between risk and the payment. For example, if a risk of death of one in one million is valued at $1 million, then based on an assumption of linearity it is assumed that a risk of death of 1/1000 should then be valued at $1 million/1000, or $1000 using the VSL approach. Some researchers argue that although this may not be a bad assumption within a small range of the risk of death for which the VSL was established, it is more questionable for risk levels that are very different from the one used in obtaining the original estimate (Department of Health, 1999).

Table 3.3: European Empirical Estimates of the Value of a Statistical Life (VSL - £ 1990)

Country	Author	Year	VSL (million)
Market Based Studies			
UK	Melinek	73	£0.2 - 0.5
UK	Ghosh	75	£0.5
UK	Jones-Lee	76	£0.6 - 6.6
UK	Blomquist	79	£0.6 - 2.1
Average UK Market			£0.5 - 2.4
Wage-Risk Studies			
UK	Melinek	73	£0.5
UK	Veljanovski	78	£5.0 - 7.0
UK	Needleman	80	£0.2
UK	Marin *et al*	82	£2.2 - 2.5
Average wage-risk			£2.0 - 2.6
CVM Studies (including contingent ranking method)			
UK	Melinek	73	£0.3
UK	Jones-Lee	76	£9.2 - 11.4
UK	Maclean	79	£3.1
UK	Frankel	79	£3.1 - 12.5
UK	Jones-Lee *et al*	85	£0.8 - 3.2
Sweden	Persson	89	£1.6 - 1.9
Austria	Maier *et al*	89	£1.9
Average CVM			£2.8 - 4.9
Source: See EC (1995) for complete references			

Table 3.4: US Estimates of the Value of a Statistical Life (mean values in US$ 1997)		
Study	**Method**	**VSL (million)**
Kneisner & Leeth (1991 – US)	Labor Market	$0.7
Smith &Gilbert (1984)	Labor Market	$0.8
Dillingham (1985)	Labor Market	$1.1
Butler (1983)	Labor Market	$1.3
Miller & Guria (1991)	Contingent Valuation	$1.5
Moore & Viscusi (1988)	Labor Market	$3.0
Viscusi, Magat & Huber (1991)	Contingent Valuation	$3.3
Marin & Psacharopoulos (1982)	Labor Market	$3.4
Gegax *et al* (1985)	Contingent Valuation	$4.0
Kneisner & Leeth (1991 – Australia)	Labor Market	$4.0
Gerking, de Haan & Schulze (1988)	Contingent Valuation	$4.1
Cousineau, Lecroix & Girard (1988)	Labor Market	$4.4
Jones-Lee (1989)	Contingent Valuation	$4.6
Dillingham (1985)	Labor Market	$4.7
Viscusi (1978, 1979)	Labor Market	$5.0
R. S. Smith (1976)	Labor Market	$5.6
V. K. Smith (1976)	Labor Market	$5.7
Olson (1981)	Labor Market	$6.3
Viscusi (1981)	Labor Market	$7.9
R. S. Smith (1974)	Labor Market	$8.7
Moore & Viscusi (1988)	Labor Market	$8.8
Kneisner & Leeth (1991 – Japan)	Labor Market	$9.2
Herzog & Schlottman (1987)	Labor Market	$11.0
Leigh & Folson (1984)	Labor Market	$11.7
Leigh (1987)	Labor Market	$12.6
Garen (1988)	Labor Market	$16.3
Source: US EPA (1999): derived from EPA (1997): see Viscusi (1992) for complete references for each study.		

Hammitt (2000) does not support this view, concluding instead that respondents to contingent valuation surveys are often incapable of understanding very small changes in risk, such as a move to 1 in 10,000 to 2 in 10,000. This finding is based on the proportionality (or rather the lack of it) of a range of VSL estimates, *i.e.* the assumption that WTP should vary in (near) proportion to the change in risk. It is concluded that many VSL estimates should be treated with caution, indeed "…investigators need to develop methods of conducting CV studies that yield demonstrably valid results…VSL estimates…that do not demonstrate near proportionality between estimated WTP and risk reduction implied by theory must be viewed with some scepticism…".

Risk Context and Characteristics

As indicated by Tables 3.3 and 3.4, the literature reveals substantial cross-study variation in estimates of VSL, regardless of the valuation technique adopted. These variations are in part attributable to inconsistencies and differences in survey design, methodological approach, data analysis, and survey population. Kidholm (1992) also notes that differences arise between countries as the general perception of safety differs. It can further be argued that such variations should be expected given that the decision context and the characteristics of the risks being addressed also vary.

Table 3.5 presents a number of factors that may influence WTP depending on the type of risk. As can be seen from this table, they include context specific factors as well as factors relating to scale, age and temporal variations in the risk activity. Several researchers have argued that conventional WTP studies have often omitted crucial information on the risk characteristics (other than the size of the risk reduction and the initial risk level) which may influence the value that people are willing to pay for risk reduction (see for example Slovic, 1987). Research combining both psychological and economic approaches has been undertaken recently which indicates that peoples' WTP may vary according to a number of social and psychological factors. The argument here is that variations in estimates should be expected as they reflect real variations in preferences for risk reduction according to the context and characteristics of the risk considered.

Table 3.5: Factors Influencing WTP

Factor	**Example**
Type of health effect (acute, chronic, latent)	People may dread a lingering death more than a sudden death.
Factors relating to risk context (voluntariness, control, responsibility, familiarity, *etc.*)	People seem to regard involuntary risks over which they have no control, risks which are someone else's responsibility and novel risks, as worse than others.
Futurity of health effect and discounting	Effects that happen sooner are expected to be regarded as worse than those which happen later.
Age	People may attach particular value to life and health at certain ages.
Remaining life expectancy	WTP is expected to be positively related to the number of years of life expectancy at risk.
Attitudes to risk	Risk aversion is expected to affect willingness to trade wealth for risk; younger people may be less averse to risk.
State of health-related quality of life	People are expected to be keener to extend life in good health than life in poor health.
Level of exposure to risk	People may be keener to reduce high risk by a set amount than a low risk by the same amount.
Wealth/income/socio-economic status	People with more wealth are likely to have a higher WTP to reduce a given risk than those with less, and there may be other differences between social groups.
Source: UK Department of Health, 1999	

Only limited empirical research has been undertaken to address the question of how significant such contextual effects are on estimates of WTP and the findings of this work are varied. The conclusions are, however, mixed. For example, Mendeloff & Kaplan (1989) used risk ranking techniques to assess the relative values placed on preventing different types of deaths (as opposed to the derivation of VSL estimates) by spreading resources across eight different programmes which varied in terms of the risk context, the age of those at risk, and whether the deaths prevented would be immediate or in the future. The results indicate that while there may be fairly large differences in individual preferences for specific programmes, when aggregated across individuals these differences may partially 'balance out', leaving more modest overall differences in preferences.

Similarly, Horowitz (1994) found that consumers had distinct and consistent preferences for regulation of pesticide residues when compared with automobile exhaust controls, if both options cost the same and saved the same number of lives. As soon as the number of lives saved varied over the two options, the contextual effect was diminished and the stronger preference was for the programme that saved the most lives. Similarly, work by Savage (1991) has confirmed that certain hazards, such as nuclear facilities, result in both a heightened psychological fear and a higher WTP to reduce risk; however, for most everyday risks no such systematic relationship was found to exist.

Research by McDaniels *et al* (1992) also examined the relationship between risk perceptions by examining responses to ten hazards which represented 'well-defined' familiar risks (automobiles, flying, power tools, liquefied natural gas, and a workplace chemical - vinyl chloride) and 'less well-defined' risks characterised by greater uncertainty in exposure and effects (chlorinated water, hazardous waste, nuclear energy, sulphur air pollution and electromagnetic fields). An analysis which looked at the two groups of hazards separately showed a sharp contrast between them, with WTP to reduce well-defined risks being most affected by perceived levels of exposure and WTP to reduce the less well-defined risks being most influenced by the characteristics of dread and severity.

The immediacy of death may also be an important factor. Some hazards may lead to prompt fatalities while for other hazards, like exposure to low concentrations of toxic chemicals, the effects of exposure may be delayed to far into the future, or death may follow only after many years of continued exposure. Whether deaths prevented now should be valued more than deaths prevented in the future is a contentious issue, reflected in the ongoing debate over discounting and the use of VSL or an alternative measure referred to as the value of life years lost (VLYL), discussed in Section 3.2.9.

There is some evidence to suggest that individuals treat voluntary risk differently from involuntary risk. Because people are more accepting of voluntary risks, the level of compensation required in order to accept a voluntary risk is much lower than that required to accept an involuntary risk. Starr (1976) has estimated, on a judgmental basis, the difference between the level of compensation (in terms of WTA) required for an individual to accept a voluntary increase in risk and an involuntary increase. He finds the latter to be around ten times as high as the former for probabilities of death between 10^{-6}-10^{-7}. Unfortunately, for low probability risks, such as those associated with the types of the impacts arising from air pollution and many other chemicals in the environment, estimates of the differences in WTA are not available.

Whether a premium should be added for catastrophic or multiple fatality risks is a further important issue for some regulators. Is the loss of fifty lives in one accident more important than the loss of fifty lives in separate accidents? Many of those who specialise in the risks of major industrial hazards believe that a premium should be associated with the prevention of multiple-fatality events. Research by Slovic *et al* (1984) and Jones-Lee & Loomes (1994), however, suggests that there is little public support for this view. Instead the preference of most surveyed is for minimising the number of lives lost overall, rather than for reducing the risk of catastrophic accidents in particular.

What are the implications of the above for policy analysts? The key conclusion is that benefit transfer must be undertaken with care. Justification should be provided for the values adopted, with factors that may affect the reliability of transfer values spelt out. This should include an indication of the degree of uncertainty that may be introduced by any significant differences between the original policy context and the transfer policy context.

The Treatment of Age Dependent Fatality, Ill Health and Latency Effects

Following on from the above, it has been suggested that adjustments be made to VSL values to account for some of differences in context and risk characteristics [see for example UK Department of Health (1999) and US EPA Science Advisory Board (2000)]. In addition to adjustments related to the risk context and income (addressed above), are questions as to whether:

- VSL values should be adjusted for the fact that many of those affected are old?
- should some adjustment be made for their state of health? and

– should some adjustment be made for a lapse of time between the exposure and the impact?

It is interesting to note that recent work undertaken for the UK Department of Health on the ‘Economic Appraisal of the Health Effects of Air Pollution’ (Department of Health, 1999) set out a series of proposed adjustment factors for deriving a relative value for fatalities associated with air pollution using a road accident VSL figure. The adjustments relate to differences in the risk context, income, age and health state of the affected population, the level or risk and other costs not captured by the roads willingness to pay estimates.

For further details on why such adjustments may be appropriate, reference should be made to Department of Health (1999) and EPA (2000). The issue of age has arisen because some of the studies, and much of the clinical evidence, suggest that pollution disproportionately affects the elderly [see for example Schwartz & Dockery (1992) and Department of Health (1999)].

The literature on age and VSL also points to a relationship that is non-linear. The VSL increases with age in the early years and then declines, with a peak value at 40-50 years of age; this phenomenon is referred to as the ‘inverted-U life-cycle, which has both a theoretical and empirical basis (Jones-Lee, 1989; Maddison, 1997). A number of factors have been identified as leading to this effect. One is that income increases with age up to a certain point and declines thereafter. Hence, if VSL is related to income, this would explain some of the inverted ‘U’ shape to be explained by that factor. Another is that one can view the age effects as the sum of two opposing forces: a true age effect in which people become more risk averse as they get older; and a ‘life expectancy effect’ in which they hold a lower VSL as they get older and have fewer years to live. In such cases, the ‘age factor’ does not result in a declining VSL with age *per se*.

Taken together, the above findings suggest that the baseline (average across the population) VSL value should be adjusted for age. However, as concluded by the EPA’s Science Advisory Board on Economics, there is still insufficient data available to indicate what the actual adjustment should be for chemicals policies, with further research required for credible adjustment factors to be developed (Science Advisory Board, 2000).

Impact of Health Impairment

Apart from age, one might also expect VSL to vary with the state of health for two reasons. The first relates to the issue of pure health impairment, while the second is the effect of a shortening of life span. If a person’s quality of life is poor, it may affect his or her WTP for a reduction in the risk of death. There is little evidence, however, that points to this, although health service professionals do use a ‘Quality Adjusted Life Years’ (or QALY) approach in which resources are allocated on the basis of paying no more than a certain amount for a QALY (see Section 3.4 below).

Adoption of a baseline VSL representing an average across the entire population provides no adjustment for pure health impairment, nor does it include any adjustment for reductions in life expectancy. For chemical risk management issues, this may be particularly important. For example, clinical experience suggests that the loss of life expectancy for those who die from exposure to air pollutants is generally very short, perhaps only a few months.

Several observers agree that it is inappropriate to take a value for VSL based on a population with normal life expectancy and apply it to a population with a very much shortened life expectancy. One way to approach this is to value life years directly, and this is discussed further below. Another is to separate out the VSL into one component for life expectancy and one for age. In other words, people may be willing to pay more to reduce the risk of death as they get older but they are willing to pay less as their life expectancy declines. The two are, of course, not inconsistent.

Although some work has been undertaken into this issue (see for example Markandya, 1997), again there is insufficient empirical research upon which to base any health related adjustment.

Impacts of Latency on VSL

If exposure to air pollution today causes the risk of death to increase T years from now, the WTP to avoid that risk is not likely to be the same as that associated with an increase in the risk of death now. The most common approach to dealing with this latency of effects is to discount future risks. If the WTP for an immediate reduction in risk is X money units, then the WTP for a reduction in a risk with a latency of T years is X multiplied by $(1+r)^{-T}$. The key question, of course, is what value should r (the discount rate) take?

Cases have been made for relatively high discount rates (*i.e.* above 10%), as well as for low rates (*i.e.* 3% or lower). There is, however, a lack of agreement amongst economists as to which is the appropriate rate. For a further discussion of issues related to discounting, see Section 2 of Part 4.

Egalitarianism and Altruism

Finally, there are arguments against the use of different VSLs based on the egalitarian view that all lives are equal. Under this rationale, the value of preventing the death of an elderly person should be equal to that of preventing the death of a young person. Similarly, some agree that the value of preventing a death now should be equal to the value of a preventing a death in the future.

On the same basis, it could be further argued that differences in the characteristics of the risk should not be used as a basis for establishing differing VSLs; if more money is spent to prevent certain types of deaths in preference to other types, then the number of deaths avoided is not maximised.

Furthermore, some studies carried out in the past have involved the valuation of a statistical life across different countries and using country specific VSL figures. Because all of the valuation approaches reflect either individuals' income constraints or other economic constraints (*e.g.* on wages), estimates of VSL for poorer countries are significantly lower than those for wealthier countries. For example, a contingent valuation study carried out by the Korean Environmental Research Institute (2000) found a mean VSL of $0.47 million for reducing a future risk and $0.87 for reducing a current risk, as compared to the US EPA recommended VSL figure of $5.8 million. This difference will remain even when the resulting figures are adjusted for the purchasing power parity of a unit of money. The implication of these findings is that this type of approach may result in international policies delivering higher levels of risk reduction and public protection in wealthy countries than in the poorer countries.

With regard to altruism,[47] there is some evidence that people hold a WTP to reduce the risks incurred by others and that the values may be significant (Jones-Lee *et al*, 1985; Viscusi *et al*, 1988). Such studies suggest that VSL studies be inflated to account for such altruism. However, others have argued that undertaking such adjustments may lead to double-counting (Viscusi, 1992; US EPA, 1999).

3.2.8 Valuation of Life Years Lost

As discussed above, concern has been expressed over the use of a VSL based on accident-related or 'wage risk differential' studies when the aim is to value the fatality-related effects of

[47] The corollary to non-use or passive use values and the valuation of environmental quality changes.

chemicals in the environment. Such studies derive willingness to pay estimates for a reduction in the risk of death for individuals where death would imply a loss of life expectancy of 40 years or thereabouts. Within a chemical risk context and across chemicals, the expected loss of life expectancy for those who die may vary greatly from this (*e.g.* for air pollution, it has been suggested that the loss of life expectancy may be a matter of only months (Miller & Hurley, 1998; Brunekreef, 1997). As the above discussion indicates, it may not be appropriate to automatically assume that the same value should apply in both cases.

This has led researchers to suggest an alternative approach based on the concept of the 'value of life years lost' (VLYL). Using this approach, the aim is to value changes in the risk of death in terms of the value that would be assigned to each of the future life years that would be lost as a result of premature death.

There are two aspects to this approach: one is conceptual and the other is practical. The main conceptual argument is that premature death matters because life is shortened and that the degree to which life is shortened is material to an individual's valuation. The theoretical models that underlie the derivation of the WTP for a change in the risk of death are sensitive to the survival probabilities that the individual faces at the time the valuation is made. Hence, *a priori* one would expect an empirical estimation of WTP also to be sensitive to the amount by which life is shortened.

The practical question is how should one derive an estimate of this WTP. Ideally, studies deriving WTP for people having different survival probabilities would be carried out and, from these, the degree to which WTP changes with life expectancy would be determined. Unfortunately, this would be difficult to do in practice. As a result, as a first approximation, the value of a life year lost is calculated as the constant sum which, taken over an average remaining life span allowing for survival probabilities, has a discounted present value equal to a pre-specified VSL. It therefore represents a distribution of the VSL over time. Thus, VLYL in conceptual terms is given by the following expression (NERA & CASPAR, 1997):

$$\mathrm{VSL}_{\alpha} = \mathrm{VLYL}\sum_{i=a}^{i=T} \mathrm{P}_i \left(1+r\right)^{-i}$$

where a is the age of the person whose VSL has been estimated, P_i is the conditional probability of survival in year i, having survived to year $i\text{-}1$. T is the upper age bound and r is the discount rate. The above formula assumes that VLYL is independent of age. This assumption will not in general be valid, but has been made as a simplifying one to allow derivation of an initial estimate for the kind of changes in survival probabilities expected to be found as part of chemical risk management.

For example, one can estimate the VLYL using the above equation based on the VSL of £2 million noted above, and estimates of survival probabilities for the EU population as available from Eurostat. If discount rates of 0%, 3% and 10% are applied then the resulting estimates of VLYL range from £56,000 to £215,000. In order to estimate a WTP for a given change in survival probabilities, the estimate of VSL associated with those survival probabilities are used and the VLYL is recomputed for each type of event that is being valued.

The assumption that a VLYL is independent of age underlying this simple model is not defensible, however, and is the main criticism levied against the use of VLYL (see for example NERA & CASPAR, 1997). However, it is possible to take age into account by adopting VSL values that have been adjusted for age or through research aimed at directly eliciting such values. The latter approach was adopted in a study by Johannesson & Johansson (1996). They used the contingent valuation method to look at the WTP of different respondents, aged 18-69 for a device that will increase life expectancy from 10 years to 11 years at the age of 75. The precise question posed to people by this survey was:

> "The chance of a man/woman of your age reaching 75 years is x percent. On average a 75 years old person lives for another ten years. Assume that if you survive to the age of 75 years you are given the possibility to undergo medical treatment. The treatment is expected to increase your expected remaining length of life to 11 years. Would you buy this treatment if it costs C crowns and has to be paid for this year?"

Although this study can be criticised for the fact that it did not communicate a health risks to individuals (in that an additional year of life is certain) and that the average respondent would have to wait 44 years for the life-year extension, it is also considered as one method that may be promising in the future (Department of Health, 1999).

The second issue raised by critics of the VLYL approach is that it is invalid to add up life years - *i.e.* saving one life year for two people is not equal to saving two life years for one person. It is correct to say that the adding-up of life years cannot be carried out in a simplistic fashion. The value of future years is less than that of the present year's on account of discounting. With regard to the adding-up of values across individuals, there are problems associated with such a procedure but they are no more or less than those associated with the adding-up of VSL values across individuals.

In general, it is not possible to derive a VLYL value that is independent of age and that declines at a constant discount rate over the remaining lifetime (as is possible for VSL).[48] As noted above, it is possible to allow for different VLYLs for people of different ages. The use of a constant discount rate for 'adding-up' the WTP for a change in the present risk of death, however, is a simplification, and requires checking as part of future research. As long as individuals have a coherent notion of life years and a WTP for life years, it should be possible to make empirical estimates of these categories.

3.2.9 Current Practice and the Application of VSL and VLYL

The limitations of the current practice on valuing fatality risks are twofold. The first relates to the reliability of the procedure used to elicit people's preferences. This reflects the concerns noted above that survey methods are not sufficiently sophisticated to extract well-considered preferences, and that representative members of the general public are unlikely to be sufficiently well informed about risk to be able to fully comprehend the commodity that they are being asked to value. This, therefore, casts doubt on the reliability of a VSL or VLYL estimated using survey instruments that do not inform respondents about the nature of risk, and check that they have understood the concepts before using their responses.

The second limitation of current practice relates to the arguments concerning the need to take into account the risk context and the characteristics of those at risk. The main issue here is that the value of life estimated in the context of reducing the current risk of accidental death should not be used to derive values for reducing other types of risk. These other risks include the chronic and latent effects of exposure to air pollution and other hazardous substances, and the risk of death from such exposures where these affect mainly people whose health is already compromised, such as the elderly. With few exceptions, past studies concerning fatality risks have focused on the risk of accidental

[48] The normal assumption is that VLYLs decline over time at a constant discount rate and it is this assumption that is challenged.

death.[49] These studies concern risks for which the average loss of life expectancy is about 40 years. Because the acute fatality risks associated with some chemicals, such as those associated with air pollution, fall disproportionately on the elderly and people already in ill health, the expected loss of life is very much lower. One therefore can justly question whether figures derived in the context of the risk of accidental death can be used to value the risk of mortality associated with chemical risks.

The recently published report by the UK Department of Health (1999) notes, for example, that currently (within the UK) "…there are no empirical studies of willingness to pay for reductions in air pollution mortality risks…". These refer both to acute mortality risks, and the mortality risks from latent and chronic effects, both of which are relevant to air pollution and other chemical risks. The report suggests as an interim solution adjusting available estimates for the value of accidental mortality risk using the type of adjustment framework for age and latency impacts outlined above. However, making such adjustments and ensuring that they are robust is problematic, and it is believed that more reliable estimates would be derived by seeking values based on the context of the risk in question than by adjusting values for other types of risk.

The above also underlines one of the report's conclusions that VLYL-based approach may be appropriate for deriving values for reduced risk of latent, chronic and future mortality when it is appropriate to the risk context. In this regard, although the use of VLYL may be appropriate when the risk of concern relates to the loss of a few life years for those affected, it may not be appropriate when the risk relates more to the loss of several years (for example from the onset of cancer).

The choice between use of VSL or VLYL may, of course, be constrained by regulatory or other policy requirements which effectively require that a certain approach is adopted in policy appraisals.

3.3 Morbidity Effects

3.3.1 General Approach

As for mortality effects, morbidity effects can be assessed using non-monetary indicators of benefit or through the application of monetary valuation techniques. In contrast to the valuation of fatalities, monetary valuation of morbidity effects may be more readily achieved as they relate to actual expenditures; it may also be more defensible and acceptable to the range of stakeholders who will be affected by the end risk management decision.

Morbidity effects may vary from illnesses which last for only a short period (less than a day) to non-fatal chronic effects. They may or may not result in hospital admissions and may or may not have significant impacts on an individual's quality of life and range of activity. Thus, when assessing the benefits of reduced morbidity risks, it may be important to take such factors into account in addition to any information on the number of reduced cases for any particular end-point. Accounting for such differences may be important in considering the trade-offs between the benefits of reduced morbidity effects and the costs of risk reduction options. It may also be important with regard to any trade-offs related to the types of effects that may arise (*i.e.* where risk reduction results in a change in the nature of the risk - for example, from chronic effects to acute effects through a change in processes or chemicals used).

The full economic cost of an illness is composed of the following parts:

[49] There are of course studies that have looked specifically at chemical risk issues, such as the one by Krupnick et al (2000) which estimates WTP values for reductions in air pollution (PM 10) related mortality risks.

(a) the costs of any expenditures on averting and/or mitigating the effects of the illness (where this includes both private - individual/family and insurance - and public expenditures);

(b) the value of any time lost because of the illness; and

(c) the value of lost utility because of restricted activities, pain and suffering, anxiety about the future and concern and inconvenience to family members and others.

The first category includes both expenditures on prophylactics (preventative treatments), as well as on the treatment of the illness once it has occurred and on-going care requirements.

To value these components, researchers have estimated the resource costs associated with illness (or the 'costs of illness'), using CVM methods as well as models of avertive behaviour (with the techniques applied similar to those described in Section 3.2 above). Valuation itself is complicated by the fact that there is a plethora of health states and a range of different means of developing them. Thus, in order to undertake a valuation exercise, information is required from risk assessments setting out the relationship between the use of a particular chemical, a given health state and the costs of that health state. Useful general references on the valuation of morbidity effects are given by Krupnick & Cropper (1989), Tolley *et al* (1994) and Johansson (1995).

However, it should be recognised that many object to the valuation of health effects – including morbidity – with it being seen as inappropriate at best, and unethical or offensive at worst (see Frederick & Fischoff, 1998). Such views have led in the past to the use of CEA over CBA, or for the use of Quality Adjusted Life Years (QALYs) as a non-monetary measure of benefit (see Section 3.4).

3.3.2 Resource Costs Based Estimates

The Approach

There are two components to the resource costs of an illness. The first is the actual costs of illness (COI), which are the easiest to measure. Estimation of these costs is based either on the actual expenditures associated with different illnesses, or on the expected frequency of the use of different services for different illnesses. Part of these costs may be incurred by the individual directly and others through private insurance or through general taxation.

The second component of resource costs is that of lost earnings and/or time. The costs of lost earnings are typically valued at the after-tax wage rate (for the work time lost), and at the opportunity cost of leisure (for the leisure time lost). Typically the latter is between one half and one third of the after-tax wage. Complications arise when a worker can work but is not performing at his/her full capacity. In such cases, an estimate of the productivity loss should be included in the valuation of lost time.

Total resource costs are then estimated as the sum of:

1) actual expenditures (*e.g.* medicines, doctor and hospital bills) per day, and
2) the value of lost earnings and leisure time per day,
3) multiplied by the number of days sick and number of cases of sickness for the illness.

Key Issues

Because the above approach focuses only on resource costs, it does not necessarily reflect an individual's full WTP to avoid an illness (Freeman, 1993). This may be the case for four reasons: First, when applying this approach, expenditure on avoiding health effects is often not included in the calculations; where a risk reduction option would reduce or eliminate the need for such expenditure, failure to account for such expenditure will result in an underestimate of benefits. Second, a person's medical costs may be paid for through general taxation or through medical insurance schemes. In addition, employers bear some part of the costs of lost working time (with these ultimately borne by the consumer (through increased product prices). Third, a focus only on resource costs fails to account for any losses in utility associated with suffering and pain. Fourth, in the simpler models no distinction is made between the relative severity (and associated implications for WTP) of an episode of illness lasting several days and one lasting only one day.

A final issue concerns the ability to collect data on the actual costs associated with a particular health endpoint given the accounting practices generally adopted by health services. Costs will generally be allocated to fairly high level headings of activity, with each of these categories covering a wide range of more specific treatments or health interventions. Data specific to the types of effects likely to be of concern for most chemical risk assessments may not, therefore, be available.

3.3.3 Avertive Expenditure Approach

The Approach

The avertive expenditure approach is related to the use of COI estimates, but in this case it is aimed at estimating only the costs associated with consumers' expenditure on avoiding different health states.[50] As in the estimation of VSLs, it involves the estimation of a 'health production function', from which one estimates the inputs used by the individual in different health states and, taking the difference in value between these, obtains the cost of moving from one health state to another. To develop such a function, the following types of data are required:

- frequency, duration and severity of symptoms;
- exposure levels;
- actions taken to avoid ill-health effects;
- the costs of such actions; and
- any other variables affecting health status (*e.g.* age, income, overall health state, persistence of ill health, and so on).

The basic health production function can be written as (after Harrington & Portney, 1987):

$$s = s(d, b)$$
$$d = d(c, a)$$

by substitution : $s = s(c, a, b)$

with: $ds/dc > 0$

$ds/db,\ ds/da < 0$

[50] It therefore ignores the opportunity costs of time, covering only part of the full economic costs.

where s = days sick
d = realised exposure to pollution
c = pollution
b = medical treatments and mitigating activities
a = averting and avoidance activities

Using data on the above for a cross-section of individuals allows estimation of health production and input demand functions, which can in turn be used to calculate marginal willingness to pay for a unit reduction in pollution (and hence health benefit).

Key Issues

The difficulty in applying this approach is in estimating the production function, as many averting 'inputs' may provide more than one service (*e.g.* bottled water, air conditioners, organic produce); in addition, changes in consumption as a function of the state of illness are difficult to estimate.

The second issue is similar to one discussed above with regard to estimation of VSLs through this approach: the approach assumes that individuals correctly perceive both the level of risk they face and the actual benefits that they will gain through the avertive expenditure. This assumption is questionable in practice.

At present, there are few estimates of morbidity health endpoints based on such models. Cropper & Freeman (1991) report on research that examined individuals' expenditure on help with various household tasks in order to avoid angina pain. However, they note that the study suffers from many of the problems noted above and in Section 3.2, with the issue of 'joint products' being of key concern to the estimated values.

3.3.4 Hypothetical Markets

The Approach

As for the derivation of VSLs, the use of direct survey approaches (such as CVM and other stated preference methods) provide the main alternative to the resource cost based approaches discussed above (see Sections 3.2.5 and 3.2.6). Based on an assumed health production function, surveys are used to ask respondents how much they are willing to pay to either avoid certain symptoms or to reduce pollution. One of the earlier studies was that undertaken by Loehman *et al* (1979) which used a simple payment card format to elicit from respondents their maximum WTP for the reduction of symptom days. Since this study, several others have followed with the aim of valuing a range of symptoms, such as those by Krupnick & Cropper (1992), and Viscusi *et al* (1991), Tolley *et al* (1986), Chestnut & Rowe (1986).

By way of example, in the studies conducted by Viscusi *et al* (1991) and Krupnick & Cropper (1992) surveys were used to estimate individuals' WTP for reducing the risks of developing chronic respiratory disease (adopting definitions reflecting severe cases of such disease). In both of these studies, respondents were presented with trade-offs concerning the risks of developing chronic bronchitis or respiratory disease in general and the choice of hypothetical locations to live, which varied across the costs of living. Additional questions were asked about the risks of developing chronic bronchitis and immediate accidental death. For the study undertaken by Viscusi *et al* (1991), an implicit WTP per statistical case of chronic respiratory disease avoided was estimated at about

$570,000 for the median value and $1.1 million for the mean value ($1994). In this case, the authors caution against the use of the mean because it is affected by a small number of high valuations.

Chestnut & Rowe (1986) used survey methods to obtain the WTP of asthmatics to prevent an increase in 'bad asthma days'. In this study, each respondent defined for him/herself what a bad asthma day related to on a severity scale of 1 to 7 for different asthma symptoms. The study found that WTP responses were positively associated with the number of annual attacks experienced by an individual and how an individual defined a bad asthma day. For example, when a bad day was defined as a day with any symptoms then WTP was $13 per day ($1994), while it rose to $58 per day ($1994) when it was defined as a day with more than moderate symptoms (the central figure was $36 per day).

Tolley *et al* (1986) and Loehman *et al* (1979) used survey methods to elicit WTP to avoid a day with a single minor respiratory symptom such as head congestion or coughing. Combined, these two studies suggest median values ranging from $6 to $17 ($1994) and far higher mean values - for example $70 to avoid a day of severe shortness of breath.[51]

A more recent example of the use of hypothetical markets is given by Navrud (1997). Over 1,000 Norwegians were questioned on their WTP to avoid a symptom-day involving coughing, sneezing, headaches, and so on. A summary of some of the results is provided in Table 3.6.

Table 3.6: Examples of Morbidity-Related WTP from Navrud (1997)

Symptom	**WTP (£1996)**
Coughing	£8
Headache	£14
Acute bronchitis	£16
Shortness of breath	£21
Asthma attacks (non-asthmatics)	£45
Asthma attacks (asthmatics)	£93

Using the above format of results, the value per symptom day is then multiplied by the predicted change in the number of symptom days (based on dose-response data) to provide an estimate of total benefits.

As part of these surveys, data are also usually collected on the variables which may affect an individual's willingness to pay, such as age, health status, income, education, attitudes towards the environment, *etc.* These data are important to understanding how WTP varies with the characteristics of the target population, and are essential if the results are to be used in different contexts from the original study purpose.

Key Issues

The advantage of using survey techniques is that the valuation exercise can be tailored to the chemical risk issue of concern. Thus, the hypothetical market used to elicit the WTP values can be directly related to the costs of risk reduction and to the benefits of this in terms of the specific reduction in the risks of morbidity effects. So, for example, individuals can be asked their WTP to reduce the risk of developing a particular illness at some point in the future.

[51] The median values may be more appropriate indicators of WTP as the mean values were not adjusted for potentially inaccurate high values of WTP for a small number of respondents.

However, the same disadvantages that arise with the valuation of a statistical life arise here. For example, similar issues arise concerning an individual's ability to understand and hence value small changes in the probability of a risk occurring. With regard to chronic effects, individuals may also find it difficult to value effects that will not arise until far into the future.

3.3.5 Issues in the Valuation of Morbidity Effects

Most of the issues that arise in the valuation of fatality effects also arise with the valuation of morbidity. Thus, the discussion presented in Section 3.2 on the following issues in particular is also relevant here:

- the transfer of estimates derived for different probability ranges;
- the decision contexts and the characteristics of the risk; and
- the relationship between acute and chronic effects.

Key issues associated with each of the valuation approaches were discussed above. As for VSLs, research has also been undertaken into the potential relationship between the results generated through the different approaches.

One line of research relates to the fact that the derivation of resource cost estimates can, in some cases, be achieved using existing data. Because cost of illness figures do not reflect an individual's full WTP to avoid a given health end-point, the development of adjustment factors representing the relationship between these two measures would allow a better estimate of the total costs of illness to be calculated.

As noted above, cost of illness figures reflect actual expenditures and may or may not reflect some measure of consumer surplus and, hence, the additional amount an individual would be willing to pay to avoid an illness. To arrive at the total cost of an illness, therefore, one should use estimated WTP, plus the part of the cost of illness that is not reflected in WTP. This will be the component that is paid for through taxation and, possibly, through insurance.

As a result, one cannot simply add COI and WTP together to arrive at the total cost. In part, this relationship has been studied, by making a direct comparison of the two estimates and looking at their ratio. Rowe *et al* (1995) have done this for US data and find that the ratio of WTP to COI is in the range 1.3 to 2.4. On the basis of their analysis, they recommend a value of 2 for adverse health effects other than cancer and a value of 1.5 for non-fatal cancers. These researchers argue that although the relationship between COI and WTP is complex, it offers one method of arriving at an indicative cost figure for morbidity endpoints, many of which have not been subject to WTP studies.

3.4 Non-Monetary Assessment Methods

3.4.1 Introduction

In public health and medicine, health and mortality effects are often measured using some form of health adjusted life year (HALY). The most common of these measures is the "Quality Adjusted Life Year" (QALY). More recently introduced alternatives include the "Disability Adjusted Life Year" (DALY) and "Healthy Years Equivalent" (HYE). Each of these concepts can be used to measure the utility of a specified "health profile" *(i.e.* a time path of health states ending in death) in terms of an equally valuable length of time lived in full health. QALYs are a measure of an individual's preferences for his/her own health and longevity that can be added across people to measure the social value of health improvements.

3.4.2 Quality Adjusted Life Years

The number of QALYs in a specified health profile is calculated as the quality-weighted lifespan,

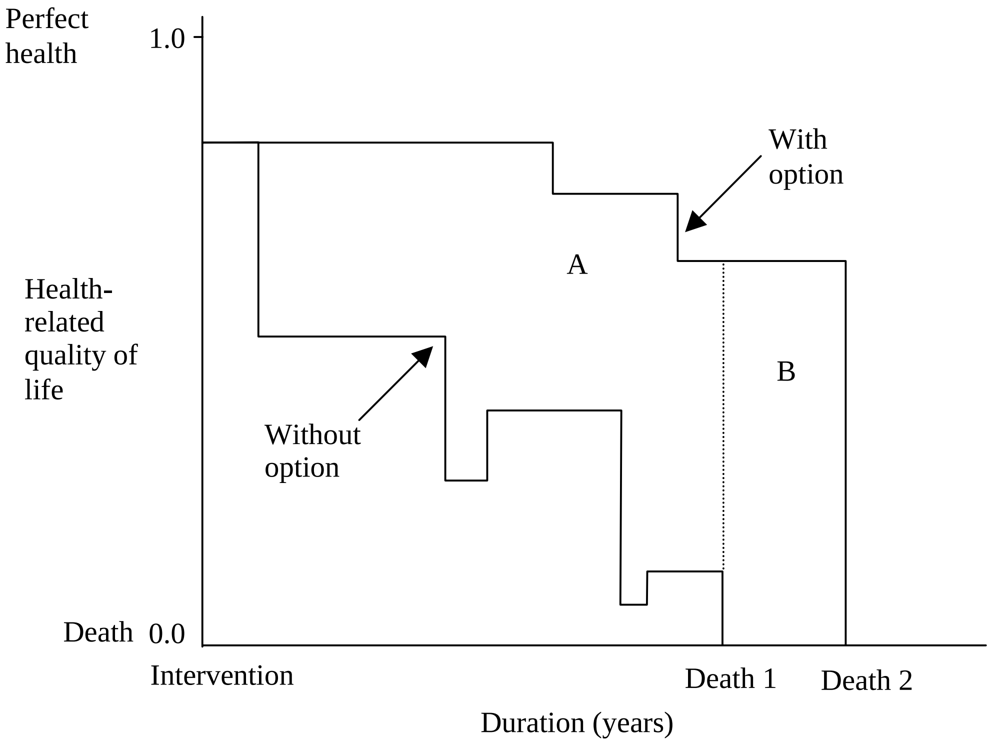
$$QALYs = \sum_{i=1}^{M} q_i T_i \ , \qquad (1)$$

where lifespan is divided into a number of periods corresponding to health states. In equation (1), there are M periods that are indexed by i. The duration of each period is given by T_i, and the "health-related quality of life" (HRQL) associated with that period is characterised by a weight q_i. The value of an intervention that affects health and/or longevity is measured as the difference in QALYs between the two health profiles, as illustrated in Figure 3.2 below.

The HRQL is a number that represents the quality of health. It is scaled so that a value of one corresponds to perfect or excellent health, and a value of zero corresponds to health that is equivalent to death (*i.e.* an individual would not care if he/she were to live in this health state or die). Typically, q is between one and zero, but values of q less than zero can be used to represent states of health perceived worse than death.

Uncertainty about future health and longevity can be represented as a lottery or gamble across alternative health profiles, in other words a set of possible future health profiles with associated probabilities. In evaluating a one-time risk like a single airplane trip, for example, it might be sufficient to consider a lottery over two health profiles. One profile corresponds to the case in which the airplane crashes, in which case the health profile terminates at the individual's current age. The second profile corresponds to the case in which the airplane does not crash, in which case the individual might survive to his age-specific life expectancy, experiencing a series of health states that are typical for each age. If uncertainty about future health were important, this second outcome could be represented as a set of several possible health profiles (*e.g.* living to age 65 with cardiovascular disease, living to age 70 with cardiovascular disease, living to age 70 with excellent health, *etc.*) with associated probabilities of following each health profile.

Figure 3.2: The QALY Concept



A measure such as QALYs can be justified as an individual utility function for health if it represents an individual's preferences between alternative health profiles and between lotteries (gambles) on health profiles. That is, if whenever an individual prefers one health profile to another, the preferred profile offers a larger number of QALYs. Similarly, if the individual prefers one lottery over health profiles to another lottery, the preferred lottery offers the larger expected number of QALYs.

The conditions under which QALYs represent a valid individual utility function were identified by Pliskin, Shepard & Weinstein (1980). These authors restricted their attention to the special case of chronic (stable) health states, in which case equation (1) simplifies to:

$$QALYs = qT \qquad (2)$$

where T is remaining lifespan and q is the HRQL for the constant health state in which the individual will live until death. In this case, QALYs represent a valid utility function for an individual's preferences if his or her preferences satisfy the following conditions:[52]

[52] Bleichrodt, Wakker & Johannesson (1997) subsequently proposed a somewhat different set of conditions such that QALYs represent a valid utility function for the special case of chronic health conditions and risk neutrality on lifespan.

1. ***Mutual utility independence***: This condition has two parts: (a) preferences between lotteries on different health states, holding duration of life constant, do not depend on remaining lifespan; and (b) preferences between lotteries on lifespan, holding health state constant, do not depend on health state. An example of part (a) is that, if an individual is indifferent between living 40 years in "good" health[53] and a 70-30 lottery between living 40 years in "excellent" health or in "fair" health, he/she is also indifferent between living 25 years in "good" health and a 70-30 lottery between living 25 years in "excellent" or "fair" health. An example of part (b) is that, if an individual is indifferent between living 30 years and a 50-50 lottery between living 40 years and 25 years, with all years lived in "excellent" health, then he/she is also indifferent between living 30 years and a 50-50 lottery between living 40 years and 25 years, with all years lived in "fair" health. Mutual utility independence is necessary for the utility to be represented as a product of separate health and longevity terms (*e.g.* q and T).

2. ***Constant proportional trade-off of longevity for health***: The fraction of remaining lifespan the individual would be willing to sacrifice to improve his/her health from one state to another does not depend on the value of his/her remaining lifespan. For example, if an individual is indifferent between living 40 years in "fair" health and 30 years in "excellent" health, he/she is also indifferent between living 24 years in "fair" health and 18 years in "excellent" health. This condition implies that the HRQL associated with a health state does not depend on the length of time spent in that state.

3. ***Risk neutrality over lifespan***: Holding health state constant, the individual prefers whichever lottery on longevity provides the greatest life expectancy. For example, the individual would prefer to live 40 years for sure to a 50-50 lottery between 42 and 36 years, and he/she would prefer that lottery to living 38 years for sure (where all years are lived in the same health state, *e.g.* "excellent" health). A risk-adjusted form of QALY (which does not require risk neutrality) has also been developed, but it is rarely used in practice (Pliskin, Shepard & Weinstein, 1980).

For the more general case in which health state can vary over lifespan (equation 1), an additional condition is required:

4. ***Additive independence across periods***: The individual's preferences for lotteries on health in any subset of the periods do not depend on health in the other periods. For example, if the individual is indifferent between (a) spending 10 years in "good" health and (b) spending 5 years in "good" health followed by a 70-30 lottery between 5 years in "excellent" health and 5 years in "poor" health, then he/she is also indifferent between (c) spending 5 years in "excellent" health followed by 5 years in "good" health and (d) spending 5 years in "excellent" health followed by a 70-30 lottery between 5 years in "excellent" health and 5 years in "poor" health.[54] Additive independence also implies that the individual is indifferent between health profiles offering the same total time spent in each health state, regardless of the ordering of the health states. This implies that QALYs can be calculated as the sum of HRQL-weighted time spent in each health state.

Empirical research suggests that, while people's preferences for health and longevity may not satisfy these conditions exactly, the conditions provide a reasonable starting point for representing preferences (Dolan, 2000; Gold *et al*, 1996).

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53 As described below, health states are typically described in much greater detail. Simple descriptions such as "excellent, "good," and "fair" are used here for illustration.

54 The alternatives (c) and (d) can be obtained from the pair (a) and (b) by changing the health state for the first 5 years from "good" to "excellent."

3.4.3 Methods for Estimating Health-Related Quality of Life

There are two types of methods used for estimating the HRQL associated with a particular health state: direct elicitation and generic health utility scales. These are described in turn.

Direct Elicitation

The HRQL may be elicited directly, using any of several question formats: time trade-off, standard gamble, visual analog scale, and patient trade-off. In general, HRQL for a health state is elicited assuming the health state will be chronic (constant).

- The ***time trade-off*** (TTO) format requires the respondent to indicate the number of years in perfect health (with $q = 1$) he/she considers to be indifferent to a specified health profile. For example, if the respondent indicates that he/she is indifferent between living 20 more years in a particular impaired health state, and 15 more years in perfect health, the value of q for the impaired health state is calculated as 15/20 = 0.75. Both health profiles offer 15 QALYs.

- The ***standard gamble*** (SG) format requires the respondent to indicate the smallest chance of survival in perfect health he/she would accept in a lottery where the alternative outcome is immediate death. This may be motivated by considering a surgery that would alleviate a health impairment without affecting longevity, except for the chance of dying in surgery. For example, if the respondent is indifferent between living 20 more years in a particular impaired health state and a lottery which offers him a 3/4 chance of living 20 more years in perfect health and a complementary chance of immediate death, the value of q for the impaired health state is again 3/4, and both the certain health profile and the lottery offer an expected value of 15 QALYs.

- The ***visual analog scale*** (VAS) is a linear scale with one end representing perfect health and the other representing health states as bad as death. The scale may include interval marks, and is sometimes designed to look like a thermometer (and called a “feeling thermometer”). The respondent is asked to place a mark on the scale representing how good or bad the specified health state is to him/her, relative to the endpoints. A similar verbal format may be used where the respondent is asked to report a number between zero and 100, where zero represents states as bad as death and 100 represents perfect health.

- The ***person trade-off*** (PTO) format asks the respondent to consider the relative value of improving health for people in different health states. For example, he or she might be asked to judge the relative value of extending longevity for people in different health states, *e.g.* if one were to choose between extending the life of 1,000 healthy people for a year and extending the life of x blind people for a year, for what value of x would he or she be indifferent? The HRQL of living with blindness is estimated as 1,000/x. Alternatively, the respondent might be asked to judge the relative value of improving health for people in one state and extending life for people in another state, *e.g.* if one were to choose between extending the life of 1,000 healthy people for a year and restoring the sight of z blind people for a year, for what value of z would he or she be indifferent? In this case, the HRQL of living with blindness is estimated as 1 (1,000/z) (Murray & Acharya, 1997).

In principle, both TTO and SG formats should yield exactly the same value, if the conditions under which QALYs provide a valid utility function are satisfied. In practice, the results may differ because the formats make different aspects of the health profiles more salient: SG emphasises risk and uncertainty, while TTO emphasises relative preferences for near-term and future health. SG values may be slightly larger than TTO values (Torrance, 1986). VAS values, because they are not tied to an explicit decision, have a weaker theoretical justification. In practice, however, they may be more reliably assessed (*i.e.*, vary less on repeat occasions) than TTO or SG values. VAS values are typically smaller than TTO or SG values, but are sometimes adjusted using an empirically estimated formula to approximate the results of TTO or SG formats. The PTO is conceptually different because it focuses on questions of distributional equity rather than on an individual's preferences for his/her own health.

An important question in eliciting HRQL is whose values to elicit? Possible sets of respondents include the general public, individuals experiencing the health states of interest, and health-care providers or others having knowledge about the health state. Experience suggests that individuals in an impaired health state assign a larger HRQL to that state than do healthier individuals. Whether this reflects improved understanding of the condition by people experiencing it or adaptation to adverse circumstances is not clear. All the elicitation methods require comparing two health states, at least one which is hypothetical to the respondent at the time of elicitation.

Generic Health Utility Scales

A number of generic health utility scales have been developed in recent years. These scales can be used to describe health states in terms of their levels on several attributes, and the HRQL associated with the state may be calculated from a numerical formula. In principle, all such scales are examples of multi-attribute utility functions, although the extent to which the scales are based on multi-attribute utility theory varies. The scales have been calibrated by fitting them to HRQL values elicited using one or more of the direct methods reviewed above.

Among the more popular generic health utility scales are the Health Utilities Index (Feeney, Torrance & Furlong, 1996), the EuroQol EQ-5D (Kind, 1996), and the Quality of Well-Being Index (Kaplan & Anderson, 1996).

The current version of the Health Utilities Index Mark III (HUI3) classifies health states using a system of eight attributes: vision, hearing, speech, ambulation, dexterity, emotion, cognition, and pain. Each attribute has either five or six levels, yielding a total of 972,000 possible health states. The attributes are designed to be structurally independent so that all of the possible combinations are at least logically possible. The attributes are approximately mutually utility independent and the multi-attribute utility function defining HRQL is a product of the factors corresponding to each attribute level. The function has been calibrated to HRQL values elicited using SG and VAS format questions from a general population of about 500 Canadians (Furlong *et al*, 1998).

The HUI formula and attribute levels are reported in Table 3.7. As an example, an individual whose functioning at the highest level on all attributes except vision (level 3: able to read ordinary newsprint with or without glasses but unable to recognise a friend on the other side of the street, even with glasses) and ambulation (level 4: able to walk only short distances with walking equipment, and requires a wheelchair to get around the neighbourhood) would have HRQL = 1.371 x (0.89 x 1 x 1 x 0.73 x 1 x 1 x 1 x 1) - 0.371 = 0.52.

Table 3.7: Health Utilities Index Mark 3								
Level	**Vision**	**Hearing**	**Speech**	**Ambu-lation**	**Dexterity**	**Emotion**	**Cogni-tion**	**Pain**
1	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00
2	0.98	0.95	0.94	0.93	0.95	0.95	0.92	0.96
3	0.89	0.89	0.89	0.86	0.88	0.85	0.95	0.90
4	0.84	0.80	0.81	0.73	0.76	0.64	0.83	0.77
5	0.75	0.74	0.68	0.65	0.65	0.46	0.60	0.55
6	0.61	0.61	na	0.58	0.56	na	0.42	Na
HRQL = 1.371 ($b_1 * b_2 * b_3 * b_4 * b_5 * b_6 * b_7 * b_8$) - 0.371 Note: na = not applicable (attribute has only five levels)								
Source: Furlong *et al* (1998)								

The EuroQoL EQ-5D classifies health states using a system of five attributes: mobility, self-care, usual activity, pain/discomfort, and anxiety/depression. Each attribute has three levels, yielding a total of 243 health states. Two additional states, dead and unconscious, have been added for a total of 245. Values for EQ-5D states have been elicited using TTO and VAS format questions in numerous European and Nordic populations. As an approximation, the HRQL can be represented as an additive function of the attribute levels, as shown in Table 3.8. An individual with impaired mobility (level 2: some problems in walking about) and some pain/discomfort (level 2: moderate pain or discomfort), with all other attributes at their highest levels, would have HRQL = 1 - 0.069 - 0.123 - 0.081 = 0.73.

Table 3.8: EuroQol EQ-5D			
		Level	
Attribute	**1**	**2**	**3**
Mobility	-0.0	-0.069	-0.314
Self-care	-0.0	-0.104	-0.214
Usual activity	-0.0	-0.036	-0.094
Pain/discomfort	-0.0	-0.123	-0.386
Anxiety/depression	-0.0	-0.071	-0.236
HRQL = 1 – sum of relevant item weights. The additional constant 0.081 is subtracted if any attribute is level 2, and 0.269 is subtracted if any attribute is level 3.			
Source: McDowell & Newell (1996).			

The Quality of Well-Being Index (QWB; Kaplan & Anderson, 1996) is one of the earliest of the generic health utility scales. It describes health using three attributes - mobility, physical activity, and social activity - plus an attribute consisting of descriptions of symptoms or “problem complexes.” Like the EQ-5D, the HRQL is an additive function of the attribute levels. The attribute levels and values are reported in Table 3.9. In addition, there are 27 different symptoms or complexes with values ranging from zero (no symptoms) to -0.727 (death). An individual with mobility 2 (in hospital, health related), physical activity 4 (no limitations for health reasons), social activity 3 (limited in major (primary) role activity, health related) with symptom and problem complex 10 (general tiredness, weakness, or weight loss, value = -0.259) would have HRQL = 1 - 0.090 - 0.000 - 0.061 - 0.259 = 0.590.

Table 3.9: Quality of Well-Being Index

		Scale	
Step	**Mobility**	**Physical Activity**	**Social Activity**
5	-0.000	na	-0.000
4	-0.062	-0.000	-0.061
3	na	-0.060	-0.061
2	-0.090	na	-0.061
1	na	-0.077	-0.106

HRQL = 1 – sum of scale weights – additional term for relevant symptom and problem complex.
Note: na = not applicable (not all scales use all steps)
Source: McDowell & Newell (1996).

3.4.4 Alternatives to QALYs: Disability Adjusted Life Years, Healthy Years Equivalent

In recent years, several alternatives to QALYs have been introduced, including Disability Adjusted Life Years and Healthy Years Equivalent.

Disability Adjusted Life Years (DALYs) were developed as a measure of the health of a society (rather than an individual) and have been used to measure the burden of disease in most countries of the world (Murray, 1994; Murray & Lopez, 1994; Murray & Acharya, 1997). They are similar to QALYs except they incorporate an age-weighting factor and measure the loss of longevity and health from an idealised health profile. The age-weighting factor represents a judgment that years lived in young adulthood and middle age contribute more to a society than years lived as a child or in old age. The weighting factor equals y $e^{-\bullet y}$ where *y* is age in years and • is a parameter conventionally set equal to 0.04. For this value of •, the weighting factor is largest at age 25; it is about three-fourths as large at ages 10 and 50, and half as large at ages 6 and 67.

In contrast to QALYs, DALYs measure the loss of health and longevity from a reference health state based on the age-specific mortality rates for Japanese women (among the longest-lived populations). For comparing interventions that affect the health of a population, measuring health as deficits from a reference health profile has no effect, as the reference health profile cancels out.

Healthy Years Equivalent (HYEs, Mehrez & Gafni, 1989) may be viewed as a less restrictive form of QALYs. The HYE for a specified health profile is simply the number of years lived in perfect health that the individual judges as indifferent to a specified health profile. For constant health states, HYEs are conceptually identical to QALYs elicited using a TTO format question. HYEs do not require that preferences over health profiles satisfy any of the four assumptions that QALYs require (described above), so they are much more flexible. Concomitantly, because HYEs impose so little structure on preferences, it is necessary to elicit the HYE directly for each health profile of interest. For example, because HYEs do not require that assumption 2 (constant proportional trade-off) is satisfied, one cannot assume that the ratio of HYEs to lifespan spent in a chronic health state is independent of lifespan. The HYE framework admits the possibility that an individual may be indifferent between 40 years in “poor” health and 20 years in “excellent” health, and also indifferent between 10 years in “poor” health and 9 years in “excellent” health. Perhaps because they impose so little structure on preferences, HYEs have not been widely used in practice.

3.4.5 General Assessment

An important reason for the popularity of QALYs and other health adjusted life year measures is that they do not require one to assign a monetary value to health improvements. As a result, these measures do not directly address the question of whether the health benefits of an intervention exceed the resource costs. In contrast, QALYs are used for cost-effectiveness analysis in which the analyst can determine whether the intervention under consideration produces health at a high or low cost, relative to other interventions. Judgements about whether a particular cost per QALY is socially acceptable or excessive are informed by cost-effectiveness ratios of interventions that have been implemented or rejected as too costly. In public health and medicine, interventions that cost less than $50,000 – $100,000 per QALY are often regarded as desirable investments, and those that cost more per QALY gained may be considered too costly (Hammitt, 2000).

When used to measure population changes in health, QALYs are often considered more equitable than willingness to pay (WTP) measures that represent individuals’ trade-offs between own wealth and health. WTP for improvements in health and longevity is generally higher in wealthier populations. In contrast, QALYs are generally considered to be independent of wealth and income. A possible exception occurs when considering health impairments that reduce an individual’s income-producing capability. There is some debate over whether losses in income due to health impairment should be aggregated with medical treatment costs in the cost term of the cost-effectiveness ratio, or incorporated in the HRQL. An expert panel convened by the U.S. Department of Health and Human Services recommends incorporating the lost productivity in the HRQL, which suggests that the loss in HRQL for health states that lead to losses in income may be positively correlated with income (Gold *et al*, 1996).

Because QALYs treat a gain in health or longevity equally across people, they have clear implications for the relative social value of reducing mortality risk to people of differing life expectancy and with different chronic health conditions. The number of QALYs lost due to premature fatality is proportional to the individual’s life expectancy (conditional on surviving that risk) and average future HRQL. This implies it is more valuable to reduce mortality risks to people with greater life expectancy (typically, younger people) and to people whose survival is likely to be in full rather than impaired health. In contrast, individual WTP to reduce mortality risk (*i.e.* the value per statistical life or VSL) is generally not proportional to life expectancy (it may rise then fall as a function of age), and is not necessarily smaller if survival will be in impaired rather than full health (Hammitt, 2000).

In cost-effectiveness analysis, future QALYs are usually discounted at the same rate as monetary costs. This is appropriate if one wishes to maintain distributional neutrality between current and future time periods and assumes that the real cost and efficacy of an intervention will not change over time. If costs were discounted but QALYs were not, the cost-effectiveness of the intervention would improve over time, which would imply it would be better to delay indefinitely any desirable intervention (Keeler & Cretin, 1983). However, discounting QALYs conflicts with the utility-theoretic basis reviewed above (Pliskin, Shepard & Weinstein, 1980). The conflict could be alleviated if QALYs were defined using the present value of HRQL-weighted life expectancy (using the accepted discount rate), and adapting the constant proportional trade-off and risk neutrality assumptions appropriately.

3.5 Data Sources

3.5.1 Outputs of the Risk Assessment

The risk assessment will provide the starting point for the assessment of human health effects. To a large extent, the manner in which such effects can be assessed within the SEA will depend on the nature of the risk assessment outputs.

In all cases, the risk assessment should provide details of:

- the population at risk, including sensitive sub-groups;
- the health end-points of concern;
- the current levels of risk (*e.g.* predicted exposures); and
- in some cases, an indication of the level or risk reduction required for the risks to be 'acceptable' based on, for example, no observable adverse effect level.

When the output of the risk assessment draws upon this data to provide its outputs in the form of a ratio of predicted exposure to a 'no effects' level of exposure, then there will be insufficient information for changes in health risk to be valued in money terms. In other words, the effects could be described qualitatively (and potentially quantitatively) within a CBA but not valued in money terms. CEA can be applied, however, to determine the risk reduction strategy which most cost-effectively meets the 'acceptable' risk targets. It should be possible though to apply MCA-based techniques relying on more qualitative indicators of performance.

When the output of the risk assessment provides quantitative predictions of the frequency of a specified consequence occurring based on data on the potential for harm with environmental concentrations, and on exposure and population at risk data, then the options open to the analyst are greater. In these cases, the ability to draw on quantitative estimates of changes in impact should allow for the monetary valuation of health effects as part of CBA, or the use of such data as part of a CEA or MCA.

3.5.2 General Data Requirements

In taking the risk assessment outputs and then preparing an assessment, the types of data set out in Table 3.10 may be required:

Table 3.10: General Data Requirements for Human Health
Risk related data: • Populations at risk • Nature and rate of exposure • Dose-response/effects-exposure relationships
Socio-economic data for the population at risk: • Age • Health state • Income • Sex • Occupation • Education
Methodology related data: • CBA: details and results of previous valuation exercises deriving WTP estimates. Where the results of previous studies are not to be used, issue specific valuation exercises will be required each having its own data requirements; • CEA: data on the number of cases avoided or details and results of previous policy decisions providing implicit valuations of cost per life saved or cost per illness avoided; and • MCA: where QALYs are to be used, details of the number of cases avoided for linking to QALY indicators; where new systems are to be developed, data requirements will be method specific.

3.5.3 Data Sources

Potential sources of data may include:

– health statistics providing details of illness and fatality rates for different health end points;

– other statistical data, such as on medical and other treatment costs;

– environmental quality data providing details of concentrations and hence exposure at a geographic level, the academic economics literature and associated research;

– previous regulatory appraisals which have involved the valuation of health effects; and

– other research prepared by/for research institutes, industry, government and international organisations such as the OECD.

3.6 The ZOZ Case Study

3.6.1 Introduction

As part of our hypothetical ZOZ case study, potential health risks to workers have been identified from the use of ZOZ in the production of chemical intermediaries. The relevant outputs of the risk assessment are as follows:

- workers at the eight sites producing chemical intermediaries are at risk from exposure to ZOZ during the manufacturing process;

- exposure to ZOZ may lead to acute respiratory effects, such as a shortness of breath and respiratory attacks (similar to asthma attacks); while exposure to lower levels of ZOZ over the longer-term may lead to chronic respiratory effects;

- around 150 workers are involved in the production of ZOZ. Approximately 30 cases of chronic respiratory effects are predicted to occur from on-going low level exposure to ZOZ over the 15 year time period for the analysis. Roughly 15 workers per year are predicted as being exposed to levels causing acute effects. Total cases over the 15 year time period are 30 for chronic effects and 225 for acute effects.

The alternative risk reduction options were presented in Section 1 of Part 2. Of the five types of option put forward, only three would provide worker health and safety benefits. The option involving increased worker training, with or without the use of additional protective equipment, is specifically targeted at reducing worker related health risks through awareness raising and prevention and, thus, needs to be examined within this component of the SEA. Similarly, a voluntary agreement to reduce use of ZOZ could reduce risks to workers, as would marketing and use restrictions if these were levied across all uses of ZOZ including in chemical intermediaries. These latter two options would also reduce risks to consumers and the general public and to the aquatic environment. They might, however, also lead to the introduction of new risks depending on the substitute chemical adopted as a replacement for ZOZ.

Modelling work was therefore undertaken to predict the change in risk (to workers for this example) that would occur following introduction of each of the options. The results of this work are presented in Table 3.11 for the assumed 15 year period. As can be seen from the table, training on its own cannot reduce the risks to zero; even with the inclusion of sophisticated worker protection equipment, one case of chronic respiratory effects is predicted to occur every two years. Similarly, the voluntary agreement is not predicted as reducing all risks, with chronic respiratory effects still expected.

There are no strict requirements for the SEA to incorporate these effects into the analysis in any particular manner, so the analysts have decided to examine the results from applying three different approaches: a cost-effectiveness type of approach; placing monetary values on reductions in health effects; and examining the use of QALYs. The outputs of these three analyses are presented below for reduction of risks to workers. In practice, these analyses would have to be expanded to address reductions in risks to consumers and the general public from reductions in exposure to ZOZ either from consumer goods or *via* the environment.

Table 3.11: Predicted Change in Cases of Acute or Chronic Risks per Annum				
	Acute Cases per Annum		**Chronic Cases per Annum**	
Option	**Pre-Option**	**Post-Option**	**Pre-Option**	**Post-Option**
Voluntary Agreement	15	0	2	0.5
Training only	15	3	2	1
Training and equipment	15	0	2	0.5
Marketing and Use	15	0	2	0

3.6.2 Cost-Effectiveness of ZOZ Risk Reduction Options

First, it is apparent that if a target of no further cases of acute or chronic respiratory effects is set, then the two options which involve training and training plus equipment will fail to meet it. However, there may still be some value in considering the relative trade-offs in terms of the cost-effectiveness of the options.

The alternative approach to considering cost-effectiveness is to calculate the costs per case avoided. Again, this runs into some problems as two different types of effect are being reduced here, acute and chronic respiratory effects. For this analysis, the analysts have decided that the chronic effects are much more important and that they should provide the basis for determining effectiveness.

The results of the above calculations are presented in Table 3.12. Fortunately, much of the work on the cost component of the analysis has been completed. For both options cost data are available on the costs to manufacturers of chemical intermediaries and to regulators. In the case of training, these relate to running industry-wide courses and include estimates of lost production time and value. In the case of equipment, it includes purchase, installation and operating costs. Under the voluntary agreement, manufacturers are expected to mainly switch to other drop-in substitutes, although not all users would do so). For use restrictions, the costs arise mainly from the need to purchase substitutes, at a 10% per unit increase in costs (from 35,000 tonnes at $500 per tonne to $550 per tonne). Additional costs would arise in reformulation and marketing, but no other significant costs are expected to arise (*e.g.* to the downstream users of the intermediaries).

The cost figures quoted in the table are the present value costs for the 15-year time period of the analysis (they have been discounted at 5%). Note that in preparing these calculations, the analysts decided not to discount the number of cases avoided in future years even though this is recommended practice (see Section 2, Part 4 for a discussion on this issue). The difference that such discounting would make was to be explored in sensitivity analysis.

Table 3.12: Cost-effectiveness of Risk Reduction Options			
Option	**Present Value Cost ***	**Effectiveness (Chronic)**	**Cost per Chronic Case Avoided**
Voluntary Agreement	$19,800,000	75% (22.5 cases)	$880,000
Training only	$8,300,000	50% (15 cases)	$553,300
Training and equipment	$16,400,000	75% (22.5 cases)	$728,900
Ban on use	$22,300,000	100% (30 cases)	$743,300
* Discounted at 5%			

As can be seen from Table 3.12, the training option has the lowest cost per case of chronic respiratory effects avoided. However, the only option that meets the target of zero chronic cases in the future is the ban on use, which is the third most expensive option in terms of costs per case avoided. It should be remembered that these options also deliver benefits in terms of reductions in acute cases. Thus, the costs presented above over-estimate the costs that can be attributed to avoidance of chronic cases only.

3.6.3 Monetary Valuation of Health Effects

In order to undertake a monetary valuation of the acute and chronic respiratory effects, the analysts first reviewed the economics literature with regard to the valuation of morbidity effects to identify what relevant studies had been undertaken in the past. They also approached the health service to see what data may be available on the actual resource costs associated with the types of acute and chronic respiratory effects of concern.

With regard to previous valuation studies, the literature search found a range of contingent valuation studies that had valued similar types of risks but within different environmental contexts. Some related to other types of worker exposure issues, but many related to more general air pollution studies carried out on the general population. Owing to restrictions on the time available to undertake any valuation exercise, there was a question as to whether any of the existing studies could provide the basis for a benefits transfer based approach. A few of the studies were quite recent, but use of some of the others was probably more questionable given that they dated back to the early 1980s; levels of income and cultural changes may make these unreliable indicators of current WTP. Overall, however, it was decided that the more recent studies could provide indicative valuations for the WTP to avoid the two types of health effect. No adjustments to the valuations were proposed to account for changes in risk context. The only adjustment made to the valuations was to convert them to current prices (US$ 2000) so that they were on the same price basis as the estimates of cost.

With regard to the development of cost of illness estimates, the health service was able to provide an indication of the typical range in costs per respiratory hospital admission in terms of doctor's time, hospitalisation costs and medical support for similar respiratory illnesses (on a per episode basis).

As a starting point, the analysts decided to adopt the value from the study that appeared most robust for transfer purposes and to examine the impact which adopting other values would have on the performance of options later as part of a sensitivity analysis (see also the discussion on sensitivity analysis given in Part 4, Section 3). In choosing the contingent valuation WTP estimates to use, care was to taken to consider the degree to which combining such estimates with cost of illness data may lead to any double counting. The values used in the analysis are as follows:

- **Cost of illness estimates**: $4,000 to $12,000 per hospital admission, with a value of $9,000 chosen; no distinction could be made between acute and chronic effects;

- **Contingent valuation** (providing figures on WTP for the risks avoided):

  - for acute effects $45 to $95 per day, with this needing to be multiplied by the typical length of an episode (average 5 days); the value used was $275 per episode;

  - for chronic effects $56,000 to $100,000 per case, with the most robust study indicating a value of around $65,000.

The results of combining these valuations with the reduction in risks are presented in Table 3.13 for acute and chronic effects respectively. Note that the total present value figures have involved discounting the benefits over the 15-year time-horizon that has been selected for the analysis (see Section 2 of Part 4 for further discussion on discounting). By totalling the figures for acute and chronic illnesses avoided, the total value of health benefits provided by each option can be calculated. These can then be compared to the costs of the different options as presented in Table 3.12 above. In undertaking any comparison at this stage, however, it must be recognised that no consideration has yet been given to any potential risks that may arise from use of substitute chemicals, nor to the fact that the ban on use is also likely to lead to environmental benefits.

Table 3.13: Estimated Value of Acute Health Effects Avoided ($million)

Options	**Cases Avoided per Annum**		**Present Value of Acute Illnesses Avoided***	**Present Value of Chronic Illnesses Avoided***	**Total Present Value Benefits**
	Acute	**Chronic**			
Voluntary Agreement	15	1.5	10.25	8.18	18.43
Training only	12	1	8.20	5.45	13.65
Training and equipment	15	1.5	10.25	8.18	18.43
Ban on use	15	2	10.25	10.90	21.15

* Discounted at 5%

3.6.4 Cost per QALY

The final type of analysis that was undertaken was to calculate the number of QALYs that would be gained with the implementation of each option and to use these as a way of judging cost-effectiveness. Thus, this analysis is similar to that presented in sub-section 3.6.2 above, only it uses QALYs as the denominator in the cost-effectiveness ratio instead of cases of chronic illness avoided. A key advantage in this case is that with the use of QALYs, it is now also possible to take acute effects into account when considering effectiveness.

Based on discussions with health colleagues, the Quality of Well Being Index (QWB - see Table 3.9 in Section 3.4.3) was used to provide the basis for deriving the health related quality of life (HRQL). Because there is no empirical evidence on what QWB for the illnesses of concern should be, these were calculated on the basis of best judgement. This required making judgements on the impact of the illnesses on mobility, physical activity, social activity, and symptoms.

Assuming that the workers otherwise would be in good health, the QWB rating in the absence of any acute or chronic effects was set at a rating of 1. The QWB rating calculated for those suffering from either acute or chronic effects was then calculated at 0.62 and 0.47, respectively. The difference between these ratings and good health provides the measure of the deterioration in health.

However, the deterioration in health also needs to be adjusted for the duration of the effects. As indicated above, acute effects last for about five days, with this translating to about 1% of a life year, while for chronic effects, it is assumed that the average worker is 50 (based on the profile of the industry) when the effects first occur and then continue for a further 25 life-years.

Combining the number of life years affected with the change in QWB rating gives the following QALYs avoided for each acute or chronic illness avoided:

- acute illness: 0.0038 QALYs [(1.00-0.62)*0.01 (years)]; and
- chronic: 13.25 QALYs ([1.00-0.47)*25 (years)].

Table 3.14 provides the total number of QALYs gained under each of the options. It should be noted that the QALYs arising in future periods have been discounted in this case so as to be presented in units of measure that are comparable with the present value cost estimates.

Table 3.14: Cost per QALY Gained for the Risk Reduction Options

Option	Present Value Cost *	Total QALYs Gained*		Cost per QALY Gained
		Acute	Chronic	
Voluntary Agreement	$19,800,000	4	1464	$13,490
Training only	$8,300,000	3	976	$ 8,480
Training and equipment	$16,400,000	4	1464	$11,170
Ban on use	$ 22,300,000	4	1952	$11,400

* Discounted at 5% per annum over 15 years

The figures presented in Table 3.14 suggest that both of the training options are more cost-effective than the ban on use, although the difference between the training and equipment option and the ban is small. The voluntary agreement is the least cost-effective in this case, owing to the lower level of QALYs expected to be delivered. More importantly, the figures also highlight the fact that chronic effects dominate in terms of QALYs gained (with this contrasting to the monetary valuation example).

Note that if the rules which apply to health intervention (and if the cost per QALY were less than $100) were followed here, then all options would be perceived as providing good value for money.

4. ENVIRONMENTAL BENEFITS

4.1 Introduction

In theory at least, there are few fundamental differences between predicting environmental impacts and predicting human health impacts. The key differences are that while human health effects relate to only a single species, environmental positive and negative costs and positive and negative benefits are complicated by potential diversity of relevant species, the numerous levels of biological organisation, the number of interrelationships between organisms and the number of endpoints and criteria that might be relevant.

Assessing the full range of environmental effects and then determining their significance and acceptability is, therefore, far from a straightforward process. A comprehensive assessment (in the fullest sense) of the benefits of chemical risk reduction would have to consider a wide range of factors including:

- the physical and chemical properties of the chemical;
- the nature of the receiving (physical) environments;
- the behaviour of the chemical in these environments (*e.g.* binding to particles, breakdown, *etc.*);
- the concentration of the chemical and any breakdown products in the environment;
- the susceptibility of organisms in the receiving environments;
- the populations of these organisms in the receiving environments;
- how populations of organisms interact as communities and ecosystems;
- the biological effects of the chemical and its breakdown products on these organisms;
- whether synergistic/antagonistic effects with other substances might occur; and
- the predicted risks to these organisms and communities in terms of lethal and sub-lethal effects.

Drawing together these factors into an assessment of environmental benefits requires that information concerning each of the affected communities/ecosystems is combined with information concerning the fate and behaviour of the chemical substance and its breakdown products. This then enables further predictions concerning:

- the change in direct effects on susceptible species and associated changes in populations through impacts on mortality, dysfunction, reproductive effects, *etc.*;

– the indirect effects on other species (susceptible or not) through impacts on the directly affected species and associated alterations to community dynamics; and

– any changes to the physical environment caused by changes in community dynamics.

Unfortunately, setting out key factors and the steps involved in predicting environmental effects can be much easier than actually incorporating them into an assessment, as the modelling that is required is by no means an exact science. This means that comprehensive assessments of the environmental advantages and drawbacks associated with changes in chemical exposure are rarely possible. As a result, there are essentially two starting points that are adopted to prediction and subsequent assessment of impacts within SEA.

The first is to determine what risks should be reduced by comparing predicted environmental concentrations to predicted no effects concentrations for the most susceptible species within a particular environmental compartment (the EU approach). The second starting point is to predict a range of possible environmental outcomes associated with elevated concentrations for different risk end-points of concern.

Table 4.1 summarises the broad categories of environmental risks that are examined in assessments undertaken in the EU. The table also provides examples of further possible end-points related to both direct and indirect effects that might be associated with the different types of environmental risk identified through EU assessments. These are intended to be illustrative, other end-points may also be considered depending on the case.

The result is that the SEA must be based on the first of the two approaches outlined above. The SEAs carried out in these cases thus generally take the form of either a semi-quantitative CBA (where quantification is mainly focused on the costs of risk reduction and benefits are discussed in qualitative terms) or a CEA.

Where the second approach is adopted and direct and indirect effects are addressed in more detail, then more quantitative CBAs can be undertaken. This includes the potential for the monetary valuation of positive and negative benefits. Although the use of economic valuation techniques will generally be limited to those cases where the second approach relying on a fuller, more probabilistic analysis has been undertaken, CEA and MCA are not. The use of MCA could be expected to require that more data are provided than just details of predicted concentrations to no effect concentrations. However, whether an assessment is based on the use of other quantitative or qualitative appraisal methods, the range of potential risks and associated effects (or endpoints) may need to be considered even if they cannot be assessed with equal reliability in terms of the impacts of a proposed risk reduction option.

Table 4.1: Summary of Environmental Risks Considered in EU Risk Assessments		
Risk Categories	**Medium of Exposure**	**Associated and Indirect Effects Not Explicitly Identified by the Risk Assessments**
Aquatic organisms	Surface water	- Impacts on natural fisheries and associated ecosystems in terms of species mix, population numbers and support function - Impacts on commercial fisheries through loss of food sources - Impacts on recreational fisheries through loss of certain species, changes in catch rate, size of fish, *etc.*
Benthic organisms	Sediment	- Impacts on natural ecosystems in terms of species mix, population numbers and support function - Through the above may have impacts on dependent activities (commercial or recreational)
Terrestrial organisms (flora and fauna)	Soil	- Impacts on natural ecosystems in terms of species mix, population numbers and support function - Impacts on agricultural, forestry and other forms of land use - Through the above impacts on amenity or aesthetic quality of land
Fish eating predators	Fish	- Impacts on natural ecosystems in terms of species mix and populations - Impacts on commercial and recreational fisheries - Impacts on recreation value - *e.g.* bird-watching (change in number/types of species that can be supported)
Worm eating predators	Earthworms	- Impacts on natural ecosystems in terms of species mix and populations - Impacts on agricultural, forestry and other forms of land use - Impacts on recreation value of affected land areas - *e.g.* through change in number/types of species that can be supported
Atmosphere		- Impacts on natural ecosystems in terms of species mix and populations (aquatic ecosystems, forests) - Impact on agriculture (yield and quality) - Impacts on building materials (corrosion and reduced life) - Impacts on recreation - *e.g.* loss of visibility

The remainder of this section sets out how monetary and non-monetary techniques can be applied to the assessment of environmental benefits (whether positive or negative):

- application of monetary valuation techniques is discussed first, with this including a review of the techniques available and issues arising with their use;

- the use of non-monetary techniques is then discussed, again with a discussion of the types of techniques available and application issues; and

- the section ends with a discussion on potential sources of data to aid in these assessments.

Table 4.2 provides a summary of the key factors which analysts may want to consider when deciding between the use of monetary and non-monetary assessment methods.

Table 4.2: Decision Factors for Selecting the Approach Towards Environmental Impact Assessment

Decision Factors		Monetary Valuation	Multi-criteria Analysis
Type of Output		The output is a money value that reflects the economic (or social) value of the predicted environmental impacts. These may relate to changes in the market value of goods affected by an environmental change, actual expenditure on environmental improvements, revealed expenditure or direct expressions of willingness to pay for environmental changes.	Quantitative MCA-based methods have rarely been applied to chemical risk management issues to date. More qualitative approaches tend to be used where valuation is not considered appropriate or is not feasible given data availability. Some of the simpler methods may be of value in summarising information on impacts.
Acceptability		Monetary valuation of market related effects tends not to be controversial, while valuation of more ecological goods and services is complex and may be considered inappropriate. In particular, there are queries over the ability of the methods to capture the full range of ecosystem impacts that may arise from environmental stresses, such as those caused by chemicals.	Use of MCA, where this involves the use of subjective weighting systems, may be less acceptable to many than the use of CBA; questions arise as to whose weight should be used. Increasing interest in the use of these methods, however, with applications existing across a range of business and policy areas.
Process Issues		Few chemicals specific studies exist to provide readily adopted data. Several issues arise in the transfer of data from other risk contexts. New studies may be required.	Several of the methods can be used as part of an interactive approach to decision making, which could provide the basis for a consensus-based approach to the SEA.
Data Requirements		Data requirements vary considerably across the different valuation techniques that are available. In some cases, market data can be used, while in others surveys would need to be carried out. In all cases, data is required on cause and effect relationships, and the environmental stock at risk for valuation to take place.	Quantitative or qualitative data are required on expected impacts to provide the basis for the analysis, with this then combined with subjective weights as required by the specific method being applied. Will require consideration of many of the same factors that must be taken into account in monetary valuation.
Resource Issues	Time	Transfer of valuations from previous studies can be carried out quickly; new valuation studies would take many months.	The less complex methods can be applied quickly, with the more sophisticated analyses taking a period of months.
	Expertise	Requires specialist economic valuation expertise and input from environmental experts.	Requires specialist input from decision analysts combined with experts on environmental economics.
Overall Advantages		Allows costs and benefits to be compared in same unit of measure within a cost-benefit analysis framework.	Depending on approach, can treat different impacts separately or allow information across impacts to be aggregated into a single unit of measure; can be combined with cost data.
Overall Disadvantages		Valuation may be controversial, can require that a number of different assumptions are made in transferring values from different studies, and may not be able to cover all environmental and ecosystem impacts of concern.	When aggregation takes place, results are presented in a unit-less measure of impact that may have little meaning for stakeholders. Use of subjective weights is regularly raised as an issue, particularly where these are based on the view of just one set of stakeholders (*e.g.* political decision makers). Few previous applications to use as a model.

4.2 Monetary Assessment Methods

4.2.1 The Values of Concern

In deriving an economic value for environmental and other non-market goods and services, it is important to consider the total economic value (TEV) of the asset under consideration (as also discussed in Section 2 of Part 2). This is the sum of 'use' and 'non-use' (or 'passive use') values. Use values are those associated with the benefits gained from actual use of an environmental asset and may include private sector uses (such as industry, agriculture, pollution assimilation and dilution and so on), recreational usage, or educational and scientific use. A sub-set of use values are referred to as 'option' values which reflect the willingness to pay of a potential user who wishes to protect an environmental asset for use in the future, *i.e.* the individual wishes to retain the option to use the resource some time in the future.

Non-use values (or passive use) are generally considered to be of two types: 'bequest' and 'existence' values. Bequest values reflect an individual's willingness to pay to conserve or secure the future of an asset so that other generations are able to use the asset. Existence values reflect an individual's willingness to pay to preserve an environmental asset and ensure its continued existence into the future, separate from any use by him or herself or others.

A particular problem with defining 'value' in the above way is that all aspects are anthropocentric, in other words they are based on human 'values' rather than reflecting some intrinsic ecological value. Indeed, it is argued by some that the full contribution of the ecosystem cannot be captured in economic valuation. Instead, the economist's definition of what comprises total economic value captures only secondary ecological values and does not include primary values (*i.e.* the 'value' of the aggregate ecosystem). The prior existence of a healthy ecosystem is necessary before the range of use and non-use values can be utilised by humans - secondary values are therefore anthropocentric by their very nature.

4.2.2 Overview of the Valuation Techniques

As highlighted in Part 2, many of the environmental use and non-use values of concern are not traded in normal markets. Because most of these fall outside the marketplace (or are 'external' to it), they are not traded in the same way as other goods and services. As a result, their economic value cannot be imputed from market prices and some other means to estimate value is necessary.

A range of economic valuation techniques has been developed to assist in imputing the monetary value attached to environmental goods and services. These techniques attempt to derive an individual's willingness to pay (WTP) for an environmental improvement or willingness to be compensated for an environmental loss (WTA) as revealed in the marketplace through individuals' actions or as directly expressed through surveys. The general aim is to determine the trade-offs that individuals would make either directly or, as is often the case, indirectly in labour, housing and other markets.

The techniques which are most commonly used are:

– conventional market price or effect on production approaches;

– household production function approaches;

– hedonic pricing methods; and

– experimental markets.

The techniques falling under these headings are discussed in more detail below. Note that because some of the techniques for deriving monetary values for environmental impacts are the same as those available for estimating the monetary values for human health effects, there is some repetition between the discussion that follows and Section 3. It is important to recognise, however, that although the theoretical underpinnings are the same, the context within which the techniques are applied varies. Section 4.2.9 at the end of this section on monetary valuation provides some details of the types of effects to which the different techniques are most applicable.

4.2.3 Effect on Production Approaches

The Approaches

These techniques rely on the use of market prices to value the costs/benefits associated with changes in environmental quality. There are essentially two key approaches falling under this heading:

- the first is variously referred to as the **effects-on-productivity approach** or the **dose-response technique**, which determines the economic value of changes in environmental quality by estimating the market value of the impact which these have on the changes in output of an associated good. For example, changes in crop yield are linked to changes in atmospheric pollutant concentrations and deposition; and

- a second related technique is calculation of the costs of replacing or restoring an environmental asset after it has been impacted. The **replacement costs** approach does not provide an economic value, but a minimum figure indicating only the engineering and other costs of re-creation (and assumes that the economic value would be higher as the site would not be re-created if it were not ‘valued’ more than such costs).

Effects on Production

In many cases, changes in environmental quality can also have a direct effect on the ability to produce a natural resource related good or environmental service and/or on the costs involved in producing that good or service. For example, changes in water quality can affect the ability of a body of water to support a fishery. As a result, the amount of fish that can be harvested from the water body will be a function not only of the level of effort that goes into harvesting but also water quality. This relationship can be represented by a simple production function (OECD, 1992):

$$Q = f\{R, S(E)\}$$

Where Q = the quantity of the natural resource (*i.e.* fish) related good that is commercially harvested

R = the amount of effort devoted to catching fish, and

$S(E)$ = the stock of fish in the water body, with this stock depending on the quality of the water environment

Given the above relationship, when water quality changes a producer faces two possible outcomes. The first outcome relates to a change in fish stocks, while the second relates to the level of fishing effort required in order for the quantity of fish harvested to remain constant.[55] In a chemical risk context, suppose that discharges of a particular substance have been linked (through the use of

[55] Assuming a constant reproductivity of fish stocks.

dose-response relationships) to a reduction in fish stocks (or in populations healthy enough to withstand harvesting). Reducing these discharges would then lead to either an increase in fish stocks or in producers being able to reduce the level of effort put into harvesting fish. The economic benefits of reducing discharges would then be either: the value of the additional output (*i.e.* fish stocks which can be harvested assuming that the level of harvesting effort remains the same); and/or the savings in costs associated with harvesting, where this now requires less resources (whether labour or other inputs to production).

In practice then, the technique is based on determining physical changes (in productivity) that result from actions affecting the environment. Applying it involves three basic steps:

1) identification and prediction of potential direct and indirect productivity effects stemming from the proposed policy;

2) determination of the correct prices to be used to reflect the value of marginal changes in productivity; and

3) estimation of the value of changes in productivity across the stock of the affected resource.

In applying this approach, the analyst must ensure that any changes in the levels of outputs predicted stem solely from changes in environmental quality or resources, and that these are wholly attributable to the action in question. This requires separating out the effects of human effort (or other factors such as weather) from the effects on output due to changes in the environment. Isolating these effects is complicated in practice as data are generally based on total effects as reflected by harvests.

The approach is applicable to all cases where a change in environmental quality may impact on the output of, or production costs of, a good that is traded in the marketplace. Further examples of such goods then are forest products, agricultural products, and use of water as an input to production (as changes in quality may affect pre-treatment requirements). Note that the approach can only be applied to market goods and services.

Replacement Costs

The replacement costs technique is based on the principle that the costs that would be incurred to restore the environment to its original state provide an estimated of the value for the environmental good or service threatened with damage or loss. Thus, potential expenditures serve as the means of placing a value on previously unvalued goods or services. Values are based, therefore, on the true costs of replacement and not on individual preferences as measured by consumer surpluses or willingness to pay.

The valuation of an environmental asset through application of the replacement costs technique requires:

- the existence of environmental ambient standard and/or sustainability constraint;
- definition of the existing asset and requirements for its replacement;
- identification of the replacement/re-creation activity required (where this should provide the same function as the existing asset or performance level);
- development of the options for the required replacement/re-creation activity; and
- costing of alternative options.

Use of this approach is valid where it is possible to argue that the replacement/recreation would have to take place due to either a demand for the good or some other constraint (such as an environmental standard). In a chemical risk management context, the approach could be used to derive values for functions or services provided by the environment which have a market equivalent, such as the ability of wetlands to remove heavy metals from effluent discharges.

Key Issues

Both of the above approaches have advantages and disadvantages.

The effects-on-productivity technique can be an extremely useful technique with considerable applicability in the field of chemical risk management. It is straightforward to apply, assuming that necessary adjustment can be made to market prices to correct for any transfer payments (subsidies) or market distortions (trade restrictions) so that economic values are used in the analysis. Even where correcting prices is difficult, order of magnitude estimates can be derived for preliminary appraisal purposes. The resulting values may also be considered more 'reliable' than those derived through some of the other valuation techniques.

The key limitation of the effects-on-productivity approach stems from problems related to uncertainties on dose-response relationships. These relationships are rarely well established, making it difficult to determine the change in output that would result from marginal changes in levels of chemical emissions. In addition, if an environmental improvement is great enough to cause significant changes in output, then current prices may no longer hold. In these cases, one would need to have an understanding of the price elasticity of demand for the good in order to determine the price that should be assumed in the analysis.

With regard to the replacement costs technique, it is generally used for the valuation of a single attribute good or service. In applying the approach, the following considerations should hold:

– existence of standard/constraint;

– the replacement should provide the same services as the asset to be valued:

– the replacement should be the least-cost alternative; and

– there should be evidence that people would be willing to pay for the services provided by the asset if they were sold at the cost of the replacement.

Because the approach is not based on willingness to pay but on potential expenditures, it should be used cautiously with regard to the valuation of environmental goods and services. It assumes that the existing system is optimal and that if the environmental good was removed or changed then those currently benefiting from the good would replace all lost attributes. If beneficiaries were not willing to replace all aspects, then the values derived through this approach would be greater than the benefits as indicated by willingness to pay. Conversely, if beneficiaries were willing to replace all aspects, then the value derived is likely to be an underestimate of the true benefits.

There are other more practical considerations relating to the degree to which replacement or re-creation of an environmental good is possible. With regard to habitats, there is some uncertainty as to whether or not it is possible to recapture the value of the goods and services provided. The value of the site may actually lie in its location, or it may result from proximity to other similar or complementary nearby areas.

4.2.4 *The Avertive Expenditure Approach*

The Approach

The avertive expenditure (or averting behaviour) approach relies on estimation of expenditure on substitute goods. It attempts to infer individuals' willingness to pay to reduce or prevent damages by observing and placing a value on the behaviour used to avoid that damage. In the case of environmental effects, it is based on determining the amount which people are willing to spend on actions that mitigate impacts. So, for example, expenditure on bottled water or water filters can be seen as a substitute for cleaner drinking water supplies, with expenditure on these substitutes providing an indication of individuals' willingness to pay for policies aimed at improving drinking water quality through, say, reductions in pesticide concentrations in supply sources.

To determine the value placed on an environmental good, the approach uses data on actual expenditures incurred by individuals to control against environmental damages. Demand for environmental damage mitigation is viewed as a surrogate demand for environmental protection. In other words, the willingness to pay to reduce adverse environmental effects is interpreted as the value of the benefits of a certain level of environmental quality. This is based on the principle that individuals will commit their own resources only if their own subjective estimates of the benefits gained are at least as great as the costs. Observed expenditure, therefore, provides an indirect measure of the benefits as perceived by individuals. But because this method can only reveal an individual's expenditure and not his or her total willingness to pay for an environmental improvement, the measure only provides a lower limit of the benefits gained.

The approach involves the following steps:

- identifying the avertive action;
- undertaking a survey to determine the number of people taking that action and the estimated costs of the action; and
- estimating the total expenditure, where this may require use of econometric modelling where survey numbers are large.

Key Issues

Several different issues arise with application of the avertive expenditure/behaviour method (see also Sections 3.2 and 3.3 with regard to the application of this approach in a health context). The approach is straightforward and simple and the required data on the costs of mitigating expenditures may be readily available. However, in attributing the value of any avertive expenditure to an environmental characteristic, care must be taken to ensure that there are no secondary benefits associated with the expenditures (the issue of 'joint products'). Another key issue concerns the ability to decompose the reasons underlying the avertive expenditure; for example, is the aim of the purchase to reduce the risk to one or many individuals? Other problems include: individuals not understanding the level of environmental protection that they are getting for their money; the proportion of expenditure aimed at reducing impacts; and the degree to which purchase of the item is considered a 'second' best option.

4.2.5 *Simple Travel Cost Models*

The travel cost method (TCM) places a value on an environmental good, generally related to recreational activities, by using the costs of consuming the services as a proxy for price. The approach is based on the concept that people spend time and money travelling to a recreational site and that these expenditures, or costs, can be treated as revealing the demand for the site. Surveys of site visitors are undertaken to determine the demand for a site, where visit rates are a function of travel expenditure, leisure time, income, entry fees, environmental characteristics and the availability of substitute sites. Through inclusion of the latter two aspects, the method can be used to determine how changes in environmental quality would change demand and, therefore, the valuation placed on a given site.

The method assumes that recreation is a divisible good and that a set of individual demand functions can be developed for different sites. Where the method is used to examine changes in site quality, it is assumed that changes at one site will change demand for other sites; the change in demand provides the measure of benefits resulting from a change in quality.

There are two basic forms of the travel cost method, with the first relying on a 'zonal' distance relationship between visit frequencies. The second form categorises visits by 'individuals' according to whether or not they are site oriented (pure visitors); derive value from the journey to the site (meanderers) or whether they make their journey for another purpose but call in on the way (Turner *et al*, 1994).

The general procedure followed in applying the zonal form of the method can be summarised as follows:

1. The area around the site in question is divided into contours for the purpose of measuring travel costs to the site.

2. Visitors are surveyed to determine their places of origin and to gather data on journey times and distances, direct travel expenses, entry fees, on-site expenditure and socio-economic characteristics (such as income, education, *etc.*).

3. Visitation rates are calculated for visitors coming from different distances (or zones). These may be expressed either as visits made by a given individual (visits per annum) or visits from a given distance (visits per capita).

4. A demand function is developed for the site, relating visitation rates to the costs of travel. The costs of travel are assumed to form the 'entry price' for the site. Regression analysis techniques are used to determine the relationship between visitation rates and travel costs, socio-economic characteristics, *etc.* Generally, the frequency of site visits is inversely related to distances; *i.e.* the greater the travel costs in time and money, the less likely an individual is to use the site.

The functional relationship used in the regression analysis will take a form based on the following:

$$V_{ij} = f\,(P_j, D_i, C_i, t_i, h_i, Q_j, M_i)$$

Where V_{ij} = number of visits by individual i to site j
P_j = vector of entry fees to the various sites
D_i = vector of distances from home of individual i to the various sites
C_i = unit travel cost of individual i
t_i = vector of travel times to the various sites for individual i
h_i = opportunity cost of travel time for individual i
Q_j = vector of environmental quality attributes
M_i = money income of individual i

5. The results of the regression analysis provide the basis for developing a demand curve for visits to the site. Once the demand curve has been estimated, the effects on demand of raising the entry fee to the site, or of changes in quality can be determined. Through this process a second stage demand curve is developed which provides an estimate of consumer surplus.

6. Dividing estimated consumer surplus by the number of visits to the site gives a figure for average consumer surplus per visit for those surveyed. By combining this figure with estimates for the total number of people visiting a site (in a given time period) an aggregate estimate of economic value, as measured by consumer surplus, can be calculated.

The ‘individual’ form of analysis varies from the above in terms of the way in which visitors are grouped together. Instead of being grouped by distance, they are grouped by the individual category into which they fall and consumer surplus estimates are then generated for each category of users (*e.g.* dog walker, anglers, boaters, *etc.*). It is therefore possible to consider how each group values the site with respect to its own activity. The number of visits made by an individual (to a site or activity) is recorded in a random sample survey. This method does not rely on aggregating visitors and thus the sample size and number of visitors is much smaller than with the zonal method.

Use of the travel cost method is limited in its applicability to the valuation of recreation activities and changes in the quality of those activities. Its applicability within the context of chemical risk management may, therefore, be limited to some extent. However, it has been applied in the US, Canada and Europe to examine water quality related issues (Smith & Desvousges, 1985), as well as in Environment Canada’s Nature Survey, and recreational fisheries.

Key Issues

The travel cost method is generally viewed as having several advantages:

– it is based on observed behaviour and is a valuable approach at the site specific level;

– it can be applied effectively and relatively inexpensively (although the information requirements can be considerable); and

– it is a well tried technique that yields plausible results.

However, use of the TCM is also criticised on a number of grounds. First and foremost of these, and one highly relevant to use of this method in the context of chemical risks, concerns the assumptions made within the models on individuals’ ability to accurately value environmental characteristics. Within these models, the quality of a site at any given time is the combined characteristics of the site and may be measured using:

– scientific data (such as dissolved oxygen, pH, level of contaminants, and so on);

– perceived quality based on a survey of visitors; or

– a combination of the two.

However, research has shown that visitors associate the visible forms of improvement with ‘real’ improvement rather than improvements that they cannot see or easily perceive. For example, visitors might easily be able to respond to the removal of litter or sewage effluent at a site, while it would be more difficult for them to perceive improvements related to dissolved oxygen levels.

There are also difficulties with regard to choosing the functional form to be used in the regression analysis. Economic theory provides no guidance on whether the demand relationship should be linear, log-linear or take some other form, yet results may be sensitive to the form used.[56] In addition, TCM assumes that all users of a given category or from a given distance would make the same number of a visits at a given entry fee. It also assumes that people know how much enjoyment they will experience when deciding to take the trip, and that visitors know the costs of travelling to the site before setting out. Related to this is whether or not the analysis recognises that travel to the site might itself form part of the benefits and that trips may be multi-functional in nature.

4.2.6 Random Utility Model for Travel Cost Analyses

The Approach

The Random Utility Model (RUM) is used in travel cost recreation demand analyses to value characteristics of recreation sites or even entire sites. For example, the model may be used to value the recreational benefits of improving water quality on lakes and rivers in a given region or to value the benefits of improved access to a beach. The model may also be used to value the closure of a site or sites due to say an oil spill or a ban on fishing. The strength of the RUM over conventional travel cost models is its ability to account for substitutes in the demand model and hence in valuation. For the original formulation of the RUM see McFadden (1978). For examples of applications in recreation see Bockstael, Hanemann & Kling (1987), Parsons & Kealy (1992), or Montgomery & Needelman (1997).

The travel cost RUM considers an individual's discrete choice of one recreation site from a set of many possible sites. The choice of site is assumed to depend on the characteristics of the sites and to reveal preferences for those characteristics. In other words, how individuals implicitly trade off one site characteristic for another in their choices. Since trip cost is always included as one of the characteristics, the model implicitly captures trade-offs between money and other site characteristics. This revealed trade-off with money makes the economic valuation possible.

The trade-offs are easy to see in a simple example. If individuals are observed travelling a distance to recreation sites to obtain better "site quality", such as nicer amenities or better fishing, they are implicitly revealing something about the value of quality by passing by the nearer sites of lower quality. They are willing to incur a higher trip cost to obtain more "site quality". By the same reasoning, if individuals choose not to travel to more distant sites, they also reveal implicit values. With a variety of sites located at different distances from individuals' homes (giving variation in trip cost) and with many characteristics, it is possible to reveal implicit values for the characteristics of the sites and even the sites themselves.

The RUM may also be used in stated preference studies such as contingent valuation and conjoint analysis. In a contingent valuation study, for example, an individual's vote for or against a hypothetical project is a choice of one alternative from a set of two. A conjoint analysis or contingent ranking exercise is similar, except that an individual chooses among several hypothetical alternatives. In either case, discrete choices are being made and this fits the RUM framework quite naturally.

[56] Freeman (1993) also raises questions over the traditional assumption that consumer surplus provides a fair approximation of the 'true' value of a site based on price changes for which positive quantities are demanded at both the initial and second prices. He argues instead that the true value of the site should be found by determining the price change sufficient to drive the quantity demanded to zero, in order to derive the maximum willingness to pay.

The remainder of this sub-section focuses on the travel cost RUM, but the approach, whether revealed or stated, is basically the same. There are even instances in the literature now where revealed and stated preferences are combined in the context of a RUM.

Basic Model

In a travel cost RUM, an individual chooses one recreation site from a set of many possible sites. Assume there are C sites denoted as $i = 1,\ldots,C$. Each site is assumed to give an individual some utility. Let U_i be the utility for site i. U_i is assumed to be a function of the characteristics of the site and a random error:

$$U_i = \beta x_i + \varepsilon_i \qquad (1)$$

The term x_i is a vector of the characteristics describing the site, β is an unknown parameter vector, and ε_i is a random error. The recreation sites may be beaches for swimming, lakes for fishing, trails for hiking or some other destination for recreation. The characteristics are attributes of the site that matter to individuals in choosing a site. These typically include trip cost, environmental quality, access, site size and so forth. If individuals in the sample are believed to have different preferences over the site characteristics, this may be accommodated by interacting site characteristics with individual characteristics in the model.

The above equation (1) is intuitive. Sites located nearby an individual's home, with high quality, and good access are likely to have high site utility. More distant sites of lower quality and poor access are likely to have low site utility. Other site utilities will lie somewhere along this continuum. The parameters β convey the differences in site utility. Survey techniques are used to gain information on the sites that individuals have visited, with the chosen sites assumed to have the highest utility. Functions are then developed to reflect the likelihood of observing the pattern of visits actually chosen by individuals. These are then combined with the characteristics of the site and the expected maximum utility of a trip to a site is then calculated, with this being a preference weighted average of all the sites in a person's choice set. It is a preference weighted average because sites with a higher probability of being visited weigh more heavily in the calculations.

The expected maximum utility then provides the basic building block in all valuation work in the RUM. Whether valuing a change in site characteristics or the loss of a site, the expected maximum utility is calculated with and without the environmental change in question, with the difference in utility providing the measure of the resulting change in individual welfare. For further details on how these models are applied in practice see Box 4.1.

Box 4.1: Application of the Travel Cost RUM

In application, β is estimated using data on observed site choices. In a survey of the general population or some targeted population, individuals report sites they visited over a specified time period. For each individual the chosen site is assumed to have the highest utility. If site k is chosen, it is assumed that:

$$\beta x_k + \varepsilon_k > \beta x_i + \varepsilon_i \text{ for all } i \neq k \qquad (2)$$

Since each utility has some random error associated with it, it is necessary to consider the **probability** of observing an individual's choice of a given site in the sample. The probability of observing the n^{th} individual choosing site k is:

$$\Pr_n(k) = \Pr_n(\beta x_k + \varepsilon_k > \beta x_i + \varepsilon_i) \text{ for all } i \neq k \qquad (3)$$

This probability is written for each individual in the sample, and a likelihood function for observing the pattern of alternatives actually chosen in the data set is

$$L(\beta) = \prod_{n=1}^{N} \prod_{i=1}^{C} \Pr_n(i)^{y_{in}} \qquad (4)$$

where $y_{in} = 1$ if individual n chose alternative i and 0 otherwise. N is the number of people in the sample. In estimation, β is selected to maximise the likelihood function. Put differently, the parameter vector β that is most likely to generate the pattern of visits actually observed in the data set is used as the estimate.

The form of the probability Pr(k) depends on the assumed distribution of the random errors ε_i in equation (1). The most commonly used distribution is the type I extreme value distribution for each ε_i which gives rise to a multinomial logit of the form:

$$\Pr(k) = \frac{\exp(\beta x_k)}{\sum_{i=1}^{C} \exp(\beta x_i)} \qquad (5)$$

Other distributions give different forms for the probability, but this is the easiest to estimate and has proven to be quite flexible. Substituting the probabilities from equation (5) into equation (4) and choosing the parameters β that maximise $L(\beta)$ give the maximum likelihood estimates $\hat{\beta}$. The utility for each site then is estimated as:

$$\hat{U}_i = \hat{\beta} x_i + \varepsilon_i \qquad (6)$$

where ε_i, of course, remains unknown. Recognising that ε_i in each site utility is random, an **expected** maximum utility of a trip is computed and used in valuation. Again, if the model is estimated using the logit probabilities in equation (5), the ε_i are assumed to have a type I extreme value distribution. Using this distribution it can be shown that the expected maximum utility of trip is:

$$EU = \ln \sum_{i=1}^{C} \exp(\hat{\beta} x_i) \qquad (7)$$

The form varies with different distributional assumptions for ε_i. The expected maximum utility, often called the “log-sum”, is a preference weighted average of all the sites in a person’s choice set. It is a preference weighted average because sites with a higher probability of being visited weigh more heavily in the log-sum expression.

For a change in site characteristics at one or more sites, the expected maximum utility is computed with and without the change in characteristics using equation (7), to give an estimate of the change in the expected maximum utility due to the resource change. The utility difference is then divided by the parameter estimate on the trip cost variable in the model. This parameter is interpreted as the marginal utility of income, and thus works to monetarise the utility difference. This gives:

$$\Delta W_1 = \left\{ \ln \sum_{i=1}^{C} \exp(\hat{\beta} \tilde{x}_i) - \ln \sum_{i=1}^{C} \exp(\hat{\beta} x_i) \right\} / \hat{\beta}_{tc} \qquad (8)$$

where x_i is vector of site characteristics **without** change in the resource at site i and $\tilde{x}_i$ is the vector **with** the changes. $\hat{\beta}_{tc}$ is the marginal utility of income – the estimated parameter on trip cost. Notice that changes occurring at sites with higher site utility (and hence a higher probability of being visited) have a larger impact on ΔW_1 than sites with lower site utility. Also, note that ΔW_1 is a **per trip** gain (or loss depending on the scenario).

The change in site characteristics (going from x_i to $\tilde{x}_i$) may occur at one or more of the *C* sites and may include changes in water quality, fish catch, addition of hiking trails, improved access, and so forth. If several sites have fishing prohibitions (for example owing to a chemical risk) and these are captured via a simple dummy variable turned on for these sites and off for the other sites, the value of removing these prohibitions may be estimated using equation (8) as well.

The procedure for valuing site closure is similar. The expected maximum utility of a trip is computed with and without the loss of a site or sites. This amounts to dropping sites from the log sum in equation (7). For example, if sites 1 through 5 are closed due to an oil spill or release of a toxic substance, the estimated loss is:

$$\Delta W_2 = \left\{ \ln \sum\nolimits_{i=6}^{C} \exp(\hat{\beta} x_i) - \ln \sum\nolimits_{i=1}^{C} \exp(\hat{\beta} x_i) \right\} / \hat{\beta}_{tc} \qquad (9)$$

Notice that the expected maximum utility with the closure excludes the first 5 sites from the log-sum. These sites are no longer in the choice set and hence have no role in an expected maximum utility calculation. Again, the higher the utility of the lost sites, the larger the loss. Like ΔW_1, ΔW_2 is a **per trip** measure of value.

There is a shortcut to estimating site characteristic values that sometimes shows up in the literature and can be useful in some circumstances. To see this measure, consider a simple RUM estimated with two characteristics: trip cost and water quality. For site *i* the estimated utility is:

$$U_i = \hat{\beta}_{tc} x_{tc,i} + \hat{\beta}_{fc} x_{wq,i} + \varepsilon_i \,. \qquad (10)$$

For a unit change in water quality the ratio $\dfrac{\hat{\beta}_{wq}}{-\hat{\beta}_{tc}}$ tells us the rate at which trip cost must change to keep site utility constant. This is an implicit value for a unit change in water quality in terms of money. This is easy and convenient but is rather restrictive. The ratio implicitly assumes that an individual selects alternative *i* with and without the change in water quality. If the individual substitutes to another alternative, this measure is missing the adjustment in value due to substitution. This adjustment is picked up in the more general measure of value in equation (8). Still, for a quick idea of the relative values of characteristics and even as a first order estimate the ratio works quite well.

The basic model presented can be extended to a number lines. Some of the more important extensions which have become commonplace in travel cost RUMs include the following.

- In many studies, a ‘no-trip’ option is added to the set of alternatives. An individual can visit one of many sites or take no-trip on a given choice occasion. In effect, this version of the model allows for a broader set of substitutes. If this model is repeated in each choice occasion (perhaps daily) over an entire season, it is possible to estimate seasonal values for quality changes and site losses. Because this version of the model allows for non-participants, a broader measure of values is possible. For example, if improvements in water quality lead to increases in total recreation trips in a region, this model picks up that added benefit. The characteristics in the “no-trip” alternative are usually individual characteristics believed to influence whether or not an individual is inclined to take recreation trips. See Morey, Rowe & Watson (1993) for more on this version of the model.

- The basic model suffers from a restriction known as the independence of irrelevant alternatives (IIA). This restriction implies that the relative odds of choosing between any two alternatives in the choice set are independent of the presence of other alternatives. IIA implies a very restrictive pattern of substitution of alternatives in the choice set. This has led to a variety of extensions to the model to allow for a more flexible substitution. The two most important extensions to the basic model along these lines are the nested and mixed logit models. In both cases, a more general pattern of substitution enters the model by allowing for correlation among the error terms in the site utilities. These models are quite popular in recreation demand analyses. For more on nested models see Morey (1999). For more on mixed logit models see Train (1999).

- It is possible to incorporate other decisions into the basic model. For example, fishing models sometimes incorporate the decision of which species to target or which mode to use (boat, shore, or pier) when fishing. Depending on the policy context, a model with a broader set of decisions is sometimes quite sensible. See Parsons & Hauber (1998) for an example.

- As noted in the introduction, in some applications the travel cost RUM is estimated jointly with stated preference data. For example, individuals may be asked how they might respond to hypothetical changes in the resource. These responses generate a new set of "choices" which may be estimated as part of the travel cost model. This allows the analyst to consider behaviour outside the range of observation and to construct hypothetical scenarios that fit specific policies quite neatly. See Adamowicz *et al* (1997) for an example.

Key Issues

A variety of issues surround the use of travel cost RUMs, many of which are shared with the simple single site travel cost model. These include:

- the difficulty of measuring trip cost, especially the component capturing time cost;

- dealing with multiple destination trips (single trips taken to more than one site or for more purposes than recreation);

- modeling overnight trips;

- modeling on-site time;

- defining the choice set to use in estimation; and

- allowing for trip interdependence over time.

Each of these has been dealt with in some way in the context of a RUM, but weak points still remain in the model as well as possible sources of bias.

4.2.7 Hedonic Pricing Methods

The hedonic pricing method (HPM) is based on the concept that the price paid for a complementary good implicitly reflects the buyer's willingness to pay for a particular environmental attribute (*e.g.* a high quality river), or his/her willingness to accept an increased risk (see Section 3 for a discussion of this technique in its application to health risks). These methods determine an implicit price for a good by examining the 'real' markets in which the good is effectively traded.

In the environmental context, hedonic property (land) prices have been used in the valuation of characteristics such as air quality, water quality, fishery quality, noise and other amenity characteristics associated to residential and other properties. It is still commonly used to assess amenity effects, although many analysts have argued that the technique is not reliable in the valuation of environmental effects that are not readily perceptible in physical terms.

The Approach

Hedonic price theory provides a basis for explaining the prices of houses in the market as a function of the levels of characteristics 'embedded' in each property. The approach derives the relationship between an environmental change and the prices of properties in order to attribute this value as a near willingness to pay value. The method assumes the following:

– that environmental quality and attractiveness will be reflected in the prices people are willing to pay for property (where willingness to pay is a function of the following characteristics: physical, accessibility, public sector, neighbourhood and environmental);

– that individuals can distinguish small changes in environmental quality and, where relevant, can understand the implications of these changes;

– that all of the many variations in physical and other attributes can be accounted for in the analysis and any remaining price differences can be attributed solely to differences in environmental quality; and

– that there is free mobility within the property market and that property is available in sufficient quantities to meet demand.

Under this approach, control data are obtained on physical attributes of properties, making it possible to estimate the value of the environmental and other non-physical attributes, for which there is generally no direct market. To these is added a variable that represents the environmental attribute to be valued (*e.g.* air or water quality). The data may be time series, cross sectional or pooled data. A multiple regression analysis is then run, with the model relationship being as follows in its simplest form:

$$P_i = f\,(S_i, L_i, Y_i, E_i)$$

Where
P_i = market price of i^{th} house
S_i = structural attributes (rooms, *etc.*) of i^{th} house
L_i = locational and access attributes of i^{th} house
Y_i = socio-economic attributes of local area of i^{th} house
E_i = environmental attributes of surrounding area of i^{th} house

The partial derivatives (*i.e.* •P/•S and so on) of the above give the analyst the implicit price for each attribute, including for the environmental quality attribute of concern. The differential price between different properties is then equal to the capitalised value of the benefits associated with differences in environmental quality. When summed over all properties, this provides a measure of total benefits.

Key Issues

The key issue with regard to the application of this method in the context of chemical risk management concerns the ability of individuals to perceive changes in environmental quality or to understand scientific information on quality parameters. Where individuals are unable to recognise differences in quality, then their 'true' willingness to pay for environmental quality improvements may vary significantly from the estimated values.

In order to derive a hedonic price equation, three key issues need to be considered:

- variable selection: the data set should be of sufficient size (and variation) to include all relevant variables. Too few variables may lead to bias whilst too many variables may lead to multicollinearity (*i.e.* a relationship between dependent variables);

- variable measurement: it may not be possible to gain measurements for all of the variables included in the analysis. This may be especially true for pollution data, further complicated by the potential subjectivity of those purchasing property; and

- functional form: statistical criteria are usually used to select the preferred functional form of the hedonic equation. The choice of function can clearly heavily impact marginal changes in a pollutant's effect upon property prices.

With reference to the latter point, research by Cropper *et al* (1988) attempted to address this problem. The work simulated the performance of a housing market using real data. They found that when all housing characteristics were included in the hedonic price function, linear and quadratic versions of the Box-Cox transformation provided the most accurate estimates of marginal implicit prices.

Freeman (1993) identifies several additional issues that should be addressed in the application of these models because they rely on the estimation of implicit prices rather than an individual's true willingness to pay for an environmental quality change. While these models provide welfare effect estimates based on the consequences of individuals' choices of residence, they do not capture willingness to pay for environmental improvement. Given that they are based upon observing behavioural responses, they only capture willingness to pay for perceived differences in amenities and their consequences.

Another major limitation includes the underlying assumption that consumers of housing can select their most preferred bundle of characteristics from a complete range of levels across all characteristics (where in reality there are often multiple trade-offs to be encountered in choice of characteristics). Similarly, the property market may not be in equilibrium, causing biases in estimates of the value attached to a particular attribute.

4.2.8 Hypothetical Markets

The Approaches

The two key techniques that involve the use of hypothetical (or experimental) markets are the contingent valuation method and attribute-based stated preference methods. Under the **contingent valuation method** (CVM), individuals are surveyed to determine their willingness to pay for a specified change in the quality or quantity of an environmental good (or how much compensation they would expect for an increase in risk or in environmental damages). The mean willingness to pay value across all valid bids is then used to provide an indication of the economic value of the specified change. For authoritative references concerning the use of CVM in an environmental context see Mitchell & Carson (1989) and Bateman & Willis (1999).

Stated preferences methods (covering conjoint analysis and contingent ranking) involves the elicitation of individuals' ranking of preferences amongst a bundle or 'basket' of different environmental outcomes. Values for changes in environmental goods are derived by 'anchoring' preferences to either a money sum or the real market price of one of the goods included in the bundle/basket of outcomes. See Section 3.2 for a more detailed discussion of the use of these methods.

Within these surveys, questions relating to the construction of hypothetical markets are usually administered by trained interviewers via phone, letter, in person or computer. Comprehensive surveys also break down the sample into subsets and approach the problem from different viewpoints in an attempt to ensure the studies are robust and defensible. The survey instrument itself normally consists of three elements:

- a description of the choice of setting in which respondents are to imagine themselves;

- the choice of questions from which values are derived (either through direct expression, referendum, contingent ranking or contingent activity); and

- questions about the respondents themselves (*e.g.* age, income, sex, education, and so on).

Direct expressions of value are the simplest forms of questions and are usually referred to as contingent valuation surveys. In these surveys, respondents are asked to state their maximum willingness to pay for an improvement or willingness to accept compensation for a loss. Over the years of development of the method a number of elicitation techniques have been used:

- 'the bidding game': where individuals are asked if they would be willing to pay $*x*. If they answer yes then the price is increased until the respondent says no. Unfortunately this approach suffers from 'starting point bias' whereby the entry bid influences the resulting willingness to pay;

- open-ended questions: this approach simply asks respondents how much they would be willing to pay with no start or end point. This approach presents respondents with an alien situation whereby they offer a price with no guidance resulting in many non-responses or very high or low values; and

- open-ended with card choices: this is an extension of the previous approach with the use of a card with a range of values presented to the respondent. The respondent then either chooses a value from the card or offers his/her own valuation.

Referendum questions are based on asking respondents whether they are willing to pay a specified amount for an environmental change. If the response is yes then this means that willingness to pay is equal to or greater than the specified amount; if no, then that amount is taken to represent the upper bound. Variation in the specified amount is achieved via the use of sub-samples, the results from which are used to derive 'bid functions' *i.e.* a certain change in value for a change in environmental quality. This approach has four main advantages over direct expression:

- the questioning takes place in a familiar social context, *i.e.* goods offered on a 'take it or leave it' basis;

- a simple yes or no response is required, raising willingness to participate in the survey;

- there is no (or at least minimal) starting point bias especially if the survey is designed to minimise ‘value clues’ by varying the phrasing of the question each time; and

- the referendum format is incentive-compatible, in that respondents should believe that decisions are being made as an outcome of a voting process removing a need to bid (or vote) strategically.

Key Issues

It could be argued that the distinction made between versions of CV based on direct expression and referendum format is rather arbitrary, as CV applications may fall within a continuum varying between choice/open ended formats and discrete choice/closed ended formats. For example, a dichotomous choice referendum format is a peculiar case of bidding game while elicitation procedures based on payment cards can be closer to a referendum or to an open-ended question according to their specific design.

In using these survey methods, controversy surrounds not only the questions asked (and how they are asked) but also the answers to these questions. Critics of the creation of such hypothetical situations frequently question whether respondents really know what they are valuing *i.e.* can they really separate out a single environmental asset rather than valuing ‘the environment’ as a single entity. In addition, direct questioning methods may suffer from what is termed ‘strategic bias’ where respondents make a conscious effort to either change the value they state or the preferred environmental change in an attempt to influence decision makers.

Key sources of error that need to be controlled for when designing and undertaking a CV study include the following:

- **scenario mis-specification**: it is essential that respondents understand and feel comfortable with the scenario being presented to them by interviewers. In some cases, respondents may simply misunderstand what is being asked and provide an answer that they would not provide if they had full understanding. This source of error can be minimised via the use of focus groups, pilot surveys and pre-tests; these will aid in the design of question formats, scenario design, and to gain an understanding of the perception of the environment amongst the sample group;

- **implied value cues**: if respondents are unsure as to what is being asked of them or are unsure about the concept of a hypothetical market, they may attempt to find ‘clues’ in the way questions are asked or phrased in order to give the ‘right’ answer. If the respondent can find any guidance in this way then the end result will be systematically biased. In order to minimise the occurrence of such errors, questions and scenarios can be randomised in their presentation to respondents; and

- **incentives to misrepresent values**: this problem arises where respondents believe that the response provided will have an effect on either the provision of a public good, environmental quality, taxes, payment responsibilities, and so on. Respondents may then attempt to use their response to influence decision makers. As already mentioned, this is referred to as ‘strategic bias’, a subset of which is referred to as the ‘free rider problem’ where respondents will deliberately state low values in the survey in case they are asked to actually pay the amount stated. It may also be the case that true preferences are over- or under-stated in order to gain more (or less) of a good. There may be danger that strategic bias could become a problem if decision makers frequently make decisions based on the outcome of hypothetical market construction. At present, however, such studies are usually treated with an element of caution by decision makers and, hence, strategic bias tends not to be a major problem.

4.2.9 Application of Monetary Valuation Techniques in Practice

The applicability of the different valuation techniques varies across different types of environmental impacts, as illustrated by Table 4.3. The market price, household production function and hedonic pricing techniques are restricted in their application to valuation of positive and negative benefits on use related services. They are preferred by a number of economists and non-economists, however, in that they do not rely on the use of survey techniques, but instead depend on data that are revealed through actions undertaken by individuals or in the marketplace.

Table 4.3: Valuation Techniques and Environmental Impacts

Impacts	**Valuation Technique**				
	Effects on Production	**Avertive Expenditure**	**TCM**	**CVM***	**HPM**
Water Quality and/or Quantity	Y	Y	Y	Y	Y
Air Quality	Y	Y		Y	Y
Soil Quality	Y	Y		Y	
Recreation			Y	Y	
Landscape				Y	Y
Heritage			Y	Y	
Habitat/Ecosystem				Y	
Wildlife				Y	
Noise		Y		Y	Y
Fisheries	Y		Y	Y	Y
Aesthetics				Y	Y
Forestry	Y		Y	Y	Y

* this includes contingent ranking and contingent activity

All three of these techniques could be used (either directly or indirectly taking the results of previous studies) within a cost-benefit analysis of environmental regulations. Effect on productivity techniques can be applied to the valuation of effects on crop production, fisheries or forestry from the existence of damaging pollutant concentrations. Where an ecosystem has suffered damage as a result of an activity, the costs of replacing or re-creating that ecosystem could be used to develop an estimate of the minimum value which would have to be placed on the original resource for a regulation to be considered worthwhile. Similarly, the amount of money spent by individuals on, for example, water purifiers to reduce concentrations of a particular contaminant (*e.g.* heavy metals) in drinking water could be estimated.

The hedonic pricing method has been applied to the valuation of air or water pollution effects (both diffuse and point source releases) where these are specific to a given chemical, although a number of such studies have found difficulty in making direct links between marginal changes in environmental quality and changes in property prices.

Although the travel cost method (a household production function technique) is constrained to the estimation of recreation benefits at particular sites, it can be applied to the calculation of site-specific losses in recreational activity or quality as a result of the presence or concentration of different chemicals. Past examples include valuation of the impacts of sewage related effluents on bathing beaches and estimation of the impacts on a recreational fishery as a result of ‘high’ concentrations of a specified chemical. In both cases, however, extrapolations from specific sites to an aggregate national level were required for policy purposes.

The experimental market techniques, however, are more flexible than the above techniques; both contingent valuation and contingent ranking methods provide a means of estimating both use and non-use (or passive use) values. As a result, they have been used in the past to derive economic values related to the regulation of environmentally damaging products and activities and their use is likely to continue to be important or grow in importance.

These methods are the most flexible as surveys can be tailored to specific issues. In order to develop reliable estimates, however, surveys need to be carefully administered. The survey instruments used as the basis for deriving valuations must be designed with great care as there are a number of biases that can be introduced into the survey that must be controlled against. In addition, large samples may need to be questioned in order to provide results that are statistically representative of the population affected, particularly in attempts to derive passive values. There is also on-going debate as to what is being measured when people are questioned about non-use values, with several practitioners contending that until such values are better understood estimates should not be included.

Research on non-use values makes it clear though that people hold a true willingness to pay to protect environmental resources that they have never used and do not intend to use. Where their use has been questioned, such as in debates in the US over regulatory reform, the deeper objections are partly about the reliability of survey techniques (and in particular contingent valuation) designed to capture such values, and partly about who should be liable for damages to these values. In order to address concerns over the reliability of contingent valuation surveys, best practice requirements have been established in the US. A panel appointed by the US National Oceanic and Atmospheric Administration in 1991 (Arrow *et al*, 1993) established requirements for US studies submitted as part of Natural Resource Damage Assessments required under the Oil Pollution Act 1990 and under the Comprehensive Environmental Response, Compensation and Liability Act (CERCLA) which concerns hazardous substance releases.

4.2.10 Use of Environmental 'Benefit Transfer' Approaches

As it stands, one of the biggest constraints to the valuation of environmental effects as part of any one cost-benefit analysis exercise is the financial costs of undertaking a valuation exercise and the time required to do so (McConnell, 1992). Clearly, it is not feasible to estimate all environmental damages for each location and time specific situation for each SEA. Much of the work required is extremely time consuming and expensive, making the transfer of estimates from one study to another an important part of the exercise. As a result, economists are increasingly adopting benefit transfer approaches as a cost-effective alternative to the commissioning of issue specific valuation studies. Benefit transfer eliminates the need to design and implement a new and potentially expensive valuation exercise for different sites or for different policies. The difficult question, indeed a key issue, is to know when a damage estimate is transferable and what modifications, if any, need to be made before it can be used in its new context.

Benefit transfer can be defined as the process of taking a value or benefit estimate developed for a previous project or policy decision and transferring it to a proposed project or policy decision. In other words, estimates of the value of a recreational user-day for one specific site and environmental quality change are assumed to provide a reasonable approximation of the value of a recreational day for another site given a similar type of environmental improvement. A generalised approach for using benefit transfer is set out in Figure 4.1 below.

There are three different approaches which might be adopted in benefit transfer (OECD, 1992):

- the transference of mean unit values;
- the transference of adjusted unit values; and

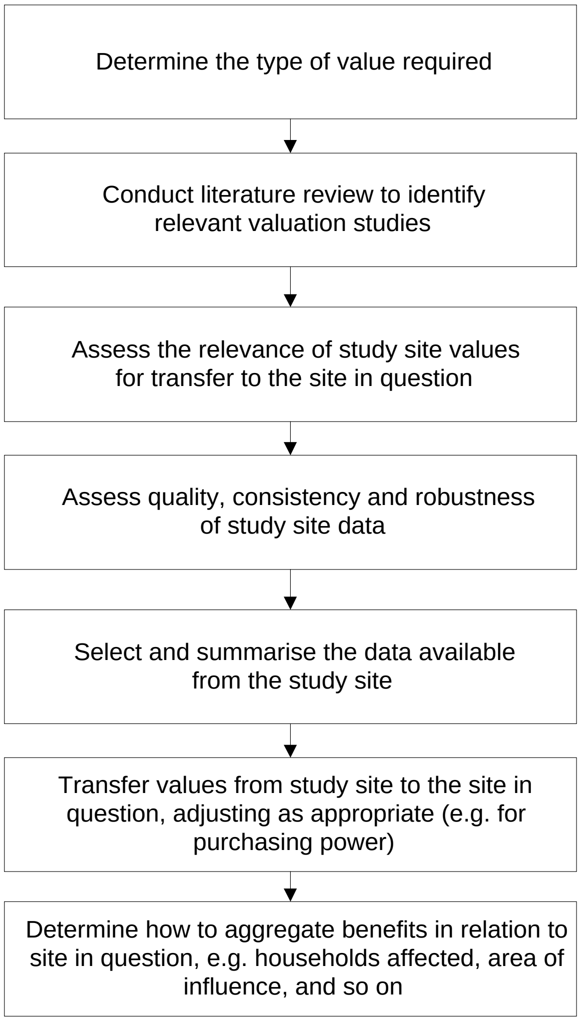
– the transference of a demand function.

The use of mean unit values is obviously the simplest approach that can be adopted. A range of factors may affect, however, the validity or reliability of such an approach. For example, the environmental change measured in the original study may differ significantly in one or more key attributes from the problem currently under consideration; or, indeed, measurement may have been undertaken for a different purpose and some factors relevant to the current decision may not have been considered. At a project level, there may be substitute sites or other opportunities that could affect individuals' valuations.

The second approach, the adoption of an adjusted unit approach, involves the analyst adjusting past estimates to correct for biases incorporated in the original study, or to take into account differences in socio-economic characteristics, project/problem components, levels of damage reduction, site characteristics and the availability of substitute goods. This approach is open to many of the same questions concerning validity and reliability as the use of unadjusted mean values.

The third approach is preferable and involves taking the demand function from a previous study, inputting new data relevant to the project in question and re-running the analysis (Bateman, 1995). The advantages of this type of approach are that calculated benefits are based on information on use and unit values that are derived from the same data set. Adoption of this type of approach, however, is likely to be constrained by there being insufficient information for developing a transferable demand function.

Figure 4.1: The Steps in Benefit Transfer



Where several studies, reporting a similar final estimate of environmental damage, exist, and where there are significant differences between them in terms of the background variables, a procedure known as *meta-analysis* has been developed to transfer the results from one study across to other applications. What such an analysis does is to take the estimated damages from a range of studies of, for example, coal fired plants and see how they vary systematically, according to affected population, building areas, crops, level of income of the population, *etc.* The analysis is carried out using econometric techniques, which yield estimates of the responsiveness of damage to the various context-specific factors that render them more transferable across situations. This can then be used to derive a simple formula relating environmental costs to per capita income, which could then be employed to calculate damages in countries where no relevant studies were available.

Estimates of damages based on meta-analysis have been provided in a formal sense in two studies carried out in the US and the UK on water and forest based recreation demand (Smith & Kaoru, 1990; Bateman, 1995), and on air pollution. The results in the recreation studies indicate that, as one would expect, the nature of the site is significant for the WTP attached to a visit, as are the costs of substitutes and the opportunity cost of time. Choice of functional form in the estimating equations also appears to play a part. In the air pollution study referred to above, it was found that damages per unit concentration vary inversely with the average price of property in the study (the higher the price the lower the unit value of damage). If correct, it would enable an adjustment to the estimated value to be made on the basis of the average prices of properties in the area being investigated. However, the authors are cautious about the validity of the estimates obtained.

A formal meta-analysis is difficult to carry out, and will not prove possible for most projects. However, some of the 'expert' adjustments can make an informal meta-analysis. For example, adjusting estimates of damages for size of population to obtain a per capita estimate and transferring that to the new study implicitly assumes that damages are proportional to population. Such adjustments are frequently made.

At a general level, therefore, a number of potential difficulties arise in transferring values from one study to a new policy or programme context. The first of these concerns a lack of previous studies that have examined the same environmental quality change under consideration. For example, difficulties occur in benefit transfer when considering new policies as these will not have been considered by past valuation exercises. Extensions of current policies may also be difficult to value using benefit transfer methods. Water or air quality changes, for example, are unlikely to be linearly related to benefits, so a policy that is more stringent than any previously may not justify the extrapolation of previous benefits.

There are also problems in transferring estimates developed for one country with particular cultural and socio-economic characteristics (*e.g.* the US) to other countries (*e.g.* EU countries); this may be particularly true with regard to the transfer of values from developed countries to a developing country context. Cultural factors may be significantly different, as may be perceptions of relative damage levels and risk, and these may invalidate the straight transfer of benefit estimates. In addition, WTP measures will depend upon income and care must be taken to consider how these values should be adjusted in moving between countries with different income levels (Krupnick *et al*, 1993). Several different approaches have been suggested, such as adjustments according to relative income, according to purchasing power and/or environmental awareness. Using such approaches assumes that WTP for environmental quality varies proportionately with income, but damage costs are not necessarily constant across countries in terms of income.

In addition, with regard to the transfer of single mean values, it must be remembered that no one model will provide the all embracing or an unassailably valid estimate of the value of environmental damage. Individual studies and their results are specific to particular issues and situations, in both space and time, and their applicability to other cases is questionable. As a result of people's perceptions of risk and environmental quality, some forms of impact may be viewed more seriously than others. People may be prepared to pay more to reduce or avoid some types of effects (*e.g.* loss of a 'charismatic' species or particular habitat type – such as rainforest versus mudflats) than others. Questions over applicability are even more important when trying to compare values for dissimilar environmental issues, for example sewage effluent discharges versus a catastrophic oil spill.

The accuracy and quality of the original value is therefore crucial to the use of benefit transfer. To ensure robustness, Desvousges *et al* (1998) set a number of criteria for assessing the quality of values chosen for transfer.

- Scientific soundness: the transfer estimates are only as good as the methodology and assumptions employed in the original studies. Specific criteria may include:
  - data collection procedures;
  - sound empirical practices;
  - consistency with scientific and economic theory; and

  - appropriate and rigorous statistical methods.

- Relevance: the original studies should be similar and applicable to the new context. Specific criteria may include:

  - magnitude of impacts should be similar;

  - baseline levels of environmental quality should be comparable so that the underlying dose-response relationships remain valid;

  - affected goods or services should be similar;

  - affected sites should also be similar;

  - the duration and timing of the impact should be similar;

  - the socio-economic characteristics of the affected populations should be similar; and

  - the property rights should reside with the same party in both contexts.

- Richness of information: the existing studies should provide a ‘rich’ dataset and accompanying information. Specific criteria may include:

  - inclusion of full specification of original valuation equations as well as mean values;

  - explanation of how substitute commodities were treated;

  - data on participation rates and extent of aggregation employed; and

  - provision of standard errors and other statistical measures of dispersion.

In order to minimise the uncertainty associated with benefit transfer, it may be the case that the analysts use certain statistical techniques taking into account the level of perceived uncertainty. This may simply mean using an upper and lower bounds within an analysis or running a full Monte Carlo simulation to determine the most probable value if probability density functions are known (see also Section 3 of Part 4). Of course, sensitivity analysis provides the opportunity to vary key parameters within the SEA; it may even be appropriate to derive switching values to determine ‘how far’ the transferred estimates can be altered to change the outcome of the SEA. A full range of testing ensures that the analysis should be as robust and defensible as possible.

4.2.11 More General Issues in Economic Valuation

Even where a comprehensive risk assessment has been undertaken, assessing the significance of the environmental benefits of risk reduction is a difficult problem. When dealing with risks to human health, levels of damage are generally applicable and these are fairly universal across different geographical areas - for example, a case of severe dermatitis to an individual in London is as significant as a similar case in Paris or Milan.

For the risk manager, this consistency between outcomes makes it relatively easy to collect the data necessary to assess the trade-offs between risk reduction and costs where human health is concerned. In the case of environmental risk management, consistency between either the affected ecosystems or the resulting benefits is rare, making comparisons extremely difficult.

This is one of the reasons underlying the use of economic valuation and CBA within regulatory appraisals. Economic valuation stresses the importance of ensuring that the *total economic value* of an environmental asset is considered when assessing the benefits of environmental risk reduction, where this includes values related to the use of the asset – today and in the future – and to the desire to ensure that the asset is being preserved. A range of monetary valuation techniques can be drawn upon to assist with this including those which use actual data on the market value of losses or gains, those which infer values from consumer behaviour and those which directly elicit individual's willingness to pay for a specific environmental outcome.[57]

As described in earlier sections, conversion of predicted environmental damage into monetary values provides one means of improving consistency and has the added advantage of converting environmental damage into the same unit as the costs of undertaking risk reduction - *i.e.* money. However, it may also focus the benefit assessment on those species, communities and ecosystems which have a recognisable value to people as these are likely to be the easiest to 'convert' into a monetary value. Where risk assessments are not targeted as much towards such endpoints and/or where these endpoints have not been translated into a readily understood expression of damage, valuation is much more difficult to undertake.

As a result, concern has been voiced that the attributes of environmental assets traditionally valued by the general public (*e.g.* hunting, fishing, recreational activities, scenic views, pollution assimilation capacity) do not sufficiently reflect the requirements of well-functioning ecosystems. As such, while the public may appreciate the value of an asset as a whole, they may still overlook the ecological significance of a given species or community. This means that there may be a tendency to undervalue, or to not value at all, many of the services provided by ecosystems and, hence, the value of protecting these services. Most of the economic techniques are suitable to monetarise changes in environmental assets rather than the loss of entire ecosystems. The need to address such concerns, however, has been recognised and preliminary efforts have been made to derive more holistic values for ecosystems (see, for example, Costanza *et al* (1997) which provides one such attempt at meeting this challenge).

4.3 Non-Monetary Assessment Methods

4.3.1 Introduction

Part 2 of this document summarised the key principles of multi-criteria analysis (MCA) and provided an overview of how it can be used as part of chemical risk management. One of the points that was stressed was that these techniques focus on addressing the multi-objective nature of environmental (and other) decisions. They were developed explicitly with the aim of allowing decision makers to incorporate their judgements and values (on behalf of stakeholders) into the decision making process. It is often argued that this is particularly important in environmental decision making given that such issues are generally characterised by the need to manage competing interests, conflicting objectives and different types of information. This is certainly true within the context of chemical risk management.

This section discusses in more detail the MCA techniques that are or could be applied to chemical risk management decision making. It draws on the overview presented in Part 2, but

[57] For further discussion see, for example: OECD (1995), Turner *et al* (1992), Freeman (1993).

expands on this to provide a better understanding of how these techniques are used in practice and some of the issues involved. Given the range of MCA techniques that are available, however, it is impossible within a document such as this to go into each in much detail.[58]

Note that the nature of the discussion provided here is different to that given in the section on human health with regard to non-monetary assessment methods. The key reason for this difference lies in the fact that there are well developed methods in the health field which are relevant to chemical risk management, while these are still lacking with regard to the environment and chemical risks.

4.3.2 Value and Utility Measurement

The steps involved in undertaking a MCA-based assessment were set out in Section 4 of Part 2. Following the initial activities of screening options and defining the criteria, the next step is to determine the approach to be taken to assess, or 'value' the impacts. Most impact assessors favour quantifiable criteria. However, leaving aside the costs of obtaining the necessary data for assessing performance against such criteria, quantitative methods of comparison are by no means always the best for describing an impact. Moreover, because relevance and quantifiability do not necessarily go hand in hand, qualitative criteria will have to be used in many assessments.

Scores can be assessed in many ways. Examples are simulation models, laboratory tests, direct measurement, and expert judgement. Scores can be measured on a quantitative scale such as ratio, interval or monetary scale, or on a qualitative scale such as an ordinal, +++/--- , or a binary (yes/no) scale. Any systems used for scoring (which involves measuring values and utilities) are simply a set of prerequisites and rules for assigning numbers to valued objects, with human judgement being a component of this process.

Three key steps can be identified in the process of value and utility measurement:

- **Definition of criteria/impacts**: the criteria or impacts to which values or utilities are to be attached must be carefully defined.

- **Scale construction**: when it makes sense to do so, a scale is selected or constructed that represents some natural quantitative attribute of the objects to be assessed. This makes sense if the objects are, for example, costs or environmental concentrations of a particular chemical compound. However, such quantitative scales are not always readily available (*e.g.* for distrubutional effects) or are too remotely related to the values of the decision maker to be useful. In such cases, qualitative scales can be constructed that carefully define the attribute, its end points, and perhaps some intermediate marker points by means of verbal descriptions. In such cases, it can be useful to associate numbers with qualitative descriptions that are at least ordinally related to the decision maker's preferences for the levels of the qualitative scale.

- **Construction of a value function (utility function)**: the 'natural' scale, if there is one, is converted to a value scale (or utility function) by means of judgements of relative value or preference strength. If scale construction has been skipped, for whatever reason, Step 3 works directly with the criteria for evaluation rather than with the physical scale against which they have been measured.

The techniques available for assessing value and utility functions were categorised in Table 4.1 of Part 2 according to the type of outcome of concern and the judgement required of the decision maker. As indicated by the table, the techniques can rely on the use of either direct numerical

[58] For more detailed discussions of the various methods, see Janssen (1992).

judgements or on indifference (choice) judgements [see also Keeney & Raiffa (1976), Keeney (1992) and Beinat (1997)].

Value Functions through Numerical Estimation Methods

Direct rating, **category estimation**, **ratio estimation** and **curve drawing** are versions of numerical estimation methods in which respondents are presented with some anchored scale and asked to rate or otherwise numerically estimate the attractiveness of an outcome relative to the anchors (*e.g.* a predicted environmental concentration post risk reduction compared to an 'anchor' level which could be lower or higher). No equivalent of the numerical estimation methods exists in the case of utility measurement.

Direct rating, with or without a graphic aid, is probably the simplest and most commonly used numerical estimation approach for value measurement. Under this approach, a respondent is presented with a number of alternatives and asked what is the worst and best alternative. These alternatives then serve as the end points of the scale and receive a value of 0 and 100 respectively. The respondent is instructed to consider carefully the relative value of the alternatives. Next, the respondent is asked to rate the remaining alternatives, which fall between these end points, in such a way that the relative spacing on the scale reflects the relative value of all the alternatives.

Two simplified variations of this approach are **category estimation** and **ratio estimation**. In category estimation, the possible responses are reduced to a finite number of categories. For example, when judging the level of environmental improvement, the respondent may be given the following category scale:

Very low improvement, –3, –2, –1, 0, +1, +2, +3, Very high improvement

The respondent is asked to sort possible levels of improvement so that adjacent categories are considered equally spaced in value. The categorisation of the scale makes the task somewhat simpler, but fine distinctions may get lost, unless the number of categories is increased. Another characteristic of this technique is the loose use of qualitatively defined end points.

Category estimations are easier to make than simple ratings; ratio estimates are more demanding. In ratio estimation, one outcome (*e.g.* a defined level of environmental impact, such as the predicted no effects concentration) is presented as a standard, and the respondent is asked to compare all other outcomes (predicted concentrations) with that standard. For each, the respondent is then asked to state how much more or less valuable that environmental concentration is than the standard, in a ratio sense. This judgement assumes a well understood point of 0 value. Based on experience with this method in health, it is probably best to use the disvalue version in which the well understood point of 0 disvalue is a zero concentration or perhaps the no effects concentration. Then, all other concentrations can be compared to each other on a ratio scale of how many times worse they are compared to the least bad.

Value Functions through Indifference Methods

Difference standard sequence and **bisection techniques** are based on concepts concerning the strength of preference or value difference. Both require that outcomes are within a narrow range and that they can be varied by small increments. In the difference standard sequence method, the respondent identifies a sequence of outcomes that are equally spaced in value. For example, outcomes *a, b, c, d, …* are found such that the strength of preference of *a* over *b* is equal to the strength of preference of *b* over *c* over *d,* and so on. In the bisection technique, the most preferred outcome and the least preferred outcome are identified, and subsequently a midpoint outcome is found that is equidistant in value from both extremes.

The **variable probability method** and **variable certainty equivalent techniques** are indifference methods in which a sure thing (a so-called certainty equivalent) is matched to a gamble, either by varying the probabilities of the gamble or by adjusting the certainty equivalent [Keeney & Raiffa (1976), Winterfeldt & Edwards (1986) and French (1988)]. Using the variable probability method, the best and worst alternatives are selected first from a set of alternatives. Next a gamble is constructed where the outcome associated with the best alternative is assigned a probability of *p* and with the worst alternative a probability of *1-p*. The respondent is asked to compare this gamble with the certain outcome of one of the intermediate alternatives and to provide the probability where the certain outcome is equivalent to the gamble. This probability is used to estimate a utility function.

The certainty equivalent method works along the same principles. In this case, a gamble is constructed with a 50% probability of the best outcome and a 50% probability of the worst outcome. The respondent is asked to compare this gamble with the certain outcome of one of the intermediate alternatives and to provide the intermediate value where the certain outcome is equivalent to the gamble.

Other non-linear, multiplicative or multilinear utility functions are proposed in the extensive literature on utility analysis. These functions are generally believed to provide a better theoretical foundation for describing the utilities of decision makers. They are, however, seldom used in practical applications: first due to computational problems and second due to lengthy assessment procedures [Winterfeldt & Edwards (1986); French (1988)].

4.3.3 Assessment of Weights

Weights can be perceived as trade-offs: how much of criterion/impact X is a decision maker willing to sacrifice for the benefit of criterion/impact Y. A number of methods are used in assigning weights, with these again varying from those relying on quantitative information to those relying on qualitative information on priorities. Assigning weights is a complex task. The best known technique is the use of swing weights, which can be used to support quantitative trade-offs between ranges of scores of two or more criteria (Winterfeldt & Edwards, 1988). Other techniques, such as the expected value method or the random weights method, ask the decision maker to order the criteria according to importance and translate this ordering into weights (Janssen, 1992).

In terms of the quantitative approaches, the **direct rating** of criteria (*i.e.* allocating 100 points across the various criteria so that the number of points assigned to each reflects its relative importance - see for example, Belton, 1999) is rarely adopted as a weighting procedure because of the difficulties which decision makers experience in performing such exercises. Instead, it is more common for a swing weighting procedure to be adopted.

Swing weighting (also referred to as trade-off assessment) is closely related to utility analysis and can be used in situations where a linear utility function applies, or can serve as an approximation for a more complex utility function. Under this method, the decision maker is asked to indicate values for weights by answering questions of the type: how large should X be in order to guarantee that an improvement of one unit of criterion 1 is equally attractive to an improvement of X units of criterion 2. By repeating this question for all pairs of criteria, trade-off or swing weights can be determined. Keeney & Raiffa (1976) refer to the weights as scaling factors because these elements are used to relate the utility functions estimated for all attributes to each other. Related methods are cross attribute indifference and cross-attribute strength of preference methods (Winterfeldt & Edwards, 1986).

The use of **pairwise comparisons** is one of the more commonly used qualitative techniques. The aim of this method is to derive quantitative weights from qualitative statements on the relative importance of different criteria. The weights are obtained from undertaking systematic comparisons of all pairs of criteria. Saaty (1980) proposes the application of a nine-point scale to express differences in importance:

- equally important;
- moderately more important;
- strongly more important;
- very strongly more important; and
- extremely more important.

Intermediate values (2, 4, 6, 8) can be used if it is too difficult to choose between two successive classes (owing to uncertainty for example). The decision maker is asked to compare all pairs of criteria. The results are included in a pairwise comparison matrix with the largest Eigenvalue[59] of this matrix being used to calculate a set of weights.

Ranking methods can be used if the decision maker is able to rank the criteria in order of importance. Ranking methods treat this ordering as information on the unknown quantitative weights and try to make optimal use of it. For example, assume that weights are ordered according to priority. Criterion 1 is more important than criterion 2, which is more important than criterion 3, and so on. By convention, the weights across all criteria add up to one. This results in a set of feasible weights that conform to both constraints: the priority order and the sum of 1.

There are also a series of methods that use the information on the set *S* of feasible weights to produce quantitative weights [see also Janssen (1992) and Beinat (1997)], with these including:

- the expected value method;
- the extreme value method; and
- the random value method.

These three ranking methods use qualitative information on priorities to derive a ranking of the alternatives. The **expected value method** is straightforward and results in a complete ordering but is dependent on strict assumptions concerning the distribution of weights. The **extreme value method** supplies all possible rankings but therefore often results in an incomplete ranking. The advantage of the **random value method** is that information is provided on the probability of the rankings, given the qualitative information on weights.

The expected value method assumes that each set of weights within *S* has equal probability. The weight vector is then calculated as the expected value of the feasible set. This method gives rise to a convex relationship between ordinal and quantitative weights: the difference between two subsequent weights is larger for more important criteria. In contrast, the extreme value method uses the weight vectors on all extreme points of the set *S*. Rankings are determined for each of these weight vectors. Only orderings of alternatives that are found for all of the weight vectors are included in the final ranking.

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59 The Eigenvalue is a value • for matrix **A** such that • **A** - • **I** • = 0, where **I** is the identity matrix. They are also known as characteristic roots of square matrices.

Similar to the expected value method, the random value method assumes that the unknown quantitative weights are uniformly distributed within the set of feasible weights. Different vectors of feasible weights can result in different rankings of the alternatives. Using a random generator, the ranking of alternatives is determined for a great number of points in *S*. This results in an estimate of the areas in *S* that are linked to a certain ranking. A summation procedure is used to translate the probabilities assigned to rankings to probabilities that a certain alternative will obtain a certain rank number. This method results in the same ranking as the expected value method but provides the decision maker with information on the robustness of this ranking given the limited information on priorities.

Ranking of Options and Sensitivity Analysis

All multi-criteria methods transform the input, performance scores and weights to a ranking using a decision rule specific to that method. The end comparison of options can then involve:

- identifying the best alternative from all available alternatives;
- identifying a number of alternatives as potential candidates for implementation; or
- providing a complete ranking of the alternatives.

A vital element of this process will be the inclusion of a sensitivity analysis, which should examine the reliability of the results by making small changes in the allocated weights and scores to see if this alters the ranking of options (see also Herwijnen *et al*, 1995).

4.3.4 A Typology of the Methods

Building on the above discussion, it is important to recognise that the available overarching evaluation methods differ as to the characteristics of the set of options and measurement scales that they can handle, the decision rule that they apply, and the way scores are standardised. It is impossible to review all of methods that make up the enormous array of those available. A selection has been made from various classes of methods with particular relevance to environmental problems. Extensive reviews are given in Nijkamp *et al* (1990), Janssen (1992) and Vincke (1992). See also DETR (2000) for a review of the different methods and guidance on their application, including case studies examining use as part of government decision making.

In general, the various methods can be classified in terms of three main characteristics.

1. The set of alternatives: discrete versus continuous problems. All multi-criteria decision problems can be represented in multi-dimensional space. Discrete decision problems involve a finite set of options. Continuous decision problems are characterised by an infinite number of possible alternatives. The selection of a risk reduction option from nine possible options is an example of a discrete choice problem. The allocation of nuclear, coal and natural gas resources for the production of electricity is an example of a continuous decision problem.

2. The measurement scale: quantitative versus qualitative attribute scales. Many problems include a mixture of qualitative and quantitative information. Qualitative and mixed multi-criteria methods such as the regime method, permutation method, evamix method and expected value method can process mixed information. Evaluation by graphics can be used in quantitative, qualitative and mixed decision problems.

3. The valuation function: Quantitative scores can be measured in a variety of measurements units. To make these scores comparable, they must be transformed into a common dimension or into a common dimensionless unit. Scores can be transformed into standardised scores using a linear standardisation function, or by using value or utility functions. Value and utility functions transform information measured on a physical measurement scale to a value or utility index.

Following the above classification, three main categories of methods can be identified:

- multi-attribute utility analysis including: weighted summation, ideal point method, and evaluation by graphics;
- outranking methods; and
- qualitative methods including: the analytic hierarchy process (AHP), regime method**,** permutation method, and the evamix method.

In addition to these formal evaluation methods, decision makers and others commonly rely on the use of single elements of these methods to provide simplified decision aiding tools; in particular, pairwise comparisons and ranking methods are used in this manner. For this reason, these are discussed in more detail below, prior to discussion of the formal evaluation methods. It should be noted, however, that use of these techniques outside the more formal evaluation methods is not recommended as they do not on their own meet theoretical validity requirements.

4.3.5 Simple Choice Methods

Pairwise Comparison Techniques

At a simple level, pairwise comparisons are often used as a means for conveying information to decision makers on the degree to which one option outperforms another across a range of decision criteria. In these simple applications, however, no attempt is made to incorporate any judgements as to the relative importance of different magnitudes of impact or of the different criteria.

As described above, the first stage in undertaking pairwise comparisons involves listing the criteria or impacts and comparing options in pairs against each of these, indicating a preference for one option over another. The results are then recorded in a table, such as Table 4.4, to illustrate which alternative performs better or worse for each of the criteria. An overall preference is then identified, or the information is used to highlight the trade-offs involved in selecting one option over another. The information is then provided to decision makers who must make a judgement on the relative importance to be assigned to the different criteria and thus to determine the ‘best’ option.

Table 4.4: The Use of Pairwise Comparisons

Risk Reduction Measure	Health Risks	Environ-Mental Risks	Costs to Industry	Costs to Regulator	Costs to Consumers
A versus B	A>B	A=B	A<B	A=B	A<B
A versus C	A>C	A>C	A<C	A<C	A=C
B versus C	B>C	B>C	B>C	B<C	B<C

From the above comparisons, the preferred options are, in terms of:

- Health risks: Option C is preferred as it results in lowest level of risk: C<B<A
- Environmental risks: Option C is preferred as it results in lowest level of risk: C<A, B
- Costs to industry: Option A is preferred as it results in lowest costs to industry: A<C<B
- Costs to regulator: Option A or B is preferred as both are equal and lower than C: A, B < C
- Costs to consumers: Options B is preferred as it results in lowest costs: B<A, C

Although this approach is readily applied to problems with only a few options or criteria, undertaking the comparisons and ensuring consistency becomes increasingly complex as the numbers of criteria and options increase. Applying pairwise comparison techniques in such cases can only effectively be achieved through the use of the more sophisticated mathematical approaches (such as the analytical hierarchy process) which have been developed for these purposes.

Ranking Methods

Ranking involves the ordering of options or impacts into ranks using verbal, alphabetical or numerical scales and provides an indication of relative performance. Value judgements (*e.g.* expert opinion or a decision maker's) are used to decide on the order of preference for different options or impacts. So, for example, if there were five options and a numerical scale was being used, the 'best' option would receive a ranking of 1 and the 'worst' a ranking of 5.

Related to the use of ranking methods is trend analysis, which provides an alternative (rating based) method of comparing options and can be used to provide quick indicators of the potential implications of a proposed regulation. Table 4.5 illustrates the type of table that could provide the basis for trend analysis as part of chemical risk management to indicate the magnitude of predicted impacts for different criteria of concern (see also EC, 1998). As can be seen from the example, this type of analysis may be useful at a preliminary level to identify the potential impacts of a risk reduction measure (through initial consultation of stakeholders and a review of existing scientific and other data). Because it provides an instant overview of the key impacts, like screening techniques, it may be quite valuable in communicating information to decision makers.

Table 4.5: Trend Analysis of an Option for Chemical Risk Management				
Key Sector	**Advantages**	**No Effect**	**Drawbacks**	**Comments**
Producers/Manufacturers			TT	
Related Industries			T	
Consumers			T	
Distributional Issues		T		
Change in Health Risks	TT			
Changes in Environmental Risks	TT			
Key: T Negligible impact; TT Medium impact; TTT Large impact				

These methods obviously provide a simple means of evaluating the performance of different options over a range of different criteria. However, when used on their own, they provide little information on the degree or magnitude of any differences in impact between options. They, therefore, hide any uncertainty that may exist as to the extent of such differences. In addition, when there are several options under consideration, it may be difficult to select a preferred option. This latter problem has led to the tendency for people to add ranks (or trends) together, a mathematical operation which is invalid unless it is assumed that: decision makers places an equal value on impacts falling under the various criteria (*i.e.* that impacts on consumers are equally important to changes in environmental risks); and that all trend scores or ranks reflect proportional changes in level of impact (*i.e.* +++ is three times better than +).

Such methods must, therefore, be backed up by further descriptive information if decision makers and others are to be provided with an accurate picture of the implications associated with alternative risk reduction options.

4.3.6 Multi-Attribute Utility Analysis

The Approach

Multi-attribute utility analysis in its simplest linear form is usually referred to as weighted summation. More complex variations include the use of non-linear value or utility functions or methods based on distance to target such as the ideal point method. All of the methods that fall under this heading evaluate the attractiveness of alternative options in terms of two discrete elements:

1. The consequences of the options in terms of the decision criteria. Consider i $(i=1,\ldots I)$ options and j $(j= 1,\ldots J)$ decision criteria. Let x_{ij} denote the impact of option i on criterion j. The matrix X of size JxI includes all information on the performance of the options (the effects table).

2. The relative preference or priority assigned to the decision criteria are denoted in terms of weighs w_j $(j=1,\ldots J)$ which are contained in the weight vector $\underline{w}$.

3. If the effects scores x_{ij} for the criteria are measured on different measurement scales, they must be standardised to a common dimensionless. The standardised scores are denoted by $\hat{x}_{ij}$

The methods vary, however, with respect to their treatment of *X* and w. In common, though, the methods require the use of quantitative scoring systems as well as the weights assigned to reflect the relative importance of those criteria being quantitative in nature. If only qualitative information is available for the purposes of weighting, then the procedures for generating quantitative weights (*e.g.* the expected value method) described above can be applied.

Weighted Summation

The steps involved in applying the weighted summation method are:

1. Standardise the scores across all criteria.

2. Assign preference weights.

3. Multiply the weights by the standardised scores.

4. Add up the resulting scores to obtain total weighted scores for each option.

5. Determine the ranking of the total weighted scores.

An appraisal score is calculated for each alternative by first multiplying each value by its appropriate weight and then summing the weighted scores for all criteria. The best option is found as follows:

$$\underset{\mathrm{i}=1,\ldots,\mathrm{I}}{\text{maximise}} \quad \sum_{j=1}^{J} \left(w_j \; x_{ji} \right)$$

If the scores for the criteria are measured on different scales, they must be standardised to a dimensionless scale (*i.e.* a value function) before weighted summation can be applied. The more simple numeric estimation methods are used for standardising scores, for example the use of direct rating methods as described above. The assignment of weights is frequently undertaken through the use of swing weighting procedures.

Although the method requires quantitative information on scores and priorities, only the relative values are used in the assessment. The method does, however, provide a complete ranking of options and information on the relative differences between options.

Ideal Point Method

Weighted summation as described above is based on the concept of value/utility maximisation. The ideal point method uses the same concept to rank the options in terms of the degree to which they achieve a pre-specified target or ideal situation (*i.e.* their distance to the target outcome). It is assumed that there is an ideal level of impact for the criteria of concern and that the decision maker's utilities decrease (monotonically) as one moves away in either direction from this level. Options that are closer to the ideal are preferred to those that are further away.

The following equation is used in the ideal point method.

$$\underset{i = 1,\ldots, I}{\textit{minimize}} \quad d_i$$

where

$$d_i = \left[\sum_{j=1}^{J} \left(w_j \, | \, \hat{x}_{ji} - \max_{i} (\hat{x}_{ji}) \, | \, \right)^p \right]^{1/p}$$

The scaling coefficient *p* makes it possible to include a relationship between relative size of the effect and weight into the decision rule. If there is a linear relationship, such as the balancing of benefits over costs, *p* must be 1; if there is decreasing marginal utility, *p* must be higher than 1. The value of *p* depends on the policy objectives. If, as is often the case with air and water pollution, only the highest value is relevant, *p* must be infinite (this metric is known as the weighted Tchebycheff metric).

Because chemical risk management is concerned with removing risks to human health and the environment by reaching a certain desired state (*i.e.* an acceptable level of risk), this method may be more appropriate than one where the decision rule focuses on maximising the overall performance of the options. Assuming that the options cover all extremes, the ideal point can be found by identifying the maximum possible value for each criterion. Alternatively, the decision maker can define the ideal target levels for all criteria. Various distance measures can then be applied to establish the distance between the ideal point and each alternative (Steuer, 1986).

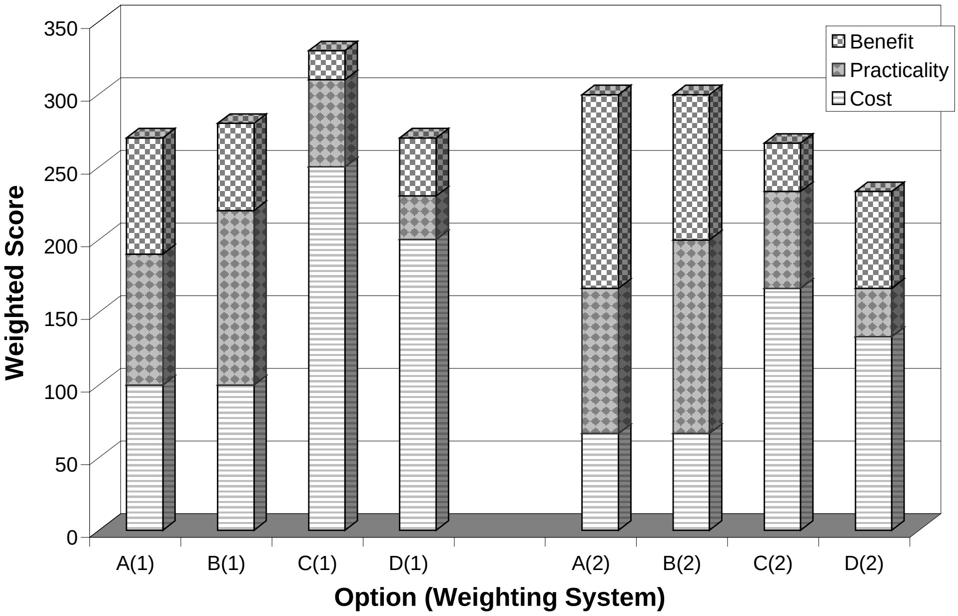
Similar to the weighted summation method, the ideal point method provides a complete ranking of options and information on the relative distance of each from the ideal solution.

Evaluation by Graphics

As with many of the decision support tools described in this guidance, computerised models often provide a graphical interface to facilitate the development and analysis of a decision problem using multi-criteria analysis (Bielza & Shenoy (1999) and Klimberg & Cohen (1999)). DETR (2000) provides a brief review of some of these models including *HIVIEW, MACBETH, VISA* and *DESYSION DESKTOP*.

One of the key benefits of using a graphical output is that it enables the analyst or decision maker to see easily the relative performance of options under different weighting systems. By way of example, consider four risk management options (A to D) which are scored against three attributes (benefit, practicality and cost) on a simple five point scale. The scores are weighted using two approaches - the first assigns weights of 50, 30 and 20 to cost, practicality and benefit respectively while the second assigns equal weights of 33.3 to each. The resultant weighted scores are shown in Figure 4.2**.** From this, it can be seen that with a greater importance (weight) given to cost, Option C appears to be the best (as represented by a weighted score of 330 for C(1)). However, with equal weights, Options A and B emerge as front runners (as represented by weighted scores of 300 for A(2) and B(2).

Figure 4.2: Evaluation by Graphics using MCA for Risk Management



With the rapid development of GIS software tools, it is possible to use graphics techniques to evaluate the geographical variation of the impacts of risk management options. For example, the benefits of some options may accrue to those working in or living close to production facilities while for other options the benefits may accrue to consumers generally.

Key Issues

Multi-attribute utility analysis is well founded in utility theory. Its simplest form, the linear additive model, also known as the 'weighted summation' method, can provide a good approach to the assessment of problems such as chemical risk management, given that transparency will be important.

The computational requirements of the methods are fairly simple. The difficulty with weighted summation does not lie in the calculation but in choosing a good standardisation method (the way in which the scores are converted to a common denominator) and attribution of the weights. In general, linear standardisation can be seen as an approximation of more complex non-linear value functions. If the range of scores is not too large for all criteria, linear standardisation will in many cases be sufficiently accurate. If the range of scores is large or if the scores are in a range where the value of a score is very sensitive to changes in the range, expert judgement should be used to determine the shape of the value function. This is, however, a relatively time consuming activity.

One of the key disadvantages of the method is that it is less suitable for processing qualitative information. In practice, this disadvantage is not very significant because in many cases qualitative scores, for example plusses and minuses, represent underlying classes of quantitative data. With a well chosen method of standardisation, this underlying quantitative scale can be used in the weighted summation of these scores.

The results of weighted summation can easily be presented in bar graphs. A stacked bar can be used to present the relative contribution of all criteria or objectives to the overall score. In addition, Monte Carlo analysis (see Part 3, Section 5 for further discussion) can be used to generate probabilistic rankings of the alternatives. These rankings can be used to analyse the sensitivities of ranking of alternatives to overall uncertainty in both effects and priorities (Herwijnen *et al*., 1995).

Line graphs can easily be generated to show the effect of changes in weights or scores to the total scores and ranking of the alternatives.

4.3.7 Outranking Methods

The Approach

The various Electre methods (also referred to as concordance analysis) are the most important representatives of the class of outranking methods. These methods are widely used especially in French-speaking countries, and are based on what Roy calls the 'fundamental partial comparability axiom' (Roy, 1985). Outranking methods first translate criterion scores to an outranking relationship; and then analyse this relationship.

Within this method, a dominance relationship for each pair of alternatives is derived using two indices, one index indicating *concordance* and the second index indicating *discordance*. The concordance index represents the degree to which alternative *i* is better than alternative *i'*. This index, $c_{ii'}$, is simply the sum of all the weights for those criteria where alternative *i* scores at least as highly as option *i'*. The set of such criteria (*i.e.* those for which alternative *i* is at least equally attractive as alternative *i'*) is referred to as the concordance set $C_{ii'}$. The formulae used in calculating the concordance index is set out below.

$$c_{i'i} = \sum_{j \varepsilon C_{i'i}} w_j$$

The discordance index, $d_{ii'}$, is more complex. It reflects the degree to which alternative *i* is worse than alternative *i'*. So, if alternative *i* performs better than alternative *i'* on all criteria, the discordance index is zero. If not, then for each criterion for which alternative *i'* performs better than *i*, the discordance index is given by the ratio between the difference in performance level between *i'* and *i* and the maximum observed difference in score on the criterion of concern between *any* pair of alternatives in the set being considered (*i.e.* out of *i, i', i'',…)*. This ratio, which must lie between zero and one, is the discordance index.

The discordance set $D_{ii'}$ is thus defined as the set of evaluation criteria for which alternative *i* is worse than alternative *i'*. It reflects the idea that, beyond a certain level, bad performance on one criterion cannot be compensated for by good performance on the other criterion.

$$d_{i'i} = \max_{j \varepsilon D_{i'i}} |\hat{x}_{ji} - \hat{x}_{j'i}|$$

Thresholds supplied by the decision maker, in combination with the concordance and discordance tables, are used to establish a weak and a strong outranking relationship between each pair of alternatives. A procedure of step-by-step elimination is used to transform the weak and the strong graph representing these outranking relationships into an overall ranking of the alternatives.

For further reading see Vincke (1992) for a discussion on the better known methods. DETR (2000) also provides an overview of the theory underlying outranking methods and includes a simple numerical example using Electre I.

Key issues

An extensive literature exists on the theoretical and practical aspects of applying outranking methods. Although the Electre methods are relatively complex, they are also flexible and can be tailored to specific decision situations. The discordance relationship, for example, can be used to limit the extent to which bad performance on one criterion can be compensated by good performance on another. This is a relevant feature in many practical situations and may be particularly important within a chemical risk management context given the range of trade-offs that may need to be made in choosing a preferred risk reduction option (*e.g.* health versus environment versus economic costs).

A key disadvantage is that analysts using these methods need to be trained well in their use since interaction with the decision makers is a complicated task. The results are very sensitive to the level of the thresholds used to define the concordance and discordance relationships. Setting these levels requires complex interactions between the analyst and decision maker, which are not always transparent. In particular, there may be difficulties for others in trying to understand the role that the various thresholds have played in determining the end ranking of options and in interpreting the weights assigned to the different criteria.

In addition, the procedure used to generate the final ranking does not always result in a complete ranking of alternative options. In some cases one or more options cannot be ranked or two partial rankings are produced.

Overall, it is likely that the complexity of the method makes it less transparent and therefore less suitable for the purposes of chemical risk management than the other techniques described above.

4.3.8 Qualitative Methods

The Approach

Qualitative methods can be used if only qualitative information on scores is available or if a mixture of quantitative or qualitative scores is available. Examples of methods falling into this category are the Analytical Hierarchy Process (AHP), the Regime method, the Permutation method and the Evamix method. Qualitative information is used in the Regime method, the Permutation method and the Evamix method in a strict ordinal sense. Only the ranking of alternatives for each criterion is used, no indication is included to the extent that one alternative is better than another. The AHP method combines ordinal information with an indication of the size of the differences between alternatives. Information is collected as qualitative information but is interpreted as quantitative information.

The Analytic Hierarchy Process (AHP)

Pairwise comparisons are not only used on their own and in methods such as Electre (described above), but also provide the basis for the Analytical Hierarchy Process (AHP). AHP structures the decision problem into levels that correspond to a decision maker's understanding of the situation: goals, criteria, sub-criteria, and options. By breaking the problem into levels, the decision maker can focus on smaller sets of decisions.

The AHP is based on four assumptions [Saaty (1980) and Saaty (1996)]:

- given any pair of alternative options, the decision maker is able to provide a pairwise comparison of these options under any criterion on a linguistic scale (better, much better and so on) which can be directly linked to a ratio scale;

- when comparing any two options, the decision maker never judges one to be infinitely better than another under any criterion;

- one can formulate the decision problem as a hierarchy; and

- all criteria and options that impact a decision problem are represented in the hierarchy.

The aim of this method is to derive quantitative scores and weights from qualitative statements on the relative performance of alternatives and the relative importance of criteria obtained from comparison of all pairs of alternatives and criteria.

Using a scale such as that discussed in Section 4.3.2 above (the nine-point scale developed for pairwise comparisons undertaken as part of this method), the decision maker is asked to compare all pairs of criteria. This results, for each pair *(j,j')*, in a value of $a_{jj'}$ that expresses the difference in importance of the two criteria. Note that a_{jj} is 1 for all criteria, and that for each pair $a_{jj'} = a^{-1}_{j'j}$. This results in a pairwise comparison matrix A. Saaty then proposes that the values of $a_{j'j}$ are interpreted as relative weights calculated as follows:

$$a_{j'j} = w_j / w_{j'}$$

If the judgements supplied by the decision maker are completely consistent, one row of the comparison matrix A would be enough to produce all relative weights. Complete consistency implies that (triangular) relationships of the type $a_{13} = a_{12}a_{23}$ hold for all sets of three criteria. This is almost never the case. An approximation of the weights, therefore, needs to be generated that makes the optimal use of the (inconsistent) information available in the comparison matrix.

Saaty then proposes that the weight vector is derived as the Eigenvector of A with the largest Eigenvalue. A consistency index is calculated that represents the extent to which the judgements supplied by the decision maker are consistent with the type of triangular relationship described in the preceding paragraph. A value of this index of 0 indicates that all triangular relations hold, a value of 1 results if judgements are made at random.

Unfortunately, the number of pairwise comparisons to be made increases rapidly with the number of criteria. Saaty therefore suggests the use of a hierarchical structure of goals, sub-goals and criteria.

It should be noted, that the AHP method can be used not only to assess relative criteria weights but also to assess the performance of options through pairwise comparisons. The resulting tables of pairwise comparisons are translated to weights and scores using the Eigenvalues of these tables.

For further reading see Zahedi (1986), Golden *et al* (1989) and Shim (1989) which all provide reviews of previous applications.

Regime Method

The Regime method is also based on pairwise comparisons of alternatives. For each criterion all pairs of alternatives are compared. The best alternative receives +1, the worst –1 and both alternatives receive 0 if they are the same. These scores are then combined with quantitative information on weights attached to the criteria to determine which of the two alternatives is preferred if all criteria are taken into account simultaneously. This is straightforward if quantitative weights are available.

If only qualitative weights are available, they are interpreted as unknown quantitative weights. A set *S* is defined containing all sets of quantitative weights that conform to the qualitative priority information. In some cases, for parts of the set *S*, one alternative is preferred while for other parts the other alternative is preferred. The distribution of the weights within *S* is assumed to be uniform and, therefore, the relative sizes of the subsets of *S* can be interpreted as probabilities which can be aggregated to produce an overall ranking of the alternatives (Nijkamp *et al*, 1990).

Permutation Method

The Permutation method addresses the following question: which, of all possible rank orders of alternatives, is most in harmony with the ordinal information contained in an effects table? This is found through the following relationships:

$$\underset{p = 1,....,i!}{\text{maximize}} \quad \sum_{j=1}^{J} w_j \tau_j^p$$

subject to

$$w_1 \geq w_2 ..... \geq w_J \quad \text{plus} \quad \sum w_j = 1$$

In the case of *i* alternatives, the total number of possible permutations is equal to *i!*.[60] Each permutation can be numbered as *p* (*p=1,…,i!*). Each rank order from the permutations is then confronted with the ordinal information contained in each of the rows of the effects table. Rank correlation coefficients[61] are then used to compute the statistical correlation between the *i!* rank orders and the *j* columns of the effects table. This results in a large number of rank correlation coefficients. The weighted sums of the rank correlation coefficients are used to determine the most attractive of the *i!* permutations (Stijnen, 1995).

The Evamix Method

The Evamix method is designed to deal with an effects table containing both qualitative and quantitative criteria. The set of criteria in the effects table is divided into a set of ordinal criteria *O* and as set of quantitative criteria *Q*. For both sets, dominance criteria are calculated:

[60] Where *i!* means the factorial of *i*.

[61] Kendall’s rank correlation coefficient is used for these purposes.

$$\alpha_{i'i} = \left[\sum_{j\varepsilon O} \{ w_j \, sign\,(x_{ji} - x_{j'i}) \}^p \right]^{1/p}$$

$$\beta_{i'i} = \left[\sum_{j\varepsilon Q} \{ w_j \, (\hat{x}_{ji} - \hat{x}_{j'i}) \}^p \right]^{1/p}$$

and :

$$\mathrm{sign}(x_{ji} - x_{j'i}) = \begin{array}{l} +1 \text{ if } x_{ji} \succ x_{j'i} \\ \;\;0 \text{ if } x_{ji} = x_{j'i} \\ -1 \text{ if } x_{ji} \prec x_{j'i} \end{array}$$

The scaling factor *p* must be a positive integer. The method requires quantitative weights but can be used in combination with any of the methods dealing with ordinal priority information described in Section 3.5. A total dominance score is found by combining the indices $\bullet_{ij}$ and $\bullet_{ij}$ calculated separately for the qualitative and quantitative scores. To be able to combine $\bullet_{ii}$’ and $\bullet_{ii}$’ both indices need to be standardised. Voogd (1983) offers various procedures for this standardisation. The most straightforward standardisation divides qualitative indices by the absolute value of their sum and does the same with quantitative indices. The total dominance score is calculated as the weighted sum of the qualitative and quantitative dominance scores.

Key Issues

The Analytical Hierarchy Process (AHP) is widely used all over the world. Many applications can be found in the literature and lively discussions on its theoretical validity can also be found. Although some controversy surrounds the theoretical basis of the method, it is easy to use and produces results that match the intuitive expectations of the users. Despite its ease of use, the procedure for processing information obtained from the decision maker is far from transparent. This makes the method less suitable for situations with many stakeholders. A special group decision version of the method is popular, however, for situations where stakeholders are brought together to negotiate their positions.

Decision situations where only strictly ordinal information is available are rare. In these rare cases the Regime method and Permutation method can be used to process this information. Both methods process the ordinal information in a theoretically sound way. Both methods are, however, very complicated and far from transparent. This prevents their use in most practical applications. The Evamix method is relatively simple and was specially designed to deal with mixed information. Although the procedure is relatively transparent, interpretation of the results is ambiguous.

4.3.9 Which Method to Use

Choosing an evaluation method is in itself a multi-criteria problem, involving a trade-off between comprehensiveness, objectivity, and simplicity. Comprehensiveness is achieved if all information is presented to decision makers. Presenting a final ranking, or even only one best alternative, results in maximum simplicity. Graphic or other presentations of the information take an intermediate position. A complete ranking provides maximum simplicity, but in aggregating all information to a final ranking, priorities need to be included and a decision rule needs to be selected.

Table 4.6 characterises the multi-criteria methods described above according to the type of information required, the type of results produced, the transparency of the method, the computational effort and finally the costs of use of the method.

As can be seen from the table the type of information available determines to a large extent the methods that can be used. Most quantitative methods produce performance scores as well as a ranking. In addition to a ranking, weighted performance scores provide information on the relative performance of the alternatives. Transparency is low across a number of the methods, suggesting that such methods should not be used if many stakeholders are involved in or concerned with decision making. Computation is complex in some of the methods. Since software is generally available to support the use of the methods, this is in itself not an important issue.[62] The costs of adopting methods based on the use of value/utility functions are likely to be higher than those associated with the use of AHP and Outranking methods. These additional costs result from the involvement of an expert in the assessment procedure.

Table 4.6: Characteristics of Multi-Criteria Methods

Method	Information	Result	Transparency	Computation	Costs
Weighted summation	Quantitative	Performance Scores/ranking	High	Simple	Low
Value/utility functions	Quantitative	Performance Scores/ranking	High	Simple	High
Ideal point	Quantitative	Distance to target/ranking	Medium	Simple	Low
Graphics	Qual./quant./ Mixed	Visual Presentation	High	Simple	Low
Outranking	Quantitative	Ranking/ incomplete ranking	Low	Very complex	Medium
AHP	Qualitative	Performance Scores/ranking	Low	Complex	Medium
Regime	Qual./quant./ Mixed	Ranking/ probability	Low	Very Complex	Low
Permutation	Qualitative	Ranking	Low	Very complex	Medium
Evamix	Mixed	Ranking	Low	Simple	Low

4.4 Sources of Data

4.4.1 Overview

When assessing impacts upon ‘the environment’ (either in monetary or non-monetary terms), there must be a clear understanding of the interrelationship between say, a hazardous

62 See, for example, Buede (1998) and DETR (2000) for a review of saftware packages that are available. For more specific packages see: VISA (Belton, 1999), Expert Choice (www.expertchoice.com) DEFINITE (Janssen *et al*, 2001), Qualiflex (Stijnen, 1995).

substance and the damage it may be causing in terms of its effect on flora and fauna. Knowledge is therefore required on:

- the degree of environmental change; and
- the impact of such changes on environmental 'value'.

By way of example, consider a situation where discharges of a hazardous substance to the atmosphere have been identified as presenting environmental risks. In this example, the aim is to derive a damage cost function. A number of data related issues arise in trying to do so:

- engineering data are required to determine emissions at source;
- monitoring must be carried out at receptor points to:
  - ascertain background concentrations and data on any other residuals, and
  - establish accurately links between discharges and receptor concentrations;
- information is required on chemical or biological reactions;
- dose-response data may be required for a wide variety of receptors; and
- data is required on the values attached to the predicted environmental effects.

This example illustrates the potential complexity of the issues that may need to be addressed within any given assessment. In view of the above, a range of information sources may need to be called upon by the analyst to ensure that the analysis is based on the best available information. This information relates not only to the extent of any changes in the environment but also to the preferences (whether in money terms or through a cardinal or ordinal score) of society in relation to those changes.

4.4.2 The Range of Sources

The methodology being adopted for the SEA, together with the nature of the data available from the risk assessment, may affect the types of data sources that need to be turned to in the analysis. Potential types and sources of data include:

- face-to-face, telephone and postal interviews;
- activity and market data to allow modelling of both revealed and observed preferences (*i.e.* as part of economic valuation);
- dose-response relationships;
- environmental quality statistics;
- academic and other research literature addressing assessment issues; and
- previous regulatory appraisals and work being carried out by government and international organisations.

In the context of the economic valuation of environmental effects, it is important to note that in recent years initiatives have been developed which are aimed at the sharing of data developed through past valuation exercises. The key initiative in this regard is that of Environment Canada with the development of the EVRI database of valuation studies. This database covers not only valuation studies undertaken in North America, but also those identified to date in Europe and in Australia and New Zealand. It is available for use by others although there is a subscription fee for access.

4.5 The ZOZ Case Study

4.5.1 Introduction

As will be recalled from earlier discussions of the case study, four different uses of ZOZ have been identified as giving rise to unacceptable aquatic risks at both the site-specific and regional levels (with the predicted environmental concentrations exceeding what experts have agreed as being the predicted no effects concentrations). There are point source problems associated with effluent emissions arising from ZOZ production, manufacture of chemical intermediaries, and emissions from textile cleaning and dying operations. More diffuse emissions arising from the use of ZOZ-based industrial cleaning products have also been identified as giving rise to unacceptable risks (both to the aquatic environment and to man via the environment).

A range of different risk reduction options have been identified with the aim of controlling these emissions (with the training-based options discussed in Section 3 not expected to generate any reductions in emissions to the aquatic environment). These include:

- a voluntary reduction in the use of ZOZ (either total reduction or reduced concentrations);
- the use of product labelling, with the aim of reducing use in the textiles industry and in industrial cleaning products;
- product taxes aimed at increasing the price of ZOZ relative to non-ZOZ based cleaning products;
- marketing and use restrictions involving a total ban on the use of ZOZ; and
- command and control measures which would reduce emissions from point sources.

Data collected during the survey of industry (see Section 2.8 of this Part) were combined with the models developed as part of the risk assessment to predict the change in predicted environmental concentrations that would arise from the introduction of the above measures. The results of this modelling work are presented in Table 4.7. As can be seen from the table, the various options are not all expected to be equally effective in reducing risks to the aquatic environment (and hence man *via* the environment) to a level where the predicted environmental concentrations no longer pose a problem (with this being at a concentration below 1 Φg/litre).

The next question that was addressed specific to the environmental risks concerned defining in more detail what the actual environmental benefits would be in reducing the risks; in other words, analysts had to move from predicted risks based on aquatic toxicity data (*e.g.* risks to *Daphnia Magna* and test fish species) to fisheries and aquatic ecosystems more generally. This was a complicated task as dose-response (cause-effect) data, although they exist, are fairly limited for ZOZ.

The conclusions that were reached from a review of the available literature and ecotoxicological data are as follows:

– the concentrations of ZOZ (10 Φg/l) currently found at certain sites (which have more than one facility emitting ZOZ in their effluent) are predicted as leading to fish mortalities and reproductive problems in key commercial and recreational fish species; as a result, biomass levels for commercial and recreational fisheries are depressed, leading to reductions in catch levels for these sites;

Table 4.7: Predicted Reductions in Environmental Risks

Option	Target Sectors	Facility-Specific Emissions	Diffuse Emissions	Comments
Voluntary Reduction	ZOZ production, chemical intermediaries, textile cleaning and dying, and industrial cleaning products	PEC/PNEC reduced at most larger facilities but unknown for smaller facilities	PEC/PNEC of around 1.6 for regional concentrations	About 70% reduction in ZOZ expected across sectors based on trade associations data.
Product Labelling	Textile cleaning and dying and industrial cleaning products	PEC/PNEC >1 for emissions from most facilities	PEC/PNEC > 1, ranging between 1 and 4	Not expected to be effective.
Product Taxes	Industrial cleaning products	No impact	PEC/PNEC > 1, ranging between 1 and 3	Would need to be combined with another measure.
Marketing and Use	Chemical intermediaries, textiles cleaning and dying and industrial cleaning products	PEC/PNEC <1 for all facilities	PEC/PNEC <1 for regional concentrations	Most effective option; could be adopted in part and added to other measures to minimise impact on some sectors.
Command and Control	ZOZ production, chemical intermediaries, textiles cleaning and dying	PEC/PNEC <1 for all production and chemical intermediaries facilities; PEC/PNEC >1 predicted for smaller textiles facilities	PEC/PNEC of 1.5 for regional concentrations	Is very effective for the target sectors and the larger facilities; not possible to implement in small facilities.

– concentrations associated with diffuse emissions (between 1 and 7 Φg/l) may be leading to some mortalities and reduced reproduction levels in certain sensitive species; impacts on commercial and recreational fisheries are unknown but the aquatic ecology in general may be affected.

The next decision concerned how to approach these mixed findings in assessing the marginal benefits arising from each of the options. Again, approaches involving the use of cost-effectiveness analysis and monetary valuation were examined. In this case a multi-criteria analysis was not undertaken because there was pressure from decision makers for a more economics based approach; a detailed discussion of how the weighted summation method might be applied to assessing environmental (and other impacts) is provided, however, in Section 4 of Part 2.

4.5.2 Cost-Effectiveness Analysis

A number of problems arise in trying to examine the cost-effectiveness of the different options. First, there are multiple environmental impacts of concern (which also affect impacts on man *via* the environment), with these pertaining to commercial and recreational fisheries and, perhaps more importantly, the aquatic ecosystem as a whole. At a simple level, it is obvious from Table 4.7, that not all options would achieve pre-specified targets of reducing risks to a level where the PEC/PNEC ratios are below 1.0 (*i.e.* predicted environmental concentrations below 1 Φg/l).

The only option that would meet the targets is marketing and use restrictions, as highlighted in Table 4.8 which provides an indication of the cost-effectiveness of each of the main options. The command and control option is the next best performing choice. If combined with one of the other options which reduces the use of ZOZ in industrial cleaning products (which could be any of those listed in the table), it may provide the needed level of reduction. (See also Section 2 of Part 4 for further discussion on the costs and benefits under each option).

Table 4.8: Cost-effectiveness of Risk Reduction Options

Option	**Sites Meeting PEC/PNEC <1 (<1 Φg/l)**	**Regional Concentrations (Diffuse Emissions)**	**Present Value Cost ($ millions)***
Voluntary Agreement	7 out of 20	1.5 to 4 Φg/l	$58.7
Labelling	0 out of 20	3 to 7 Φg/l	$6.1
Product Taxes	7 out of 20	1.5 to 4 Φg/l	$36
Marketing and Use	20 out of 20	< 1 Φg/l	$116
Command and Control	14 out of 20	1.5 to 3 Φg/l	$54.6
* Discounted at 5%			

As can be seen from the table, even when combined with other measures, the command and control option fails to be able to achieve target PEC/PNEC ratios of less than 1.0, with marketing and use restrictions remaining the most cost-effective. Only if the combined command and control plus voluntary agreement option could be made more effective, with reduced costs realised through greater flexibility, could this option challenge market and use restrictions with regard to overall cost-effectiveness.

It must be recognised though that the above assessment does not consider the indirect effects on the general public that may result from concentrations of ZOZ in the environment. In order to do so, this assessment would have to be further expanded to consider exposure routes, frequency of exposure, the number of individuals likely to be exposed and the associated health effects for the levels of exposure. This is also true with regard to the monetary valuation exercise discussed below (see also Section 3.6 on the assessment of human health).

4.5.3 Monetary Valuation

One of the key questions that the analysts had to address was the issue of site versus regional aquatic impacts. Site-level impacts are more readily predicted owing to a better ability to link changes in emissions from specific facilities to concentrations in the environment to dose-response relationships and to the characteristics of specific fisheries. However, the regional level impacts on general aquatic ecology are likely to be the more important impacts (in environmental and economic terms). In addition, there are obviously interactions between site and regional concentrations (particularly as some facility-specific emissions will be to locations not covered by the 20 main sites identified as having elevated ZOZ concentrations from background monitoring data).

Following a review of the environmental economics literature and discussion with a range of different fisheries experts and economists, the decision was taken to approach the valuation exercise as follows:

- to apply an effects-on-productivity based approach to estimate the economic damages occurring to commercial fisheries at the site level;

- to use the results of an RUM-based travel cost modelling exercise to value the impacts on changes in catch levels for recreational fisheries, again at the site level; and

- if possible to draw upon the wider valuation literature to place a monetary value on the wider ecological impacts.

After several months of work involving the analysts who prepared the risk assessment, fisheries experts and economists, the valuation work was completed.

Based on the model used as part of the risk assessment, predictions were made of the number of locations where concentrations of ZOZ exceeded the 1 Φg/l level (the PNEC). For the baseline situation (pre-risk reduction situation), exceedance of the PNEC was estimated as occurring at 20 sites, with the number of sites achieving concentrations below the PNEC following introduction of the various options as indicated in Table 4.8.

Dose-response data were then linked to commercial fisheries productivity, enabling the impacts of changes in ZOZ concentrations on catch rates to be modelled for key species. The changes in catch rate across the main commercial species affected were then linked to the economic value of the species to determine the benefits of reductions in ZOZ emissions. The estimated per annum benefits to commercial fisheries predicted for each option are presented in Table 4.9 (together with the present value of these benefits over the 15-year analysis period).

The dose-response data and modelling work also predicted changes in catch rates for the main recreational fisheries at the 20 main sites of concern. In this case, the analysts were fortunate to be able to draw upon a valuation exercise that was already underway. This exercise involved the application of a RUM travel cost model to predict recreational anglers' demand for fisheries with different characteristics, of which catch rates was a key variable. From this, they were able to link anglers' increased willingness to pay per visit to increases in catch rates (taking into account location, other site characteristics, congestion, *etc.*). These data were then linked with data on trip rates to predict the expected economic benefits at each site. Again, the results are presented in Table 4.9.

Table 4.9: Monetary Valuation of Fisheries Benefits ($ millions)*					
Option	Commercial Fisheries		Recreational Fisheries		Total Present Value Benefits
	Per Annum	Present Value**	Per Annum	Present Value**	
Voluntary Agreement	$2.0	$20.7	$0.9	$9.3	$30.0
Labelling	$0.28	$2.9	$0.15	$1.6	$4.5
Product Taxes	$2.0	$20.7	$0.9	$9.9	$30.6
Marketing and Use	$8.4	$87.2	$4.1	$42.9	$130.1
Command and Control	$5.2	$54.0	$3.0	$31.1	$85.3
* Note that benefits are not proportional to number of sites improving as the benefits arising for individual sites vary by location, fish species, numbers of visiting anglers, *etc.* ** Discounted at 5%					

After a review of the environmental economics literature, it was concluded that none of the existing studies could be applied in a reliable manner to generate monetary values for the impacts of elevated ZOZ concentrations on aquatic ecology. A contingent valuation or attribute-based stated preferences exercise would have to be undertaken which specifically addressed the types of impacts associated with ZOZ and its risk management. Given that there was neither the time nor the resources available to carry out such an exercise, the decision was taken to treat such effects in qualitative terms within the final analysis; however, it was stressed that the greater potential significance of these effects in relation to the impacts on fisheries should be recognised.

4.5.4 The Issue of Substitutes

As noted earlier, concern has been raised that the switch to substitute chemicals (rather than processes in this case) may give rise to new or similar levels of environmental risk. In particular, this concern has been raised for the textiles sector, which it is assumed moved to ZOZ as a result of possible environmental risks associated with the current main commercial substitute.

Unfortunately, there are inadequate dose-response and monitoring data to allow the risks associated with use of the substitute to be assessed in a manner similar to that undertaken for ZOZ. Generation of such data would delay decision making on ZOZ for a considerable period and this is not likely to be acceptable to decision makers and other stakeholders. The analysts concluded that because of the potential significance of the environmental risks (aquatic and terrestrial) that have been linked to the substitute chemical, the issue must be noted in the final assessment. Indeed, any voluntary agreement or other regulations should include a specific comment warning against a shift back to the use of this substitute.

5. EQUITY ANALYSIS

5.1 Introduction

Policy makers generally place great importance on equity and distributional issues,[63] wishing to consider the fairness of a proposed option in terms of the incidence and distribution of benefits and costs, as well as net benefits. Such concerns are illustrated by the inclusion of specific requirements for the examination of these issues within SEAs prepared in Canada and the US (see for example Ontario Ministry of Environment & Energy (1996) and US EPA (1999) for further details of the specific requirements of these countries).

The issues of concern will generally relate to a range of considerations, for example:

- impacts on vulnerable or other particular groups within society;
- impacts on income distribution;
- impacts on industry, taking into account size, age of plant, *etc.*;
- creation of barriers to entry into a market sector;
- regional employment effects; and
- impacts on price inflation across the economy.

Several of these issues have already been addressed in Section 2 of this Part and, to avoid duplication, are not considered further here. Instead, this section focuses on the assessment of the equity implications to affected populations associated with a proposed risk reduction option (where this includes ‘doing nothing’).

5.2 Equity Assumptions within the SEA Methodologies

Before discussing how equity analyses can be undertaken as part of a SEA, it is important to recognise the equity assumptions inherent within the different methodological frameworks. These vary across the key frameworks provided by CEA, CBA and MCA based techniques.

It is useful to start with CEA and analyses which are based on the comparative assessment of the cost per life saved (or fatality avoided) implied by alternative risk reduction options. Estimation of this figure implicitly assumes that all lives saved are considered equal regardless of the age, health state and other circumstances of the individual. Similarly, a CEA based on calculation of the cost per life year gained assumes all life-years gained are equal regardless of the circumstances.

[63] Where equity relates to ‘fairness’ and distribution to the share of costs or benefits to sub-populations within society associated with a risk reduction option.

Similarly, the types of MCA-based cost-utility analysis which have been used to date (as discussed in Section 2 of this Part) contain an equity assumption that a quality adjusted life-year gained is equivalent regardless of the age at which the gain is achieved.

In contrast CBA, owing to its emphasis on individuals' willingness to pay for particular outcome, includes an implicit assumption that the preferences of the wealthy count more than the preferences of the poor. This follows because willingness to pay is limited by the ability to pay, and thereby programmes aimed at conditions of interest to the wealthy can perform better on willingness to pay measures than programmes aimed at conditions of interest to the poor (although of course distributional weights can be used within CBAs to address this issue if desired).

Some consider the emphasis on ability to pay to be a flaw of cost-benefit analysis (Richardson, 1991), while others argue that it is simply a matter of accepting the current distribution of income and maximising welfare given that distribution (Johannesson & Jonsson, 1991). For example, an alternative formulation could be adopted which argues that CBA sets the value of, say, $1 equal across all individuals (making the assumptions more comparable to the types of approaches used in multi-criteria analysis).

As a rule, CBA is based on estimation of the net benefits of a proposed action regardless of the groups to whom benefits and costs accrue. There are three main reasons for using this net benefit criterion within socio-economic analysis:

- options that maximise net benefits are likely to benefit more people than would less efficient options;

- surpluses can be used to compensate losers (at least hypothetically); and

- distributional objectives can be achieved more cost-effectively by fiscal instruments than by individual actions.

In reality, fiscal policies often fail to produce an equitable distribution of income and losers are rarely compensated by the winners. As a result, governments are usually interested in equity implications as well as in the efficiency of a proposed option. For this reason, the equity analyses performed by economists are often focused at providing an indication of the distribution of costs and benefits across different stakeholder groups within the economy. This type of analysis does not make assumption as to the superiority of any particular distribution of impacts, it merely reports on the pattern of that distribution for each of the options under consideration.

5.3 The Form of Equity Analysis

Within any given SEA, the form that an equity analysis takes may obviously vary depending on the overarching methodological framework that has been adopted for the analysis. However, the choice of form is also likely to be limited.

Essentially, equity analyses can have two aspects: a descriptive and a normative component. The description of effects indicates the distribution of impacts on selected community groups, while the normative component shows how this distribution of (positive and negative) costs and (positive and negative) benefits could affect decisions.

In order to describe the distribution of effects, the populations that will be affected (where these effects are significant enough to matter) by the decision must be identified and the manner in which costs and benefits are shifted between groups determined. The shifting of impacts between groups matters because final impacts may differ from initial ones. Such shifts may be vertical in nature, with the types of issues considered in this case including:

– the distribution of surpluses between producers and consumers, which will depend on changes in prices;

– transfer payments that affect the distribution of costs and benefits although they do not affect the net benefits of a proposed option; and

– secondary benefits that may have substantial distributional effects, between sectors and geographic areas.

The last of the above bullet points touches on the other type of equity shift that may occur - a horizontal shift. Horizontal issues are related to impacts that may vary across gender, race, age, geographic location, *etc.* The discussions presented in the previous sections (3 and 4) highlight the fact that such horizontal issues may be at the fore of the equity issues arising as part of chemical risk management.

Moving to the normative component, incorporating distributional effects into aggregate measures of option performance requires that there are weights reflecting the importance which should be placed on impacts incurred by different sections of society. Within CBAs this involves converting individual WTP values into utility values through the use of weights. For example, instead of treating each money unit of benefit and cost as being equal, impacts on low income groups are given greater weight than those on higher income groups. This type of approach results in a modified utility function, where the marginal utility of income U_y becomes:

$$U_y = Y^e$$

where Y is the level of income and e is a parameter of the utility function.

For example, if e is unity, the weight attached to marginal income is inversely proportional to income. By adopting this type of approach, the analysis provides a measure of net social value based on equity as well as efficiency (Little & Mirrlees, 1974).

However, there are several objections to the above approach, starting with the fact that there is no generally acceptable set of weights available. Although marginal tax rates have been suggested as providing the basis for specifying weights, they cannot be argued as having been designed so as to maximise utility across all individuals. In addition, it is unclear what the meaning of a weighted net present value (NPV) actually is (as opposed to an unweighted NPV that reflects economic efficiency). The inclusion of weights into estimates of NPV could also result in an inefficient policy being adopted while an efficient one is rejected. Moreover, a policy with a weighted NPV that is greater than zero may still have significant impacts on some segments of society (Ableson, 1996). Furthermore, in economic theory terms, if the policy is deemed to be efficient, then it could be argued that a large number of efficient policies will spread benefits sufficiently widely that (eventually) all of society will gain.

A related concern is that of how to take inter-generational equity issues into account, where these relate to impacts which are expected to exceed 25 to 30 years. The manner in which this concern is managed is related to the use of discounting procedures within SEAs. For example, it may be the case that the sub-population at risk is the next generation (or later);[64] clearly the impact of discounting is to then effectively weight such impacts less in the analysis. If future generations are

[64] This may be especially true with environmental or health impacts which could have long latency periods before any effects are noticeable.

believed to be at risk, then there may need to be an alteration to the discounting process in order to take this into account.[65]

For the above reasons, the direct incorporation of utility weights into economic appraisals is rarely undertaken. There are, however, examples of the use of 'distributional weighting' systems as part of CBAs concerning land use issues and project and policy proposals in developing countries, where equity issues can be of prime importance. For most developed countries, however, the use of this type of approach is not as common, with such information being provided through supplemental analyses.

The lack of agreed weights for use in CBAs is also an issue that affects CEAs and cost-utility analyses. One of the reasons that such analyses make no assumption concerning the relative cost per life saved that should be attached to different population groups is the lack of agreement on the relative importance which should be assigned to protecting different population groups within society. As the limited research on the issue of weights is inconclusive, the advice given is to assume equal weights for all lives, life-years or QALYs. It is recommended practice for analysts to highlight the fact that they have used equal weights in the analysis, and to make it easy for users of the studies to substitute different weights. However, it would appear that a more descriptive approach is generally suggested.

The problem of deriving relative importance weights is obviously also relevant to the inclusion of equity issues within a wider MCA-based assessment. However, it should present no greater difficulties than the derivation of relative importance weights for other effects. Thus, the difficulty involved in specifying weights will depend on both the level of detail at which impacts are specified and the complexity of the approach taken to the assessment. Given that the weights in MCA are usually context specific and designed by the analyst undertaking the SEA, then MCA is open to similar criticism as CBA concerning the assignment of weights.

5.4 Steps in an Equity Analysis

5.4.1 Overview

The US EPA has developed guidance as to how equity considerations can be incorporated into regulatory policy appraisals (US EPA, 1999). In effect, this guidance requires the assessment of any disproportionately high costs and adverse human health or environmental effects on minority or low income populations. This includes consideration of the cumulative effects of a proposed action as well as other environmental stresses that may be affecting a community.

Within the context of risk management, the EPA guidelines stress the importance of ensuring that risk assessments are conducted so as to determine exposure pathways and potential effects on the communities of concern. This information is combined with geographic and demographic data to determine whether distributional issues arise either directly or indirectly through the use of land and water resources at a level above the general public's. Standard socio-economic models are then used to examine criteria such as employment, income levels, housing, *etc.* Impacts on other sub-populations, such as workers, the elderly, small businesses, small government entities, regions, *etc.* are also evaluated if they are especially relevant to a planned action.

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[65] It may be the case that no discounting is undertaken or even some form of 'inverse' discounting to weight impacts in the future greater than the present within the analysis (for more detail see Section 2 of Part 4).

The analyses may involve the partitioning out of information on the positive and negative costs and positive and negative benefits by sub-population within an overarching appraisal, or by supplemental analyses conducted for some or all of the options available to the decision maker. One of the options available is the development of scoring and weighting systems. The aim of this is to use such systems to combine preliminary information on potential economic impacts with information on other potential impacts. The analysis can then be used to both define decision criteria for additional targeted analyses or studies and to provide supplementary information to decision makers.

The remainder of this section draws on the EPA's guidelines (US EPA, 1999) to set out a general approach to undertaking such analyses.

5.4.2 Steps in an Equity Appraisal

Figure 5.1 sets out suggested steps to be followed in undertaking an equity appraisal. The approach is basically to 'determine who is affected and by how much'.

Equity Scoping

The first step (which may be undertaken at any stage of developing the risk reduction option)[66] is to determine (at a high level) which type of populations are most likely to be affected. Table 5.1 provides a starting point for this. Once it is understood which sub-populations may be affected, it is then necessary to determine how they may be affected; negative impacts upon small entities, low income groups, minority groups and children will be important considerations.

[66] It is recommended, however, that this process is undertaken as early as possible in order to use resources efficiently.

Figure 5.1: An Approach for Appraising Equity

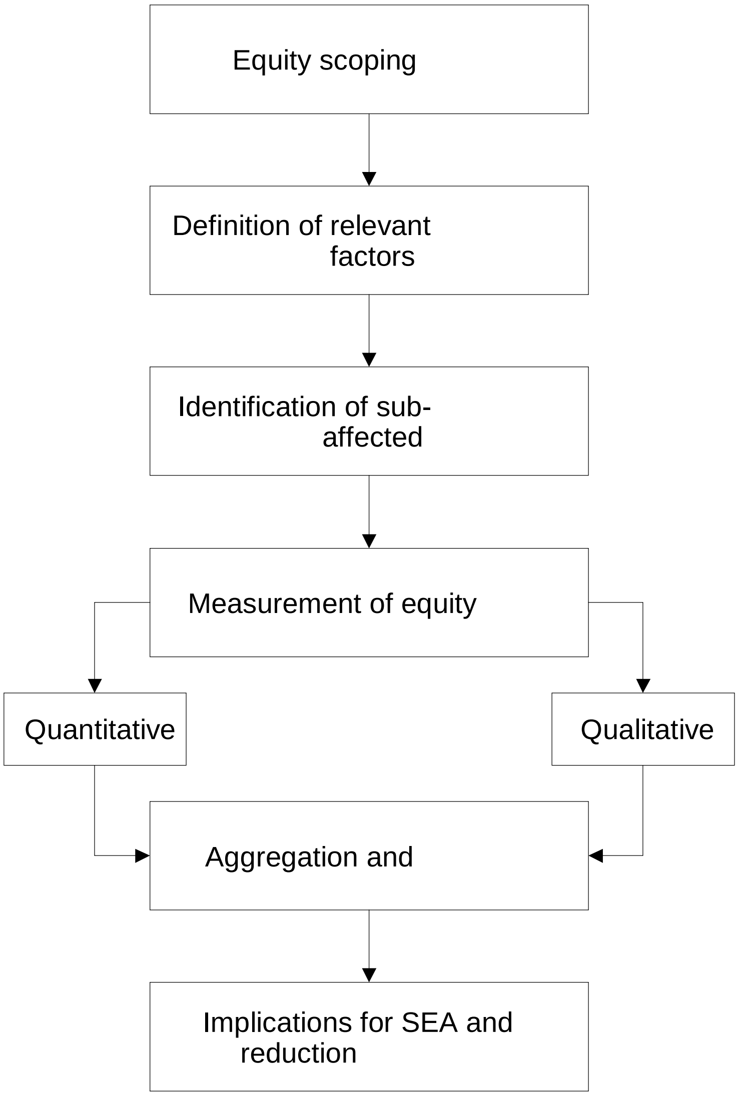


Table 5.1: Examples of Affected Sub-Populations*		
Equity Dimension	**Affected Population**	**Notes**
Minority status	Individuals or households	Usually considered a minority with less than 50% of affected population
Income	Individuals or households	
Age	Individuals or households	
Gender	Individuals	
Time	Individuals or households	Considers impact on current and future generations
Physical sensitivity	Individuals or households	
Education	Individuals or households	
Children	Individuals or households	Children may be a particularly sensitive group reliant on others for their well-being
*based on US EPA (1999)		

Even though this table sets out the types of equity dimensions to be considered, there is a clear problem in determining what issues merit further consideration. The decision should not be left to a process whereby those that 'shout loudest' (via lobbying or political pressure, for example) are given greatest priority. It may be the case that those in society without strong representation (*i.e.* the less well off, the sick, the infirm) are most affected by a change in regulation.

It should, therefore, be the responsibility of the analyst to set out the key sensitive sub-populations that may unduly suffer from the implementation of a particular risk reduction option. This process will then aid decision makers in determining whether the full SEA should examine equity and distributional impacts. If it is deemed that there will be few equity impacts, then the matter need not be pursued further. However, if the scoping process determines that a fuller appraisal of the equity impacts may be desirable then this can become part of the full SEA.

Relevant Distributional Factors and Sub-Populations Affected

The scoping process will have indicated the types of equity-related impacts to be expected. This stage decides which factors will be used to determine the 'net equity effect' on those populations of concern. It is crucial at this stage in the analysis to agree to the relevant measures of equity to be used in the analysis. For example consideration should be given to:

- what defines a low income group?
- what defines an ethnic minority?
- what defines 'poor' or 'unhealthy'? and
- who defines the importance of the equity measures?

The final question provides a crucial point as to the selection of which equity measures to use and the relative importance of these measures. For example, are impacts on the poor more important than impacts on ethnic minorities or those suffering from a particular health effect? As with all decisions, there may be significant trade-offs between stakeholders. It may be the case, however, that at risk groups are treated as one sub-population (such as a low income ethnic minority, or children with asthma, and so on). This case will result in a more straightforward analysis with no need for trade-offs between the affected groups.

Measurement of Impacts

The most reliable (and cost effective) approach is likely to involve determining the change in different end points (such as costs of goods, emissions, risk) and then the relationship between this end point and the sub-population at risk (the number of products purchased, vicinity to emissions, susceptibility). This may be achieved via consultation, surveys or may even be assumptions based on the findings of the core elements of the SEA.

Once it is known what end points are of concern and for what sub-populations, the next major questions is 'how to measure' the equity implications. It may be the case that the various sub-populations will simply incur extra costs in purchasing from products or services, with these being readily valued in monetary terms. However, it is likely that valuation of other equity effects will be more problematic. Although the techniques exist to enable the valuation of such effects, the resource implications of deriving such values may not be worth the additional expense.

Simpler approaches can be adopted instead, ranging from the use of scoring and weighting systems to the use of 'trend analysis' which indicates in general terms the likely direction of change. An example of a simple trend analysis based approach is given in Table 5.2, which focuses on the impact upon one sub-population of a risk reduction option. Assuming that these end impacts are of equal importance (an assumption which is unlikely to hold in reality), the table suggests that this sub-population would be worse off with the introduction of the proposed option.

Table 5.2: Example of a Simple Approach to Weighing up Impacts

Type of Impact	Positive	Negative
Health		--
Prices of goods or services	+	
Overall income	+	
Access to education and services		-
Mobility		-
Total	**++**	**----**

Aggregation and Discounting

Once the impacts have been collated for each sub-population at risk, the next stage is to determine whether to aggregate impacts and to adjust for time. The key question for the analyst to consider is whether it is appropriate to aggregate the total impact across the sub-populations at risk or whether it would be more appropriate to present the impact on each group separately. The best way forward (in the absence of relative importance weights) is probably to present information on the impacts on each sub-population in whatever form (or combination of forms) in which it is best measured. This leaves it the responsibility of decision makers to weigh up the relative impacts upon each group.

The issue of time can complicate matters, particularly if major negative impacts occur far into the future. It may therefore be appropriate (as stated above) for discounting to be omitted or for the rates to be altered to give greater relative weight on future effects. The reliance on absolute figures broken down by years, rather than discounted ones, may present decision makers with a clearer picture about the equity and distributional impacts.

5.5 Implications for SEA and Risk Reduction

Where a proposed risk reduction option would impose considerable social costs on certain sections of society, then there is a strong case for the equity implications of the option to be examined in greater detail. If a SEA finds that, following an equity analysis, a particular sub-population(s) will be unacceptably worse off, the potential for mitigating these effects through adjustment of the proposed option should be considered. Potential forms of adjustment include:

- derogations;
- compensation;
- additional services provided by central or local governments (such as health care to treat specific ailments); and/or
- a re-design of the risk reduction option itself.

The aim of conducting equity analyses is essentially to illustrate where specific sections of society bear a disproportionate share of the costs. The goal is to identify options that are both efficient and equitable in the way that they operate in achieving the desired reductions in risk.

5.6 The ZOZ Case Study

To illustrate the above discussion more fully, it is useful to consider the manner in which equity considerations might be taken into account in decision making concerning our hypothetical chemical ZOZ.

Following the steps set out above, the analysts preparing the SEA identified the key vertical and horizontal equity issues arising from risk management of ZOZ. These are summarised in Table 5.3. Note that in preparing this analysis, they considered both business/competition distributional issues and wider equity concerns.

Once the potential concerns had been identified, the next step was to determine how they should be measured and thus assessed. The analysts were concerned that the approach they adopted should be one which was readily understood and did not restrict them to having to adopt a CEA, CBA or MCA based approach. As a result, it was decided that a simple ‘trend analysis’ would be undertaken to provide an indication of the likely significance of any distributional and equity effects.

Table 5.3: Summary of Distributional and Equity Considerations

Type of Issue		Description	Relevant Options
Competition	Global trade	Increased costs to textiles industry and potentially the chemical intermediaries sectors have been argued as reducing global competitiveness.	Marketing and use restrictions and command and control
	Employment	Not expected to be significant except for those involved in the production of ZOZ.	Marketing and use restrictions
	Growth	No impacts expected on economic growth in general; some effects may arise with regard to growth of speciality chemicals market but not considered significant.	Voluntary agreement, marketing and use restrictions and command and control
Vertical Equity	Shift in surpluses throughout chain of trade	No significant effects expected; greatest shifts likely to arise under a tax with shift of burden onto retailers and consumers rather than producers.	Product tax
	Transfer payments	No significant effects likely except potentially under tax.	Product tax
	Indirect effects	Reductions in the quantity of ZOZ produced may increase costs to those industry sectors whose use does not pose risks to the environment.	Voluntary agreement, marketing and use restrictions
Horizontal Equity	Workers	Health risks to workers are only directly reduced by options involving training and equipment or enforced reductions in use of ZOZ - not really an equity issue.	Training and equipment, marketing and use restrictions
	Externally affected groups	Commercial fisheries and recreational anglers benefit to varying degrees under the different options; concern that other groups/activities will be affected with shift to substitutes in textiles sector. Health-related benefits may accrue to the general public with reductions in environmental burdens of ZOZ.	All options except training and equipment
	Gender and age	No specific gender or age related effects identified.	None
	Minorities	No significant minority related concerns identified; potential issue for commercial fisheries at two sites.	None
	Time	Impacts of continued ZOZ use could affect future generations with regard to fisheries effects.	All options except training and equipment
	Geographic	'Hot spots' of high levels of ZOZ have been identified causing localised impacts on fisheries, but more widespread regional concerns also exist with current concentrations. Activities facing regulation are widespread and centred in a particular location (*e.g.* ZOZ chemical intermediaries production occurs in several locations).	All options except training and equipment

The results of the trend analysis are presented in Table 5.4. Plus signs indicate a positive effect, while negative signs indicate a detrimental effect. Up to three plus or minus signs could be awarded to highlight small, moderate and large impacts. Note that those issues against which no significant impacts are expected (see Table 5.3) are not included in this assessment.

From a review of the 'trend' assigned to each of the options, marketing and use restrictions were considered to provide a reasonable balance between gains and losses in terms of distributional and equity effects. Given that some potential negative impacts were identified, however, it was also concluded that there may be merit in examining adjustments (particularly in the way in which measures were implemented) to both this and other options to reduce the potential impacts. The possibility of combining command and control with another option to provide an effective risk reduction strategy giving rise to lower distributional and equity concerns was also identified.

Table 5.4: Trend Analysis of Distributional and Equity Effects for ZOZ Risk Reduction Options

Effect	**Voluntary Agreement**	**Product Labelling**	**Training & Equipment**	**Product Tax**	**Marketing & Use**	**Command & Control**
Global Trade					--	--
Employment	(?)				-	
Growth	-				--	-
Surpluses				+/-		
Transfer Payments				-		
Indirect Effects	-				-	
Workers	++		++		+++	
External Groups	+	+		+	+++	++
Minorities	(?)				+	+
Time	++	+		+	+++	++
Geographic Effects	+	+		+	+++	++
Overall	++/-	+	++ (but limited)	+/-	+++/--	++/-

6. REFERENCES

Ableson P (1996): Project Appraisal and Valuation for the Environment, London: Macmillan Press Limited.

Adamowicz WL (1994): *Habit Formation and Variety Seeking in a Discrete Choice Model of Recreation Demand*, Journal of Agricultural and Resource Economics, Vol.19, No.1, pp.19-31.

Adamowicz WL, Louviere J & Williams M (1994): *Combining Stated and Revealed Preference Methods for Valuing Environmental Amenities,* Journal of Environmental Economics and Management, Vol.26, pp.271-292.

Adamowicz WL et al (1998a): Stated Preference Approaches for Measuring Passive Use Values: Choice Experiments and Contingent Valuation, American Journal of Agricultural Economics, Vol.80, (1 February), pp. 64-75.

Adamowicz WL *et al* (1998b): Introduction to Attribute-Based Stated Choice Methods, Final Report to Resource Valuation Branch, Damage Assessment Center, NOAA, Washington DC: U.S. Department of Commerce, January, pp.44.

Armstrong H & Taylor J (1985): Regional Economics and Policy, London: Phillip Allen.

Arnold F (1995): Economic Analysis of Environment Policy and Regulation, New York: John Wiley & Sons.

Arrow KJ *et al* (1993): *Report of the National Oceanic and Atmospheric Administration (NOAA) Panel on Contingent Valuation*, (also appears in the Federal Register, Vol.58, No.10, Friday, 15 January 1993, pp.4602-4614.

Ball DJ *et al* (1998): The Optimisation of Consumer Safety, London: Middlesex University.

Bateman IJ (1995): Benefit Transfers and the Water Environment, paper prepared for the UK Environment Agency, Norwich, UK: University of East Anglia.

Bateman IJ & Willis KG, eds (1999): Valuing Environmental Preferences, Oxford: University Press.

Beinat E (1997): Value Functions for Environmental Management, Dordrecht: Kluwer Academic Publishers.

Belton V (1999): Visual Interactive Sensitivity Analysis for Multicriteria Decision Making (VISA*)*, Glasgow: Visual Thinking International.

Bergstrom TC (1982): *When is a Man's Life Worth More than His Human Capital?* in Jones-Lee, MW (ed), The Value of Life and Safety, Amsterdam, pp.3-26.

Bertin J (1981): Graphics and Graphic Information Processing, Berlin: Walter de Gruyter.

Bielza C & Shenoy P (1999): A Comparison of Graphical Techniques for Asymmetric Decision Problems, Management Science, Vol. 45, No. 11.

Black P (1981): *Injection Leakages, Trade Repercussions and the Regional Economic* Multiplier, Scottish Journal of Political Economy.

Bleichrodt H, Wakker P & Johannesson M (1997): *Characterizing QALYs by Risk Neutrality*, Journal of Risk and Uncertainty, Vol.15, pp.107-114.

Bockstael N, Hanemann WM & Kling CL (1987): *Estimating the Value of Water Quality Improvements in a Recreational Demand Framework*, Water Resources Research, Vol.23, Nol.5, pp.951-960.

Boxall PC and Adamowicz WL (1999): Understanding Heterogeneous Preferences in Random Utility Models: The Use of Latent Class Analysis, Department of Rural Economy, University of Alberta, Staff Paper 99-02.

Boxall PC and Macnab B (2000): Exploring the Preferences of Wildlife Recreationists for Features of Boreal Forest Management: A Choice Experiment Approach, Canadian Journal of Forestry Research, Vol.30, pp.1931-1941.

Brunekreef B (1997): *Air Pollution and Life Expectancy: Is There a Relation?*, Occupational Environmental Medicine, Vol.54, pp.731-734.

Bucht (1998): Some Characteristics of Chemicals Risk Management at the International Level, unpublished paper, Sweden: KEMI.

Chestnut and Rowe (1986): *Oxidants and Asthmatics in Los Angeles: A Benefits Analysis*, Prepared by Energy and Resource Consultants Inc, EPA-230-09-86-018, Washingon DC: US Environmental Protection Agency.

Christainsen G & Tietenberg T (1985): *Distributional and Macroeconomic Aspects of Environmental Policy* in Kneese A & Sweeney J (eds): Handbook of Natural Resource and Energy Economics, Amsterdam: Elsevier Science Publishers, pp.372-3.

Costanza R et al (1997): The Value of the World's Ecosystem Services and Natural Capital, Nature, Vol.387, pp.253-259.

Cropper ML *et al* (1988): *On the Choice of Functional Form for Hedonic Price Functions*, The Review of Economics and Statistics, Vol.70, No.4, pp.668-675.

Dean J (1992): *Trade and the Environment: A Survey of the Literature* in Low P (ed.) International Trade and the Environment, World Bank Discussion Paper, Washington DC: World Bank, pp.15-28.

Department of Health (1999): Economic Appraisal of the Health Effects of Air Pollution, London: The Stationary Office.

Desvousges WH *et al* (1998): Environmental Policy analysis with Limited Information: Principles and Applications of the Transfer Method, Northampton, Massachusetts: Edward Elgar.

Desvousges WH *et al* (1997): Valuing Stated Preferences for Health Benefits of Improved Air Quality: Results of a Pilot Study, TER Working Paper No. T-9702, {www.ter.com/techpap.shtml}.

DETR (2000): Multi-Criteria Analysis: A Manual, Department of the Environment Transport and the Regions: London.

Dolan P (2000): *The Measurement of Health-Related Quality of Life for Use in Resource Allocation Decisions in Health Care*, pp. 1724-1760 in Handbook of Health Economics, Volume 1, Culyer AJ & Newhouse JP, eds., Elsevier Science B.V.

Drummond MF, *et al* (1997): Methods for the Economic Evaluation of Health Care Programmes, Oxford Medical Publications, Oxford: Oxford University Press.

EC (1995): ExternE: Externalities of Energy: Methodology. Volumes 2. Brussels: Commission of the European Communities DGXII (JOULE Programme).

EC (1998): Technical Guidance on Development of Risk Reduction Strategies, Directorate General XI, Brussels: European Commission.

Feeney DH, Torrance GW & Furlong WJ (1996): *Health Utilities Index*, pp. 239-252 in Quality of Life and Pharmacoeconomics in Clinical Trials, Second Edition, B. Spilker, ed., Philadelphia: Lippincott-Raven Publishers.

Frankel M (1979): Hazard, Opportunity and the Valuation of Life, Champaign-Urbana: Department of Economics, University of Illinois.

Freeman AM (1991): *Factorial Survey Methods and Willingness to Pay for Housing Characteristics: Comment*, Journal of Environmental Economics and Management, Vol. 20, pp.92-96.

Freeman AM (1993): The Measurement of Environmental and Resource Value: Theory and Methods, Washington DC: Resources for the Future.

French S (1988): Decision Theory, An Introduction to the Mathematics of Rationality, Chichester: Ellis Horwood.

Furlong W *et al* (1998): Mulitplicative Multi-Attribute Utility function for the Health Utilities Index Mark 3 (HUI3) System: A Technical Report, Working Paper 98-11, McMaster University Centre for Health Economics and Policy Analysis.

Gold MR *et al* (1996): Cost-Effectiveness in Health and Medicine, Oxford: Oxford University Press.

Golden, B., Wasil, E. & Harker, P. (eds) (1989): The Analytic Hierarchy Process: Applications and Studies, Springer Verlag, New York.

Graham JD (1995): Comparing Opportunities to Reduce Health Risks: Toxin Control, Medicine and Injury Prevention, Dallas: National Centre for Policy Analysis.

Gray GM (1997): Risk Assessment and Risk Management of Non-Ferrous Metals: Realizing the Benefits and Managing the Risks, Ottawa: ICME.

Gray WB & Shadbegian RJ (1993): Environmental Regulation and Manufacturing Productivity at the Plant Level**,** Discussion Paper, Center for Economic Studies, Washington DC: US Bureau of the Census.

Gray WB & Shadbegian RJ (1994): Pollution Abatement Costs, Regulation and Plant-Level Productivity, Discussion Paper, Center for Economic Studies, Washington DC: US Bureau of the Census.

Haener M, Boxall PC and Adamowicz WL (2001): *Modeling Recreation Site Choice: Do Hypothetical Choices Reflect Actual Behavior?* in Press American Journal of Agricultural Economics.

Hahn R (1996): Risks, Costs and Lives Saved: Getting Better Results from Regulation, American Enterprise Institute for Public Policy Research.

Hanemann WM (1982): Applied Welfare Analysis with Qualitative Response Models, Working Paper 241,University of California Berkeley.

Hammitt JK (2000): *Valuing Mortality Risk: Theory and Practice,* Environmental Science and Technology, Vol.34, No.8, pp.1396-1400.

Hammitt JK (2000): Evaluating Contingent Valuation of Environmental Health Risks: The Proportionality Test, AERE Newsletter.

Hanley N & Spash CL (1993): Cost-Benefit Analysis and the Environment, Aldershot: Elgar Publishing Ltd.

Harrington W & Portney PR (1987): *Valuing the Benefits of Health and Safety Regulations,* Journal of Urban Economics, Vol.22, No.1, pp.101-112.

Haveman R (1976): Evaluating the Impact of Public Policies on Regional Welfare, Regional Studies.

Helfert EA (1997): Techniques of Financial Analysis, 9th Edition, New York: McGraw- Hill.

Herwijnen M et al (1995): Sensitivity Analysis with Interdependent Criteria for Multi-Criteria Decision Making, Multi Criteria Decision Making, Vol.4, No.1, pp.57-70.

Horowitz J (1994): *Preferences for Pesticide Regulation*, American Journal of Economics, Vol.76, pp.396-406.

Jaffe et al (1995): Environmental Regulation and the Competitiveness of US Manufacturing: What Does the Evidence Tell Us? Journal of Economic Literature, XXXIII (March), pp.132-63.

Janssen R (1992): Multiobjective Decision Support for Environmental Management, Dordrecht: Kluwer Academic Publishers.

Janssen R & Munda G (1999): *Multicriteria Methods for Quantitative, Qualitative and Fuzzy Evaluation Problems,* in Bergh JCJM (Ed) Handbook of Environmental and Resource Economics, pp.1000-1012, Edward Elgar Publishing Ltd.

Janssen *et al* (2000): DEFINITE for Windows, Software package and User Manual produced for the Ministry of Finance, Amsterdam: Vrije Universiteit.

Johannesson M & Jonsson B (1991): Economic Evaluation in Health Care: Is There a Role for Cost-Benefit Analysis, Health Policy, Vol.17, pp.1-23.

Johannesson M & Johansson PO (1996): To Be, or Not to Be, That is the Question: An Empirical Study of the WTP for an Increased Life Expectancy at an Advanced Age, Journal of Risk and Uncertainty, Vol.13, No.2, pp.163-174.

Johansson PO (1995): Evaluating Health Risks: An Economic Approach, Cambridge: Cambridge University Press.

Johnson FR and Desvousges W (1997): Estimating Stated Preferences with Rated-Pair Data: Environmental, Health, and Employment Effects of Energy Programs, in Journal of Environmental Economics and Management, Vol.34, No.1, pp.79-99.

Johnson FR *et al* (1998): Using Stated Preferences and Health-State Classifications to Estimate the Value of Air Pollution, TER Working Paper G9802.

Jones-Lee MW (1976): The Value of Life: An Economic Analysis, Chicago: University of Chicago Press.

Jones-Lee MW et al (1985): The Value of Transport Safety: Results of a National Sample Survey in The Economic Journal, Vol 95, pp.49-72.

Jones-Lee MW (1989): The Economics of Safety and Physical Risk, Oxford: Basil Blackwell.

Jones-Lee MW & Loomes G (1994): *Towards a Willingness-to-Pay Value of Underground Safety*, Journal of Transport Economics and Policy, January, pp.83-98.

Jones-Lee MW & Loomes G (1995b): *Scale and Context Effects in the Valuation of Transport Safety*, Journal of Risk and Uncertainty, Vol.11, No.3, pp.183-203.

Kaplan RM & Anderson JP (1996): *The General Health Policy Model: An Integrated Approach*, pp. 309-322 in Quality of Life and Pharmacoeconomics in Clinical Trials, Second Edition, B. Spilker, ed., Philadelphia: Lippincott-Raven Publishers.

Keeler EB & Cretin S (1983): *Discounting of Life-Saving and Other Nonmonetary Effects*, Management Science, Vol.29, pp.300-306.

Keeney, RL & Raiffa, H (1976): Decisions with Multiple Objectives: Preferences and Value Tradeoffs, New York: John Wiley & Sons.

Keeney RL (1992): Value-Focussed Thinking: A Path to Creative Decision Making, Cambridge, Massachusetts: Harvard University Press.

Kidholm K (1992): *How to Put a Value on Safety in Traffic Planning*, Nationalokonomisk Tidsskrift, Vol. 130, No.4, pp.630-641.

Kind P (1996): *The EuroQoL Instrument: An Index of Health-Related Quality of Life*, pp. 191-201 in Quality of Life and Pharmacoeconomics in Clinical Trials, Second Edition, B. Spilker, ed., Philadelphia: Lippincott-Raven Publishers.

Klimberg R & Cohen RM (1999): *Experimental Evaluation of a Graphical Display System to Visualising Multiple Criteria Solutions*, European Journal of Operational Research, Vol. 119, No. 1, pp.191-208.

Koppl A *et al* (1995): Macro-economic and Sectoral Effects of Environmentally-Oriented Energy Taxation in Austria, Report prepared by the Austrian Institute for Economic Research, commissioned by the Federal Ministries for the Environment, Youth and Family, for Science and Research and for Agriculture and Forestry.

Krupnick A and Cropper ML (1989): Valuing Chronic Morbidity Damages: Medical Costs, Labour Market Effects, and Individual Valuations, Washington DC: Resources for the Future

Krupnick A and Cropper ML (1992): *The Effect of Information on Health Risk Valuations*, Journal of Risk and Uncertainty, Vol.5, pp.29-48.

Krupnick A *et al* (1993): The Benefits of Ambient Air Quality Improvements in Central and Eastern Europe: A Preliminary Assessment, Discussion Paper ENR93-19, Washington DC: Resources for the Future.

Krupnick A *et al* (2000): Age, Health, and the Willingness to Pay for Mortality Risk Reductions: A Contingent Valuation Survey of Ontario Residents, Discussion Paper 00-37, Washington DC: Resources for the Future.

Lancaster K (1966): *A New Approach to Consumer Theory*, Journal of Political Economy, Vol.74, pp.132-157.

List JA (2001): Using Choice Experiments to Value Non-Market Goods, University of Arizona.

Little I & Mirrlees J (1974): Project Appraisal and Planning for Developing Countries, London: Heinemann.

Louviere J, Hensher D and Swait J (2000): Stated Choice Methods: Analysis and Applications in Marketing, Transportation and Environmental Valuation, Cambridge University Press.

Loehman ET *et al* (1979): *Distributional analysis of Regional Benefits and Costs of Air Quality Control*, Journal of Environmental Economics and Management, Vol.6, pp.222-243.

Loehman ET et al (1982): Application of Choice Modelling to Policy Analysis of Public Goods: A Case Study of Air Quality Improvements, The Review of Economics and Statistics, Vol.64, No.3, pp.474-480.

Louviere JD, Hensher D & Swait J (2000): Stated Choice Methods: Analysis and Applications in Marketing, Transportation and Environmental Valuation, Cambridge: Cambridge University Press.

Luce RD (2000): Utility of Gains and Losses: Measurement-Theoretical and Experimental Approaches, Mahwah, New Jersey: Lawrence Erlbaum Associates.

Maddison D (1997): Valuing Health Effects of Air Pollution**,** Mimeo presented to the UK Department of Health Ad Hoc Group on Economic Appraisal of Air Pollution, also to be published by Edward Elgar.

Markandya (1997): The Valuation of Mortality from Air Pollution, mimeo prepared for the UK Department of Health Ad Hoc Group on Economic Appraisal of Air Pollution, London.

McConnell KE (1992): Model Building and judgements: Implications for Benefit Transfers with Travel Cost Models, Water Resources Research, Vol.28, No.3, pp.695-700

McDaniels TL *et al* (1992): *Risk Perception and the Value of Safety*, Risk Analysis, Vol.12, No.4, pp495-503.

McDowell I & Newell C (1996): *Quality Adjusted Life Years, Utility Theory and Healthy-year Equivalents*, Medical Decision Making, Vol.91, PP.142-149.

McFadden D (1974): *Conditional Logit Analysis of Qualitative Choice Behaviour*, in Frontiers in Econometrics, (Ed: Zarembka P), New York: Academic Press, pp.105-142.

McFadden D (1978): *Modeling the Choice of Residential Location*, Chapter 3 in Spatial Interaction Theory and Planning Models**,** Karlqvist A *et al*, eds., Amsterdam: North-Holland Press.

Mehrez A & Gafni A (1989): Quality Adjusted Life Years, Utility Theory, and Healthy-year Eq

Mendeloff JM and Kaplan RM (1989): Are Large Differences in Lifesaving Costs Justified, A Psychometric Study of the Relative Value Placed on Preventing Deaths, Risk Analysis, Vol.9, No.3, pp.349-363.

Miller GA (1956): The Magical Number Seven Plus or Minus Two: Some Limits on Our Capacity for Processing Information, Psychological Review, Vol.63, pp81-97.

Miller B & Hurley F (1998): Towards Quantifying and Costing the Overall Effects of Ambient Air Pollution on Mortality in the UK, Edinburgh: Institute of Occupational Medicine.

Mitchell RC & Carson RT (1989): Using Surveys to Value Public Goods: The Contingent Valuation Method, Washington DC: Resources for the Future.

Montgomery M & Needelman M (1997): *The Welfare Effects of Toxic Contamination in Freshwater Fish*, Land Economics, Vol.73, pp.211-273.

Morey E (1999): *Two Rums Uncloaked: Nested-Logit Models of Site Choice and Nested-Logit Models of Participation and Site Choice*, Chapter 3 in Valuing Recreation and the Environment**,** Herriges JA & Kling CL, eds., Cheltenham, UK: Edward Elgar Publishing Ltd

Morey E, Rowe RD & Watson M (1993): *A Repeated Nested-Logit Model of Atlantic Salmon Fishing*, American Journal of Agricultural Economics, Vol.75, pp.578-592.

Morgenstern RD *et al* (1997): Are We Overstating the Real Economic Costs of Environmental Protection? Discussion Paper 97-36-REV, Washington DC: Resources for the Future.

Morgenstern RD *et al* (2000**):** Jobs Versus the Environment: An Industry-Level Perspective, Discussion paper 99-01-REV, Washington DC: Resources for the Future.

Murray CJL (1994): *Quantifying Disability: The Technical Basis for Disability-Adjusted Life Years*, Bulletin of the World Health Organization, Vol.72, No.3, pp.429-445.

Murray CJL & Lopez AD (1994): Quantifying Disability: Data, Methods and Results, Bulletin of the World Health Organization, Vol.72, No.3, pp.481-494.

Murray CJL & Acharya AJ (1997): *Understanding DALYs*, Journal of Health Economics, No.16, pp.703-730.

Navrud S (1997): Valuing Health Impacts from Air Pollution in Europe: New Empirical Evidence on Morbidity, Working Paper, Norway: Agricultural University of Norway.

NERA & CASPAR (1997): Valuation of Deaths from Air Pollution, London: NERA.

Nijkamp *et al* (1990): Multicriteria Evaluation in Physical Planning, North Holland: Amsterdam.

ODA (1996): Appraisal of Projects in Developing Countries: A Guide for Economists, Overseas Development Agency, London: Her Majesty's Stationery Office.

OECD (1985): The Macroeconomic Impact of Environmental Expenditure, Paris: Organisation for Economic Co-operation and Development.

OECD (1995): The Economic Appraisal of Environmental Projects and Policies**,** Paris: Organisation for Economic Co-operation and Development.

OECD (1996): Implementation Strategies for Environmental Taxes, Paris: Organisation for Economic Co-operation and Development.

OECD (1997): Reforming Environmental Regulation in OECD Countries, Paris: Organisation for Economic Co-operation and Development.

Ontario Ministry of Environment & Energy (1996): Financial Assurance (Part XII - Ontario Environmental Protection Act) A Guide, Ontario: Ontario Ministry of Environment and Energy.

Ouwersloot H & Rietveld P (1996): *Stated Choice Experiments with Repeated Observations*, Journal of Transport Economics and Policy, Vol.30, pp.203-212.

Parsons GR & Hauber AB (1998): Spatial Boundaries and Choice Set Definition in a Random Utility Model of Recreation Demand, Land Economics, Vol. 74, No.1, pp.32-48.

Parsons GR & Kealy MJ (1992): Randomly Drawn Opportunity Sets in a Random Utility Model of Lake Recreation, Land Economics, Vol.68, No.1, pp.93-106.

Pearce D *et al* (1989): Blueprint for a Green Economy, London: Earthscan Publications.

Pearce D *et al* (1992): The Social Costs of Fuel Cycles, report prepared for the Department of Trade and Industry, London: EFTEC.

Pliskin JS, Shepard DS & Weinstein MC (1980): *Utility Functions for Life Years and Health Status*, Operations Research, Vol. 28 pp.206-224.

Porter ME & Van der Linde C (1995): *Toward a New Conception of the Environment-Competitiveness Relationship*, Journal of Economic Perspectives, Vol.9, No.4.

Revelt D and Train K (1998): *Mixed Logit with Repeated Choices: Households Choice of Appliance Efficiency Level*, Review of Economics and Statistics, Vol.53, pp.647-657.

Richardson (1991): *Economic Assessment of Health Care: Theory and Practice*, Australian Economic Review, Jan-March, pp.4-21.

Robinson JC (1995): The Impact of Environmental and Occupational Health Regulation on Productivity Growth in US Manufacturing, Yale Journal on Regulation, Vol.12, No.387.

Rowe *et al* (1995): The New York State Electricity Externality Study, prepared for the Empire State Electricity Energy Research Corporation, as quoted in US EPA (1999), Guidelines for Preparing Economic Analyses (June SAB Review Draft).

Rog L & Lootsma FA (1996): Rembrandt Program for Mulit-Criteria Decision Analysis, TU Delft: Faculteit ITS.

Roy (1985): Méthodologie Multicritère d'aide à la Décision, Paris: Economica.

Ryan M and Farrar S (2000): *Using Conjoint Analysis of Elicit Preferences for Health Case*, British Medical Journal, Vol.320, pp.1530-1533.

Saaty TL (1980): The Analytical Hierarchy Process, New York: McGraw Hill.

Saaty TL (1996): Decision Making for Leaders, AHP Series, Pittsburgh: Expert Choice.

Savage I (1991): *Psychological Features affecting Valuation of Life*, Economic Letters, Vol.35, pp.379-383.

Schwartz J and Dockery D (1992): *Increased Mortality in Philadelphia Associated with Daily Air Pollution Concentrations*, American Review of Respiratory Disease, Vol.145,pp.600-604.

Science Advisory Board (2000): An SAB Report on EPA's White Paper: Valuing the Benefits of Fatal Cancer Risk Reduction, Washington DC: USEPA.

Shim, J.P. (1989): *Bibliographical Research on the Analytic Hierarchy Process (AHP)*, Socio-Economic Planning Sciences, No 23, pp.161-7.

Slovic P *et al* (1984): *Modelling the Social Impact of Fatal Accidents*, Management Science, Vol.30(4), pp.464-474.

Slovic P (1987): *Perception of Risk*, Science, Vol.30, No.4, pp-423-439.

Small KA and Rosen HS (1981): *Applied Welfare Economics with Discrete Choice Models*, Econometrica, Vol.49, pp.105-130.

Smart B. Ed (1992): Beyond Compliance: A New Industry View of the Environment, Washington DC: World Resources Institute.

Smith VK & Kaoru Y (1990): *Signals or Noise? Explaining the Variation in Recreation Benefit Estimates*, American Journal of Agricultural Economics, Vol.68, pp.280-290.

Smith VK & Desvousges WH (1985): *The Generalized Travel Cost Model and Water Quality Benefits: A Reconsideration*, Southern Economics Journal, Vol.52, No.2, pp.371-381.

Starr C (1976): *General Philosophy of Risk Benefit Analysis*, in Energy and the Environment: A Risk-Benefit Approach, Ashley H, Rudman R & Whipple C, eds, Oxford: Pergamon.

Steuer RE (1986): Multi Criteria Optimization: Theory, Computation and Application, New York: Wiley.

Stijnen H (1995): Qualiflex, Rotterdam: Nederlands Economisch Instituut.

Thaler R & Rosen S (1976): *The Value of life Savings*, in Nester T, ed., Household Production and Consumption, New York: Columbia University Press.

Tolley GS *et al* (1986): Valuation of Reductions in Human Health Symptoms and Risks, Report to the US EPA, Washington DC: US Environmental Protection Agency.

Tolley GS, Kenkel, D & Fabian R, eds (!994): Valuing Health for Policy: An Economic Approach, Chicago: University of Chicago Press.

Torrance GW (1986): *Measurements of Health State Utilities for Economic Appraisal*, Journal of Health Economics, Vol.5, pp.1-30.

Train KE (1999): *Mixed Logit Models for Recreation Demand*, Chapter 4 in Valuing Recreation and the Environment, Herriges JA & Kling CL, eds., Cheltenham: Edward Elgar Publishing Ltd.

Tufte ER (1985): The Visual Display of Quantitative Information, Cheshire, Connecticut: Graphics Press.

Tufte ER (1997): Visual Explanations: Images and Quantities, Evidence and Narrative, Cheshire, Connecticut: Graphics Press.

Turner RK *et al* (1994): Environmental Economics, An Elementary Introduction, Hertfordshire: Harvester Wheatsheaf.

US EPA (1999): Guidelines for Preparing Economic Analyses, Washington DC: United States Environmental Protection Agency.

Vincke P (1992): Multicriteria Decision Aid, Wiley: New York.

Viscusi WK & O'Connor CJ (1984): *Adaptive Responses to Chemical Labelling*: *Are Workers Bayesian Decision Makers,* American Economic Review, Vol.74,No.5, pp942-956.

Viscusi WK, Magat WA and Huber J (1991): Pricing Environmental Health Risks: Survey Assessments of Risk-Risk and Risk-Dollar Tradeoffs for Chronic Bronchitis, Journal of Environmental Economics and Management, Vol.21, pp.32-51.

Viscusi (1992): Fatal Tradeoffs: Public and Private Responsibilities for Risk, New York: Oxford University Press.

Voogd H (1983): Multicriteria Evaluation for Urban and Regional Planning, London: Pion.

von Winterfeldt D & Edwards W (1986): Decision Analysis and Behavioural Research, Cambridge, Cambridge University Press.

Zahedi, F. (1986): *The Analytic Hierarchy Process: A Survey of the Method and Its Applications*, Interfaces, No 16, pp.96-108.

PART 4: ANALYTICAL ISSUES

AND PRESENTATION OF FINDINGS

1. INTRODUCTION

This Part (4) of the guidance document examines three key issues that are crucial to the conduct of SEAs:

- discounting;
- dealing with uncertainty; and
- presenting the findings of the SEA to decision makers and other audiences.

The appropriate treatment of these issues is of vital importance to the SEA process. All three relate to how the SEA will be perceived and understood and are crucial: poorly conveyed findings may result in inappropriate decisions being made, the findings being questioned or, worse, the SEA as a whole being rejected by key stakeholders. At all times during the SEA process, the audience for and the purpose of the SEA should be borne in mind.

2. DISCOUNTING

2.1 Introduction

In Part 2 of this document and again in the discussions provided in Part 3, it has been emphasised that comparative assessment of the costs and benefits of chemical risk reduction should be based on (or take into account) individuals' preferences. This is particularly true with regard to economic appraisal, but has also been highlighted with regard to MCA-based techniques (*e.g.* in establishing QALYs for different health outcomes).

If it is accepted that preferences at the individual and aggregate social level count, then this should take into account individuals' preferences for when in time those costs and benefits occur. In other words, one should also take into account individuals' preferences for the intertemporal allocation of costs and benefits.

Discounting is undertaken to reflect such intertemporal preferences. The process reflects the assumption that individuals would prefer to have gains earlier (now rather than some time in the future) and losses later (some time in the future rather than now); future costs and benefits are therefore 'discounted' and attributed a lower weight than those occurring immediately. The higher the discount rate used, the lower the importance placed on future costs and benefits. At any positive discount rate, costs and/or benefits that accrue more than 50 years into the future will have a very small 'present value'.

Starting with a simple example, consider which of the following risk reduction options should be recommended:

- Option A which would cost 100 million this year and provide 150 million in benefits next year; or

- Option B which would cost the same 100 million this year but provide 150 million in benefits in ten years time.

These two options are equivalent if one takes no account of the timing of the costs and benefits, but most people would prefer Option A because it delivers the benefits more quickly.

As a result of the above, a weighting factor is usually applied to adjust values through time, with the weight in any period (*t*) being:

$$d_t = \frac{1}{(1+r)^t}$$

In this equation, *r* refers to the chosen discount rate and the overall weight (*i.e.* d_t) is referred to as the discount factor.

This raises the following question: what discount rate is appropriate in calculating the present value of a regulation's future costs and benefits within a SEA?

2.2 Theories Underlying the Choice of Discount Rate

2.2.1 Introduction

There are essentially three key competing theories regarding the measure that should be adopted as the discount rate for public sector projects.[67]

- opportunity costs of capital (OCC);

- the consumption rate of interest (CRI); or

- a 'synthetic' rate which is a combination of both of the above.

The concept underlying the opportunity costs of capital approach is that public investments can displace or crowd out private investments or consumption. It sets the discount rate at the real rate of return (to society) forgone in the private sector. In contrast, the social rate of time preference is a measure of society's collective willingness to forego consumption today in order to have greater consumption tomorrow. Both of these approaches are discussed further below.

2.2.2 The Opportunity Costs of Capital

As indicated above, future costs and benefits are discounted to reflect the fact that, in general, individuals (and hence society) hold a positive rate of time preference.

[67] Other approaches exist including use of the 'accounting rate of interest' and the consumer discount rate. For a more detailed discussion on these and the subject of discounting in general, see Pearce *et al* (1989).

Discounting is, therefore, a way of accounting for the importance of time within individual preferences and how this affects the values attached to future costs and benefits. One reason for such preferences stems from the fact that capital is productive; for example, one dollar's worth of resources now will generate more than that dollar's worth of goods and services in the future. Put another way, most people require a positive real rate of return when they lend money. As a result, it is clear that they put a premium on being able to consume today and require a reward for foregoing consumption today, *i.e.* without interest on one's savings, there is no incentive to forgo consumption in the present. This concept is also referred to as the 'marginal productivity of capital'.[68]

The rate to be used in appraisals is then obtained by finding the rate of return on the next best investment of a similar risk which is displaced by undertaking a particular project (or in this case implementing a risk reduction option). In regulatory terms, adopting this rate indicates that if a private business can earn a 10% rate of return on its capital investment, then it should be able to earn a similar return on the investment required as part of risk management activities (Turner *et al*, 1994). However, there is some controversy surrounding the use of the opportunity cost of capital in that it is not always clear whether the public project will actually reduce the level of private investment.

2.2.3 Consumption Rate of Interest

Impatience or 'pure time preference' is another reason why the present is preferred to the future. Even if investment in capital were not productive, in the sense that it earned returns, people would still prefer to have money now rather than next year. This time preference arises not only because people are impatient, but also because the future is uncertain (*e.g.* there are risks of ill health and death in the future), and people expect to be better off in the future than they are now reflecting the diminishing marginal utility of consumption (see also Pearce *et al*, 1989).

At the individual level then, the rate at which individuals are willing to exchange consumption over time is usually referred to as the 'consumer discount rate' or the 'individual consumption rate of interest'. This is the rate at which consumers can borrow and lend in the marketplace. Under simplifying assumptions (*i.e.* perfect capital markets), this rate will yield discount factors that are equal to a consumer's marginal rate of substitution between income in different periods of time. The latter reflects the income that a consumer would require in the future to compensate for surrendering a unit of income today.

A number of concerns have arisen with regard to the use of the individual's consumption rate of interest (CRI) as reflected in market interest rates when the aim is to consider investments from a social perspective. One of the arguments is that the pure time preference component is individualistic and short-sighted and not relevant to social decisions. Another is that market interest rates may reflect precautionary saving motives rather than pure intertemporal consumption preferences.

In order to correct for these concerns, it is suggested that one should adopt instead the social rate of time preference. This attempts to measure the rate at which social welfare or the utility of consumption for society falls over time, depending upon:

- the rate of pure time preference;
- how fast consumption grows; and

68 This is a version of what is also referred to as the producer discount rate, which gives a discount factor equal to a producer's marginal rate of transformation. The discount rate is then the inverse of the marginal rate of capital productivity.

– how fast utility falls as consumption grows.

The rate is derived via the following equation:

$$r = (g \,.\, e) + p$$

Where r = social time preference rate of discount (or the STPR)
g = the expected growth in consumption
e = the elasticity of the marginal utility of consumption with respect to consumption (that is the rate at which marginal utility declines as consumption increases)
p = the pure time preference rate

The measure *e* represents the relationship between utility and each unit of consumption. The component '*g.e*' then accounts for the idea that as future societies are likely to be richer than the present generation, the present generation should attach less weight to future gains. However, this formula does tend to have its basis solely in theory with no empirical counterpart. As a result, it is generally not used as the basis for discounting in practical applications, such as the preparation of regulatory appraisals.

The difference between the CRI and the STPR is that the first is the rate at which individuals discount the future, while the second is the rate at which society discounts the future. The latter need not be the same as the former, but when they are, the two rates are equal. Both rates involve components related to the discounting of consumption and discounting on the basis of pure time preferences.

2.2.4 *Synthetic Rates*

The use of synthetic rates is generally aimed at deriving a discount rate that is a combination of both the consumption rate of interest and of the marginal opportunity costs of capital for the private sector. A very general form of synthetic rate is given as (Pearce *et al*, 1989):

$$W = h_1 . s + h_2 . r$$

Where W = the synthetic discount rate
h_1 = the fraction of government expenditure displacing private investment
h_2 = the fraction of government expenditure displacing private consumption (or 1 - h_1)
s = the pre-tax rate of return
r = the after-tax rate of return

2.2.5 *Choosing a Discount Rate*

What discount rates should be used in SEAs? Use of a rate calculated in terms of the OCC is the approach recommended by many international organisations. For example, the World Bank recommends that the standard opportunity cost of capital be used for environmental cost-benefit analyses. The Bank argues that a lower discount rate would cause more capital projects to pass the cost-benefit test and thus lead to additional environmental stress. Similarly, other lending agencies (*e.g.* the UK's Department for International Development) also recommends that the discount rate should be the opportunity cost of capital in the public sector and, on this basis, that a real discount rate in the range of 8 to 12% is likely to be appropriate in most countries. However, the context of these arguments is different from that of regulatory policy making which is aimed at taking a long-term perspective on behalf of society.

As a result of such differences in perspective, the US EPA recommends using (for intra-generational policies) the consumption rate of interest approach with no adjustments unless "…there are strong reasons to believe that a particular policy will affect the level of US private sector investment…" (US EPA, 1999); using this approach results in interest rates of 2-3%. However, the US OMB presently recommends a rate of 7%, based on an estimate of the average real pre-tax rate of return generated by private sector investments.[69]

The UK Treasury recommends a real discount rate of 6% for most purposes in the UK. This rate is a synthetic rate, described as one which reflects both time preferences and the cost of capital, and is based on the long-term pre-tax cost of capital for low-risk projects in the private sector.

2.3 Intergenerational Discounting

Some policies may have impacts that occur far into the future, affecting many generations in the process. Examples that are often cited include climate change, radioactive waste disposal and biodiversity. It is obvious, however, that given the persistence of many chemicals, policies concerning their management may also affect many generations. Dealing with such long time horizons presents problems in that the preferences of future generations are unknown and, hence, cannot be taken into account in the decision making process.

This aspect raises questions over the role of discounting in intergenerational decision making. Essentially, the higher the discount rate the lesser will be the weight attached to (positive and negative) costs and (positive and negative) benefits occurring in the future. Proposals which would result in costs well into the future but bring benefits in the near term would be likely to meet economic efficiency criteria the higher the discount rate. Conversely, proposals with costs in the near term but benefits arising well into the future would be unlikely to meet efficiency criteria under higher rates (or potentially any rate). In addition, the higher the discount rate, the lower will be the overall level of capital investment, leading to future generations inheriting a reduced capital stock. As a result, it can be argued that future generations are being discriminated against through the use of discount rates that reflect current preferences and capital productivity as opposed to future preferences and capital productivity.

It has been argued, though, that future generations are taken into account in decision making as they will be a factor underlying the preferences of the current generation. For example, consider an individual with the following utility function (Pearce *et al*, 1989):

$$U_i = U_I(C_i, U_j, U_k)$$

Where i = the current generation
j = the next generation
k = the third generation
U = utility
C = consumption

In this case, not only does utility in the present depend upon consumption in the present but the utility of the next and third generations as well. In other words, most individuals wish for a 'better' life for their children and grandchildren. Of course, this assumes that current generations make some sort of prediction on what future generations will gain utility from.

69 These variations emphasise the importance of sensitivity analysis.

It can be argued that components of non-use (or passive use) values, such as bequest values and altruism in the human health context, reflect such a utility function. In this case, environmental assets or health related investments are attributed value by the current generation in order for future generations to be able to enjoy the natural resource or to gain from improved health. It is not common practice though to ask respondents about their rate of time preference for such assets and investments. It may be the case, however, that these time preferences are taken implicitly into account when valuing assets in the future. Under this set of arguments, discounting effects occurring over very long time horizons may result in a double discounting (once as part of the valuation process and again through the application of an explicit discount rate).

An alternative approach is to derive discount rates from optimal growth models that maximise the utility of all present and future generations. The discount rate in this case has two components:

- a discount rate for pure time preference; and
- an adjustment reflecting the fact that the marginal utility of consumption will decline over time as consumption increases.

Most modern applications of this approach actually assume that the rate for pure time preference is zero, hence not inherently favouring the present generation. Estimates tend to range from 0.5 to 3% using these models.

At the present time, there is actually no one preferred approach for the treatment of intergenerational effects within SEA. The discount rate may be varied as part of sensitivity testing but the clearest way to actually understand any implications for future generations is to present the stream of costs of benefits undiscounted on a year by year basis. This allows decision makers to gain a better feel for the impacts of a risk reduction option over time, without the use of a discount rate clouding the analysis by weighting costs and negative benefits.

2.4 Discounting and Human Health Benefits

2.4.1 Introduction

There is no doubt that human health effects will have temporal differences depending upon the type of risk, exposure, latency, *etc*. Standard practice within CBAs would be for the valuations of these effects to be discounted over time in order to take into account time preferences. However, some researchers argue that it should not be applied to health effects. Drummond *et al* (1997) set out some arguments against the discounting of health effects:

- individuals will not necessarily invest in health through time;
- the health of future generations is given less weight in the analysis; and
- studies have suggested that individuals discount health at a different rate from other forms of monetary benefits.

In reply to the above points, it must be remembered that the whole process of discounting is designed to reflect society's time preference to consume in the current time period rather than to wait for future time periods (which may be uncertain). In this respect, situations can arise where alternative proposals will result in differing investments in health over time and in the trading of health effects over time.

It should also be borne in mind that the costs associated with a risk reduction option are likely to be discounted. Failing to discount health impacts can lead to questionable conclusions over the relative costs and benefits of alternative options. In addition, if some health effects are discounted (*e.g.* cost of illness estimates), while others are not [*e.g.* value of a statistical life (VSLs)], then this further confuses the relative relationships between costs and benefits. One of the problems that could arise is that in choosing not to discount health impacts investment will always be deferred to some point in the future, as future costs (when discounted) will always be lower. Furthermore, within the context of economic analysis, treating health impacts in a different way compared to other impacts (be they environmental, social or economic) may result in a misallocation of resources.

A key question to be asked regarding the discounting of health impacts is whether time preference actually exists for good states of health. Would individuals actually be willing to accept poor health in the future for good health today (which is effectively what time preference implies)? Examining the problem in another way: at a societal level with individuals acting altruistically, would society choose to treat the least healthy in society now and then treat the more healthy in the future? It has been argued (see, for example, Jones-Lee & Loomes, 1995) that it should be the pure time preference rate for utility that should be used rather than the social time preference rate. This would mean that much lower discount rates would be used in an analysis.

2.4.2 The Choice of Discount Rates in CBA

The debate then, once discounting is accepted, is whether health impacts should be discounted at the same rate as costs. Gold *et al* (1996) set out a number of trends emerging from the literature assessing what discount rates may be:

- individual discount rates frequently lie outside the conventional 0-10% range;
- despite variations between individuals, the mean discount rates do fall within the conventional range;
- discount rates tend to be lower when large-magnitude outcomes are being traded over time;
- discount rates tend to be lower the longer the time interval over which the trades are considered;
- discount rates for losses are typically lower than for gains;
- when a given outcome is embedded in a sequence of outcomes, the discount rate tends to be lower than when the outcome is evaluated singly; and
- the sequencing of outcomes can affect time preference, *i.e.* some people savour good outcomes and wish to postpone them; some dread bad outcomes and wish to get them over with; some attach special utility to having outcomes improve over time, even if this means that the total payoff is sub-optimal.

Given the above, a key issue is determining the length of time over which health impacts may occur; certainly, if they are within a ‘standard’ appraisal time frame (such as 15 or 30 years) there is a good case for using a similar discount rate across the whole analysis. However, if impacts will have a longer time frame then the use of the pure time preference rate for utility may be appropriate. In the meantime, however, Gold *et al* (1996) recommend that the same discount rate is adopted for health effects as for the discounting of costs. This is the approach also recommended in most governmental guidelines on the application of SEA.

2.4.3 Discounting of QALYs and other Physical Units

In general terms, all effects which arise as a result of a proposed risk reduction option should be discounted to take into account time preferences. This applies to impacts measured in physical units (*e.g.* lives saved) of options as well as those measured in monetary terms (VSLs). The implications are that where such measures are used as a numerator within a CEA-based analysis, the numerator measure of impacts should not be just an aggregation of the number of units gained over time but should include discounting.

Krahn & Gafni (1993) have argued that it is double discounting to discount QALYs if the preference instrument used to measure the QALY weight already incorporates the respondents' time preference. In view of this, Drummond *et al* (1997) highlight that such 'time trade-off methods' often capture respondents' time preference as they ask time-based questions. However, preferences derived this way are at the individual level and not at the societal level (the level at which decision makers should operate). As was highlighted in the previous section, individuals' time preference may vary considerably from societal time preference. The actual impact of discounting such individual preferences is, therefore, not clear. On the whole, Drummond *et al* recommends:

"…to continue to discount at the recommended social rate of discount regardless of how the preference weights were obtained…"

In adopting such an approach, however, decision makers should be aware of the methods behind the derivation of the QALY in order to understand whether there may be some element of double discounting.

2.5 Discounting and the Environment

There has been a longstanding debate concerning the application of discounting to environmental effects: this includes both effects measured in purely physical terms and those measured in monetary terms.

Economists argue that this debate stems from a misunderstanding as to the implications of discounting when applied to environmental goods and services. In this context, the choice of a discount rate may work in two opposing directions. Under a high rate, fewer investments are undertaken, particularly where these involve large up-front costs and long periods before the investment is paid back. As a result, natural areas may be preserved from development. Conversely, higher discount rates also imply that non-renewable resources should be developed more rapidly and that more active harvesting of renewable resources should take place. For both, this can lead to stocks being depleted (with this occurring quickly and before substitutes can be developed in the case of the non-renewables and as a result of over-harvesting in the case of renewables).

OECD (1992) raises two further considerations with regard to discounting environmental costs and benefits that are of considerable relevance in the context of risk management. The first concerns the treatment of potentially catastrophic effects when these may occur well into the future. The impact of discounting such effects is that their real importance in the future is not made clear.

The second issue relates to irreversibility. Pindyck (2000) identifies two types of irreversibility: the direct expenditure on reducing environmental risks which become *sunk costs,* and the *sunk benefits* that arise from adopting an environmental policy now rather than incurring further environmental damages.[70] An issue is the question of whether such irreversibilities may be important. Much research has been undertaken on this issue. Several authors suggest that although high rates

[70] For further reading on the issue of irreversibility see the special issue of *Resource and Energy Economics*, Vol. 22, No. 3, July 2000.

may shift cost burdens onto future generations, they may also slow down the pace of investment. The latter should reduce the pressure placed on natural resources as these are required for investment. However, what the balance of effects may be in the context of chemical risk is unclear. Certainly, many of the substances of concern and their substitutes involve the development of natural resources. They also involve the potential for unpredictable catastrophic effects occurring in the future.

However, as OECD (1992) argues, the potential for such effects to occur is not an argument for adjusting the discount rate, but rather one for ensuring that such losses are adequately taken into account in the analysis through proper valuation of environmental effects and ensuring that the assessment covers an appropriate time horizon. In other words, it should be remembered that discounting is undertaken solely to reflect the fact that individuals place less weight on impacts occurring in the future than those occurring today. If there are concerns that future environmental impacts are being given inadequate weight, then the response should be one of ensuring that the effects are properly quantified (be it in monetary terms, physical, or other units) and that the timescale over which they would occur is recognised within the analysis, rather than arbitrarily modifying the discount rate.

2.6 Practical Application

2.6.1 Overview

This section examines the practicalities of discounting within SEA, together with the associated decision criteria. In the first instance, however, it is important to begin by establishing what prices should be used, as these can also have important implications by unintentionally weighting certain impacts depending upon the basis adopted. Clearly the combined impact of the choice of a price base and the process of discounting can heavily determine the outcome of the SEA.

2.6.2 Choosing a Price Base

When making comparisons between risk reduction options, it is important to ensure that impacts, when valued in monetary terms, are consistent with regard to the year upon which prices are based. All prices should be adjusted to a common base year in order to facilitate comparison.

National statistics bodies keep records of changes in retail and producer prices that can be used to adjust prices to a common year. For example, consider a range of benefit values that are to be used in a SEA utilising benefit transfer techniques. These benefit values, at present, have different price years associated with them. In order for them to be brought together within the analysis, they should be adjusted to the most current year (ideally the year in which the appraisal is taking place, *i.e.* year 0 in the analysis). Table 2.1 sets out the adjustment process based on the UK's Retail Price Index (RPI).

Table 2.1: Example of Price Adjustment			
Original Value	**Year of Original Value**	**Adjustment Factor***	**Year 2000 Value****
£10.50	1989	1.50	£15.75
£2.00	1995	1.14	£2.28
£7.10	1985	1.82	£12.92
£37.25	1999	1.02	£38.00
£15.00	1991	1.28	£19.20
* This is derived by dividing the required base year price index (in this case January 2000, 166.4) by the original value's year price index (*e.g.* 111.00 for 1989), *i.e.* the adjustment factor will be 1.50. ** Values have been rounded.			

Using changes in retail prices often will be sufficient (especially given that these directly impact upon an individual's ability to pay); however, there may be more specific price 'adjusters' available for specific sectors, *e.g.* for construction prices, raw materials, and so on. In all cases, the most appropriate price adjuster should be used.

A potentially far more complex problem lies with expected changes in prices in the future, mainly due to the impacts of inflation on the price levels. It may seem logical to predict the level of inflation over the period of analysis and adjust any values accordingly. However, such an approach should not be adopted in CBAs or CEAs. Both of these methodologies operate on the basis of a partial equilibrium framework that relies on the concept of *ceteris paribus* ('other things being equal'), and which isolates specific markets for analysis. The assumption within this framework is to assume that all prices are constant throughout the study timeframe. The use of constant prices ensures that future costs and benefits (when valued in monetary terms) are measured in the same units throughout the whole analysis (simplifying the analysis).

An obvious exception to this approach is when such techniques as general equilibrium models or input-output analysis are used, as in these cases it may be possible to integrate fluctuating prices into the model. However, given the complexity of such models, it may be more appropriate to assume constant prices as in the partial framework.

2.6.3 How to Discount

Whether performing a cost analysis, a CEA or a CBA, discounting is used in the aggregation of the stream of costs and/or benefits occurring over time to a single figure, enabling comparison of alternative (risk management) options.

The formula for calculating the present value of any cost or benefit is:

$$d_t = \frac{1}{(1+r)^t}$$

Where d_t = the discount factor
r = the discount rate
t = the year in which the cost or benefit occurs

Table 2.2 provides an example of the type of impact that discounting may have on benefits. In this example, discounting over a five-year period diminishes the total value of benefits (compared to an undiscounted figure) by around 18% to take into account the impact of time preferences.

Table 2.2: The Impact of Discounting			
Year	**Undiscounted Value of Benefits**	**Discount Factor***	**Discounted Value of Benefits**
1	£100,000	0.9434	£94,340
2	£25,000	0.8900	£22,250
3	£150,000	0.8396	£125,940
4	£50,000	0.7921	£39,605
5	£200,000	0.7473	£149,460
Total	£525,002		£431,597
* Using the UK Treasury discount rate of 6% for government appraisals, and based on mid-year timing of benefits.			

The ultimate aim behind discounting is to adjust values appropriately to allow an overall assessment to be made as to the preferred way forward. The decision criteria used in most SEAs relate to the concept of economic efficiency, where a proposed action is deemed economically efficient (and hence justified) if the benefits of the action are greater than the costs over time (although sometimes neither are explicitly measured or quantified). The range of decision criteria used in SEA include:

– the **net present value** (NPV): the difference between discounted benefits and costs (*i.e.* present value benefits minus present value costs);

– the **benefit-cost ratio** (B/C): the ratio between discounted benefits and costs (*i.e.* present value benefits divided by present value costs);

– the **cost-effectiveness ratio** (C/E): the ratio between discounted cost and the discounted total of the pre-determined measure (such as discounted lives saved). For example, an appraisal may state that a certain risk reduction option saves 100 lives at $100 million over the life of the project; and

– the **cost-utility ratio** (C/U): this ratio usually refers to the use of QALYs with future QALYs discounted. For example, an appraisal may state that a particular measure costs $10 million per QALY gained.

The NPV is the basic measure of the economic gains (or losses) resulting from a project or policy. The formula used in calculating NPV is:

$$NPV = \sum_{t=0}^{n} \left[(B_t - C_t)(1+r)^{-t} \right]$$

Where: B = the sum of benefits in period t
C = the sum of costs in period t
r = the discount rate
t = the year in which the cost/benefit occurs, starting in year 0

As can be seen from the above formulae, a positive NPV indicates that a policy is justified in economic terms. This means that it yields a rate of return that is greater than the discount rate. When comparing alternative options (and funds are not constrained), the option with the highest NPV becomes preferred.

If one had an unlimited budget for projects or policies, it is desirable to undertake all the non-mutually exclusive projects for which the NPV is greater than zero. Additionally, in situations where mutually exclusive projects are being appraised, one should choose the project with the highest

NPV. When the budget is sufficiently limited to permit only a subset of the justifiable projects to be undertaken, capital funds take on a premium value (because there are still projects yielding a rate of return in excess of the discount rate). In such cases (clearly in most cases given the current pressures on public funds), the procedure for policy selection is usually the use of three ratios referred to above.

In cases where ratios are being generated (with the ratio depending upon the methodological route chosen), projects or policies with the highest ratio should be undertaken first. This ensures, for example in the case of the benefit-cost ratio, that benefits are continually maximised relative to costs.

It should also be noted that the use of NPV (or any other measure, for that matter) does not imply that other unquantified effects should be excluded from the decision making process. In such cases, the unquantified effects should be presented alongside the quantified effects in order to fully inform decision makers as to the probable results of various actions.

2.7 ZOZ Case Study

2.7.1 The Discounting Procedure and ZOZ

During the analysis for the hypothetical ZOZ case study, a number of impacts (costs and benefits) have been quantified in terms of either a monetary value or a physical unit (or utility unit). As this includes impacts that will occur annually, those arising in future years need to be discounted so that they are converted to the same basis as those occurring immediately.

The first step that the analysts took as part of the work on discounting was to ensure that all impacts were being valued in terms of the same price base, with year 2000 prices providing the basis for the analysis. In some cases, this involved inflating some of the money values attached to particular cost items (*e.g.* control technologies) and benefit estimates (*e.g.* medical costs) to 2000 prices.

The next step was to determine what discount rate should be used. Government policy for the country was for a standard rate of 5% to be used in public sector appraisals (with this based on the CRI). Using the formula given above, the resulting discount factors are given in Table 2.3.

Table 2.3: Derivation of Discount Rates

Year	Calculation	Discount Factor
0	$1/(1.05)^0$	1.0000
1	$1/(1.05)^1$	0.9524
2	$1/(1.05)^2$	0.9070
3	$1/(1.05)^3$	0.8638
4	$1/(1.05)^4$	0.8227
5	$1/(1.05)^5$	0.7835
. ..	. ..	. ..
15	$1/(1.05)^{15}$	0.4810

The next step is then to multiply the relevant valuation by the discount factor for that year. So, using health effects as the example, the discounted value of the benefits from reducing chronic health effects under the training option in year 5 is:

$$0.7835 \text{ x } (\$9{,}000 + \$65{,}000) \text{ x } 5 = \$289{,}895$$

which is found as:

(discount factor) x (COI + WTP) x (no. of cases) = discounted benefits

This process was repeated every year for both chronic and acute health impacts, resulting in total discounted training benefits of roughly $13.65 million, aggregated over the 15 years of the study (see also Table 3.12 in Section 3.6.3 of part 3).

A similar approach was also taken for the discounting of QALYs. For example, again using benefits of training for chronic health impacts, the derivation of benefits in year 5 is:

0.7835 x (13.25 x 5)

which is

(discount factor) x (gain in QALY per year x number of years) = discounted QALY

This process was then repeated every year for both chronic and acute health impacts, resulting in total discounted training benefits of 976 QALYs, aggregated over the 15 years of the study. For reference, the full spreadsheet used in calculating health benefits is set out in Figure 2.1, provided at the end of this section. This includes both the discounting of monetary valuations and discounting of QALYs for use in the MCA-based assessment.

This discounting procedure was repeated across all of the cost and benefit estimates, for the 15-year time period (which was selected on the basis that this reflected a reasonable period for the continued use of ZOZ, taking into account trends in the industry and on-going developments and technological changes).

2.7.2 Decision Criteria and ZOZ

Separate cost estimates were developed for each sector. The estimated total present value costs for each sector and each option are presented in Table 2.4. As can be seen from this table, the options apply to different sectors and the costs vary considerably across the options and across sectors.

Table 2.4: Summary of Option Costs by Sector ($ million)

Option	ZOZ Production	Intermediates Manufacture	Textiles Sectors	Industrial Cleaning	Total Costs All-Sector
A) Voluntary agreement	N/a	$19.8	$3.2	$35.7	$58.7
B) Product labelling	N/a	N/a	$2.9	$3.2	$6.1
C) 1) Training 2) Training and equipment*	N/a	$8.3 $16.4	N/a	N/a	$8.3 $16.4
D) Product tax	N/a	N/a	N/a	$36.2	$36.2
E) Marketing and use	N/a	$22.3	$16.7	$77	$116
F) Command and control	$15.9	$17.8	$20.9	N/a	$54.6

* The option involving training on its own was considered as a sub-option of this option.

The lowest cost option is that of product labelling but, as can be seen from Table 2.4, it also delivers the lowest level of environmental benefits and no health benefits. The two training options are then the next least expensive options, although these only deliver health benefits. Table 2.5 provides a summary of both the costs and benefits arising under the different options, assuming that there is no combining of options to produce a more effective strategy for reducing risks (*e.g.* combining training with a command and control based measure, or command and control with a voluntary agreement as discussed in Section 4 of Part 3).

Table 2.5: Summary of Costs and Benefits under Alternative Options

Option	Total PV Costs (millions)	PV Benefits		
		Health (millions)	Health (QALYs gained)	Environmental (millions)
A) Voluntary agreement[1]	$ 58.7	$18.4	1,468	$ 30.0
B) Product labelling	$ 6.1	$ 0.0	None	$ 4.5
C) 1) Training 2) Training and equipment	$ 8.3 $ 16.4	$13.6 $18.4	979 1,468	$ 0.0
D) Product tax	$ 36.2	$ 0.0	None	$ 30.6
E) Marketing and use	$116.0	$21.2	1,956	$130.1
F) Command and control	$54.6	$ 0.0	None	$ 85.3

1. Excluding any action that may be taken by the producers of ZOZ

The next step was to combine the cost and benefit data using the decision criteria discussed above to provide an indication of the relative performance of the different options. The results of this work are set out in Table 2.6 for each of the main options.

Table 2.6: Option Performance Using Different Decision Criteria

Option	Net Present Value ($ millions)	Benefit-Cost Ratio	Cost per QALYs ($ 000)	Cost-effectiveness Environmental Ranking[2]
A) Voluntary agreement[1]	- 10.3	0.82	11.2	3
B) Product labelling	- 1.6	0.74	N/a	4
C) 1) Training 2) Training and equipment	5.3 2.0	1.64 1.12	8.5 11.2	N/a
D) Product tax	-5.6	0.84	N/a	3
E) Marketing and use	35.3	1.30	11.4	1
F) Command and control	30.7	1.56	N/a	2

1. Excluding any action that may be taken by the producers of ZOZ
2. Assumes a target whereby PEC/PNEC <1, with ranking based on divergence from this

As can be seen from Table 2.6, the choice of options depends to a degree on the decision criteria adopted:

– marketing and use restrictions perform the best in terms of having the highest net present value, and is the only option to achieve the pre-set environmental target (PEC/PNEC <1) used as the basis for the cost-effectiveness analysis;

- the command and control option has a higher benefit-cost ratio than marketing and use restrictions but does not meet environmental targets; and

- the training option has a lower cost per QALY gained than the marketing and use restrictions, with both the voluntary agreement and training and equipment options also having marginally lower costs per QALY gained.

The above findings suggest that there may be merit in examining different combinations of the various options, as was highlighted in Section 4 of Part 3 with regard to environmental benefits. For example, by combining the voluntary agreement with the command and control option, both health and environmental benefits are delivered with these having an estimated value of around $125 million. Assuming that cost savings could be generated of around $23 million through increased flexibility, then the combined introduction of these measures may yield a comparable net present value.

It must be remembered, however, that not all of the predicted environmental benefits are accounted for in the above figures; it was not possible to place a monetary value on the wider ecosystem benefits that would arise under the various options. Again, environmental benefits were expected to be greater under marketing and use restrictions, with these ensuring that environmental targets (PEC/PNEC <1) were achieved.

In recommending any particular option or combination of options, the simple assessment of equity and distributional effects would also be taken into account (see Section 5 of Part 3). Given that it is suggested that marketing and use restrictions may give rise to greater detrimental effects than some of the other options, this may result in further pressure to modify some of the other options or develop new combinations of options (*e.g.* combining restrictions on use for some sectors with command and control or other options for other sectors).

Figure 2.1: ZOZ Discounting Spreadsheet

	Year																
	0	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	
Discount Factor	1.0000	0.9524	0.9070	0.8638	0.8227	0.7835	0.7462	0.7107	0.6768	0.6446	0.6139	0.5847	0.5568	0.5303	0.5051	0.4810	
Chronic COI	9000	9000	9000	9000	9000	9000	9000	9000	9000	9000	9000	9000	9000	9000	9000	9000	
Chronic WTP	65000	65000	65000	65000	65000	65000	65000	65000	65000	65000	65000	65000	65000	65000	65000	65000	
Training (cases avoided)	0	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	
Train & Equip (cases avoided)	0	1.5	3	4.5	6	7.5	9	10.5	12	13.5	15	16.5	18	19.5	21	22.5	
Ban on Use (cases avoided)	0	2	4	6	8	10	12	14	16	18	20	22	24	26	28	30	
																	Total
Training benefits	0	70,476	134,240	191,772	243,520	289,905	331,320	368,133	400,689	429,310	454,296	475,929	494,472	510,169	523,250	533,929	$5,451,409
Training & Equip benefits	0	105,714	201,361	287,658	365,280	434,857	496,979	552,199	601,033	643,964	681,444	713,893	741,707	765,254	784,876	800,893	$8,177,114
Ban on Use benefits	0	140,952	268,481	383,544	487,040	579,809	662,639	736,266	801,378	858,619	908,592	951,858	988,943	1,020,338	1,046,501	1,067,858	$10,902,818
Acute COI	9000	9000	9000	9000	9000	9000	9000	9000	9000	9000	9000	9000	9000	9000	9000	9000	
Acute WTP	275	275	275	275	275	275	275	275	275	275	275	275	275	275	275	275	
Training (cases avoided)	0	12	24	36	48	60	72	84	96	108	120	132	144	156	168	180	
Train & Equip (cases avoided)	0	15	30	45	60	75	90	105	120	135	150	165	180	195	210	225	
Ban on Use (cases avoided)	0	15	30	45	60	75	90	105	120	135	150	165	180	195	210	225	
																	Total
Training benefits	0	106,000	201,905	288,435	366,267	436,032	498,323	553,692	602,658	645,705	683,285	715,823	743,712	767,322	786,997	803,058	$8,199,214
Training & Equip benefits	0	132,500	252,381	360,544	457,834	545,040	622,903	692,115	753,322	807,131	854,107	894,779	929,640	959,152	983,746	1,003,823	$10,249,017
Ban on Use benefits	0	132,500	252,381	360,544	457,834	545,040	622,903	692,115	753,322	807,131	854,107	894,779	929,640	959,152	983,746	1,003,823	$10,249,017

Total benefits	
Training	$13,650,623
Train & Equip	$18,426,131
Ban on Use	$21,151,835

QALYs	**Acute Gain**		**Chronic Gain**	
Training	12 * 0.0038 =	0.0456	1*13.25 =	13.25
Training & Equip	15*0.0038 =	0.057	1.5*13.25 =	19.875
Ban on Use	15*0.0038 =	0.057	2*13.25 =	26.5

	0	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	Total
QALY Acute undiscounted																	**Total**
Training	0	0.05	0.09	0.14	0.18	0.23	0.27	0.32	0.36	0.41	0.46	0.50	0.55	0.59	0.64	0.68	5.472
Training & Equip	0	0.06	0.11	0.17	0.23	0.29	0.34	0.40	0.46	0.51	0.57	0.63	0.68	0.74	0.80	0.86	6.84
Ban on Use	0	0.06	0.11	0.17	0.23	0.29	0.34	0.40	0.46	0.51	0.57	0.63	0.68	0.74	0.80	0.86	6.84
QALY Acute discounted																	
Training	0	0.04	0.08	0.12	0.15	0.18	0.20	0.23	0.25	0.26	0.28	0.29	0.30	0.31	0.32	0.33	3
Training & Equip	0	0.05	0.10	0.15	0.19	0.22	0.26	0.28	0.31	0.33	0.35	0.37	0.38	0.39	0.40	0.41	4
Ban on Use	0	0.05	0.10	0.15	0.19	0.22	0.26	0.28	0.31	0.33	0.35	0.37	0.38	0.39	0.40	0.41	4
QALY Chronic undiscounted																	**Total**
Training	0	13.25	26.50	39.75	53.00	66.25	79.50	92.75	106.00	119.25	132.50	145.75	159.00	172.25	185.50	198.75	1590
Training & Equip	0	19.88	39.75	59.63	79.50	99.38	119.25	139.13	159.00	178.88	198.75	218.63	238.50	258.38	278.25	298.13	2385
Ban on Use	0	26.50	53.00	79.50	106.00	132.50	159.00	185.50	212.00	238.50	265.00	291.50	318.00	344.50	371.00	397.50	3180
QALY Chronic discounted																	
Training	0	12.62	24.04	34.34	43.60	51.91	59.32	65.92	71.74	76.87	81.34	85.22	88.54	91.35	93.69	95.60	976
Training & Equip	0	18.93	36.05	51.51	65.40	77.86	88.99	98.87	107.62	115.30	122.02	127.83	132.81	137.02	140.54	143.40	1464
Ban on Use	0	25.24	48.07	68.68	87.21	103.82	118.65	131.83	143.49	153.74	162.69	170.43	177.07	182.70	187.38	191.20	1952

3. MANAGING UNCERTAINTY

3.1 Introduction

The terms risk and uncertainty are sometimes used interchangeably. For example, it may be stated that a policy carries risks whereas one perhaps should state that there are uncertainties associated with adopting the policy. It is important to be clear about the use of such terminology as the terms may mean different things to different people.

In this report, risk is defined as the likelihood (or probability) of a specified adverse outcome. As such, risk comprises two components: one related to the likelihood/probability of occurrence of an event; and one related to the scale of the associated consequences should the event occur. For example, the risk of contracting cancer following a particular exposure to chemical X may be one chance in one million. Since, by definition, all statements of risk (usually resulting from an analysis of data in some form of risk assessment) are predictive, they carry a degree of uncertainty.

In other words, the risk associated with a particular situation is a characteristic of that situation and is measurable (in terms of likelihood/probability and consequence). The degree of confidence that one may have in that 'measurement' is reflected in the degree of associated uncertainty - the lower the uncertainty, the greater the degree of confidence in the estimate of risk.

Of course, within the context of SEA, uncertainty may be associated with many other factors apart from the risk posed by a chemical. For example, there will usually be uncertainties associated with economic valuations, particularly if extrapolated 50 years into the future.

In the next sub-section, we explore the different types of uncertainty and some of the methods used to analyse uncertainty are described in greater detail in Section 3.3.

3.2 Types of Uncertainty

3.2.1 Overview

Uncertainty may range from the mundane to the frontiers of human knowledge. At the mundane level, the lack of data makes any estimate of, say, exposure to a particular chemical uncertain. At the other end of the scale, despite extensive and prolonged research, the precise mechanisms by which dioxins harm the human body remain uncertain. One common way of thinking about uncertainty is to categorise events or outcomes into one of two types:

- some things are uncertain, but with a known structure (usually in the form of a probability density function) so that the distribution of outcomes can be predicted even if the precise outcome itself cannot be predicted (often referred to as a stochastic outcome); for example, one can predict a probability of one excess cancer in a population of 1,000 people, but not which particular individual would be affected; while

- there are other things that we simply do not know, creating problems in estimating even the distribution of outcomes.

These two categories of uncertainty can be further divided into four different types that may make the analysis of uncertainty more straightforward: knowledge uncertainty; real world uncertainty; data uncertainty and modelling uncertainty.

3.2.2 Knowledge Uncertainty

Knowledge uncertainty represents areas where there is a lack of scientific knowledge resulting, for instance, from ignorance of the mechanisms or interactions between different systems components. Currently, for example, considerable resources are being devoted to examining the mechanisms of climate change as well as their potential impacts.

Such uncertainties may or may not be significant to a given SEA. For example, a lack of knowledge over the precise mechanism of the toxic effects of a particular chemical may not be significant in terms of understanding the costs and benefits of limiting those effects through regulation. In other cases, this lack of knowledge may lead to significant over- or under-predictions of the likely costs and/or benefits of regulation.

3.2.3 Real World Uncertainty

Although the range of 'natural' conditions can be probabilistically predicted, there is always uncertainty as to whether the sun will shine tomorrow. Such real world uncertainties also include a stock market crash, a war in a specific region of the world or an environmental catastrophe sometime in the future. Such (stochastic) events could render any current SEA meaningless due to a step change in the circumstances being modelled.

Again such uncertainties may or may not be significant. By way of example, if one is considering a range of possible policy options, they may all be equally susceptible to 'real world uncertainty' and, as such, this does not impact upon the selection of the 'best option'.

3.2.4 Data Uncertainty

Data uncertainty not only covers errors in measurement and incomplete data but also the relevance of the parameter being recorded. For example, how does the monitoring of chemicals in raw vegetables relate to the human intake of those vegetables when cooked? How do cost estimates developed for a process compare to the costs that would be incurred by others producing the same goods but using a different process?

Data uncertainty characterises many of the difficulties encountered in chemical risk management. Although there is often extensive literature on the results of animal and human experiments involving particular chemicals, 'proving' a dose-response relationship (particularly at the very low doses in everyday life) is notoriously difficult in the absence of extensive data on both exposures and effects. For this reason, data uncertainty is usually expressed in terms of confidence limits.

Another common area of data uncertainty within SEAs is that associated with starting assumptions - for example, during the specification of the baseline to be used in the analysis. The appropriate baseline conditions cannot be established with certainty since the analyst will never have complete information on current and future values (for example, on the exact quantities of the chemical used in different locations and by different processes now and in the future) in the absence of risk management. Where the level of uncertainty is such that assumptions about baseline components are necessary, these need to be as realistic as possible, with all available information being taken into account.

3.2.5 Modelling Uncertainty

Modelling uncertainty relates to the validity of the methods used to represent, in mathematical terms, either the results of adopting a given risk reduction option or an individual's preferences for different outcomes. These uncertainties can arise from a lack of knowledge, from decisions made by analysts during the modelling process and from assumptions inherent within different models (the model represents our best judgement).

In the course of an assessment, 'systems' will be modelled through the use of simple spreadsheets or through complex software packages. It is well known that the precision with which computer generated results are presented often far outweighs their robustness. At the very least, any estimate resulting from the use of a model should indicate the degree of uncertainty in qualitative terms (low, medium, high).

Some examples of where modelling uncertainty in SEA may be encountered include:

- ***ex ante* and *ex post* valuation estimates**: environmental contamination generally increases the probability that an effect will occur, increase in severity, or both. People value these changes in risk differently as certain changes in health status. This means that values may differ significantly before and after particular events (such as an oil spill, the Exxon Valdez or Bhopal incidents providing good examples);

- **incomplete estimates of willingness to pay**: health insurance and paid sick leave shift the costs of illnesses from individuals to others. This may lead to problems in estimating total willingness to pay elicited through contingent valuation surveys. If these concerns are not adequately addressed, respondents may underestimate their willingness to pay, assuming that some related costs will be borne by others; and

- **timing of health and environmental effects**: environmental contamination may cause immediate or delayed health effects, and the value of avoiding a given health effect depends on whether it occurs now or in the future. Some research has shown that workers discount future risks of fatal injuries (*i.e.* they have a lower willingness to pay for a future risk than a present risk of equal magnitude); see, for example, Viscusi & Moore (1989).

3.3 Methods for Analysing Uncertainty

3.3.1 Introduction

This section sets out a number of methods for analysing uncertainty. It is worth considering three key questions with regards to uncertainty analysis:

- who will use it?

- what will they use it for? and

- what will they want from it?

Those undertaking SEAs inevitably face uncertainty as outlined above. The important point to remember is that analysts should take this into account and convey it to decision makers as part of the SEA process. The answer to the above three questions, therefore, tends to lie with the decision maker when considering a risk reduction option. For example, it may be the case that the decision maker needs to weigh up two options:

- one which has the potential to lead to significant reductions in risk at potentially low costs but with considerable uncertainty surrounding what actually might happen in terms of the end level of risks and the investment required; and

- another which reduces most of the risk, costs more but is a great deal less uncertain in terms of both the risk reduction and cost outcomes.

It is the role of those undertaking the SEA to convey this message and give an indication as to the likely success of each option and the expected reduction in risk. The tools set out in this section go some way toward tackling this problem.

3.3.2 Sensitivity Analysis

Above discussion highlights the importance of the explicit incorporation of uncertainty into SEA and suggests that the most appropriate approach for doing this is through the use of expected values. However, the discussion also makes it clear that focusing on the expected value of each outcome alone is unlikely to be adequate. Instead, decision makers will want to see information on the range of plausible outcomes associated with a given option. As indicated, such information is developed through the use of sensitivity analysis.

Sensitivity analysis is used to identify the variables that contribute most to uncertainty in predictions. The basic principles are:

- **focus on key variables**: often a full sensitivity analysis is not feasible (due to time or data constraints) and the analyst must limit the analysis to those assumptions that are considered key. In determining which parameters are key, the analyst should carefully consider the range of possible values for the input parameters and each one’s functional relationship to the output of the assessment. A plausible range of values for each key parameter should be specified, including a rationale for the range of values tested;

- **identify switching points**: switching points are those values at which the recommended policy decision would change from the selection of one option to another; they can often provide an indication of the robustness of choosing one option over another;

- **assess the need for more detailed analysis**: sensitivity analysis can also be used as a screening device to determine if more extensive analysis is required. This may include further research to minimise or better characterise uncertainty associated with one (or more) parameters or to identify the need for more sophisticated analysis. Value of information analysis can be used to prioritise further research and/or actions in terms of the expected net benefits of the research results; and

- **clearly present the results**: the results of the sensitivity analysis should be presented clearly and with accompanying descriptive text. Ideally, the end result of an uncertainty analysis should be a probabilistic range resembling a confidence interval.

The first step in a sensitivity analysis is to determine a plausible range of values for uncertain factors. The second is to consider whether the sensitivities for the various factors are related in any way. The final stage is then to assess the affect of changes in value on the outcome. A sensitivity analysis undertaken in this way has three advantages. It shows which of the uncertainties affect the decision most; identifying where further work could be targeted to improve the basis of the decision. It provides the decision maker with a good feel for the overall 'riskiness' of an option, and it can highlight inconsistencies between assumptions made about the individual factors affecting the outcome of the project.

The outcome of an initial assessment of uncertainty may be sufficient to support policy decisions. If the implications of the uncertainty are not, however, adequately captured, then it will be necessary to undertake a more sophisticated analysis. The key methods for undertaking such analyses are:

- scenario testing;
- Monte Carlo analysis; and
- use of the Delphi technique.

These are discussed in more depth below. Also covered are the use of value of information analysis, portfolio analysis and robustness analysis within the context of SEA. Annex 1 to this part contains more in-depth methods for integrating uncertainty into a SEA.

3.3.3 Scenario Testing

Most sensitivity analyses vary only one parameter at a time, leaving others at their base value. A more complete analysis presents the outcomes as the parameter is increased and decreased. Simultaneous variation of two parameters can often provide a fuller picture of the implications of the base values and also the robustness of the analysis. This approach is often referred to as scenario testing.

Scenario testing, therefore, uses a range of possible outcomes (such as minimum, maximum and most typical) taking variable and parameter uncertainty into account. Values of uncertain inputs are selected (*e.g.* best and worst cases) which give rise to the specified outcomes. These are modelled deterministically to indicate the range of outcomes.

This is a particularly useful approach when time and resources are limited. The method is limited, however, to a small number of scenarios (with Monte Carlo approaches used if more scenarios are chosen).

There are problems associated with scenario analyses, however, with these including:

- the need to persuade decision makers to take the long-term scenarios seriously;
- maintaining consistency when specifying the scenarios; and
- preventing emphasis being placed on average values to ensure that a sufficiently wide range is considered.

If the analyst is not careful, additional areas of uncertainty may be introduced into the analysis if extensive scenario testing is undertaken.

3.3.4 Monte Carlo Analysis

Any analysis of uncertainty should answer questions of the type 'what is the chance of ...?'. Where there are numerous uncertainties affecting the assessment, it may be important to go beyond a scenario analysis and to consider the distribution of possible values. Where this is the case, a Monte Carlo analysis may be more valuable.

Monte Carlo analysis is a probabilistic method, which is particularly useful since it explicitly characterises analytical uncertainty and variability.[71] It is used to quantify the uncertainty and variability in any numerical estimates to identify where the key sources of uncertainty and variability lie, and to determine their relative contribution to overall variance and the range of the results. This type of analysis may be useful when, for example, it is necessary to rank options or the factors making up options. In risk assessments this may include ranking of exposures, exposure pathways, sites or contaminants; when the cost of regulatory or remedial action is high and the exposures are marginal; or when the consequences of simplistic exposure estimates are unacceptable. It should be noted, however, that Monte Carlo analysis can only be useful when there is sufficient knowledge about the dimensions and parameters of risk.

A Monte Carlo analysis is undertaken using probability distributions for each factor and a description of any inter-relationships between the distributions. Using repeated random values drawn from the various distributions, a plot of the probability distribution of the sample is generated. It is particularly useful where uncertainties in costs and benefits are disaggregated or interdependence between factors is considered important. The result of a Monte Carlo analysis is a frequency distribution that provides an estimate of the true probability distribution of the appraisal results.

In other words, the approach describes probability density functions (PDFs) for each uncertain input which are randomly sampled (the expected variability in each being specified by a distribution), over many repeated calculations, to produce a representative PDF for the output model. This approach can handle uncertainty in input data and parameters arising from error and variability providing these can be described by a PDF. There must, therefore, be some sort of information on the uncertainty on the input data to use this method.

The number of iterations that are required to ensure that each PDF is adequately sampled is an important consideration (sometimes in the region of 10,000). It is often useful to estimate the rate of convergence of the output statistics. Alternatively the number of tests required to determine the mean with a known confidence can be calculated using statistical methods.

The accessibility to Monte Carlo simulations is now widespread, with many add-ons available for spreadsheets (although the process can be conducted by simply utilising a random number generator).

3.3.5 Delphi Techniques

The Delphi technique is essentially an expert based technique for assigning agreed probabilities to different outcomes, where these may relate to a particular aspect of a SEA (*e.g.* a dose-response relationship) or to more complex scenarios representing predicted outcomes. The technique is an approach developed by Dalkey & Helmer (1962) as an outgrowth of RAND Corporation's research on the use of expert opinion.

[71] There are numerous publication available on Monte Carlo analysis. The interested reader may wish to consider Hammersley & Handscomb (1964), Kalos & Whitlock (1986), Ripley (1987) and Morgan & Henrion (1990).

The technique usually involves written communication between a group of experts, such that they never meet face-to-face. A problem is defined and a questionnaire sent to the panel of experts who are asked to pass their opinions. All of the responses are then compiled; the compiled responses are then sent back to the panel for them to rank and comment on both the probability and nature of effect. The responses to this second phase are also compiled to identify areas where there is general consensus and where there is divergence of thought. Finally, the panel is sent the compilation of phase 2 responses and asked to comment on and/or review the widely divergent predictions that were made. The resulting responses are then compiled into a final report that gives details on:

- predicted events about which there is a consensus;
- any significantly divergent opinion from the general consensus and reasons for this divergence; and
- those events around which there is significant uncertainty.

Alternatively, the process of continually sending back compilations of responses can continue as many times as it is necessary to reach a consensus or a decision that most of the experts agree is a good one.

The step-by-step application may take the following form:

1. **Pick a facilitation leader**: this person should not be a stakeholder but should be an expert in research data collection;
2. **Select a panel of experts**: the panellists should have an intimate knowledge of the projects, or be familiar with criteria that would allow them to prioritise the projects successfully;
3. **Build a comprehensive list of appropriate criteria**: input from all interested parties is acceptable at this stage, not just the panellists;
4. **Panel prioritises the criteria**: each panellist ranks the criteria individually and anonymously where there is considerable political/emotional interest;
5. **Calculate the mean and deviation**: for each item in the list, the mean value is found. All items with means greater than a specified value are removed. The criteria are ranked and the results shown to the panel. Reasons should be given for items with high standard deviations. Panellists may also re-insert removed items after discussion;
6. **Re-rank the criteria**: the ranking procedure is repeated until the results stabilise. The results do not have to be in complete agreement, but a consensus needs to be reached that is acceptable to all;
7. **Identify project constraints and preferences**: hard constraints are used to set boundaries on the project ranking with soft boundaries introduced as preferences. Each panellist is given a supply of preference points (typically 70% of the total number of projects);

8. **Rank projects by constraint and preference**: each panellist first ranks the project according to the hard constraints. Next the panellist spreads his/her preference points among the project list as desired;

9. **Analyse the results and feedback to the panel**: the median ranking for each project is found and the projects are divided into quartiles. A table of ranked projects is produced, with preference points, to be shown to the panel. Projects between the 25 and 75 quartiles are considered to have consensus; projects outside of this should be discussed. Once the reasons for the differences in ranking are announced, the ranking process is repeated; and

10. **Re-rank the projects until the result stabilises**: all (or only some) projects will come to consensus. Further discussion is unnecessary once the results become fixed. The ranking table is presented to the panel for their final decision.

The Delphi technique is particularly appropriate when there is a strong divergence of opinions between experts. It can be undertaken formally or informally and has all the advantages of group decision making without the problems of over-dominant group members or political lobbying. An added advantage is that Delphi works as an informal, subjective model where the decisions are based on an opinion, and can be directly converted to a formal model when the data are more knowledge-based (Cline, 1997).

Clearly, the major disadvantage of the Delphi technique is that it does not allow group interaction. It can also be time-consuming and depends upon the co-operation of the experts in responding promptly yet take the time required to complete them correctly. In order to counter these (to some extent), a number of variant techniques have evolved to address some of its shortcomings. These include:

- policy Delphi: aims at generating the greatest possible number of opposing views on the resolution of a problem rather than eliminating alternative solutions to achieve consensus. The focus is to suggest as many courses of action as possible instead of narrowing down to the one considered most likely;

- decision Delphi: provides the names of all panellists so that the prestige of others acts as an incentive to put a lot of thought into responses (and to avoid the potential for apathy that may occur when there is complete anonymity). Quasi-anonymity is maintained since responses are not linked with names. Decision-makers (not experts) are polled as these are the people with the power to make changes; and

- qualitative Control Feedback: attempts to overcome the Delphi's reliance on qualitative evaluation of options by providing opportunities for respondents to include written qualitative additions to their numerical weights.

There are numerous publications discussing the Delphi Technique covering a variety of subject areas. The interested reader may consider Dalkey (1967), Brown (1968), Linstone & Turoff (1975), Cicarelli (1984), Benaire (1988), Woudenberg (1991) and Rowe & Wright (1999).

3.3.6 The Value of Additional Information

The gathering of information (which includes scientific data) is clearly crucial in determining what the risks actually are and what may be the associated human health and environmental benefits. However, it is highly improbable that any SEA can incorporate all the information that can be gathered in order to produce an appraisal. In practical terms, there will always be time and cost constraints when undertaking an appraisal. A decision is required, therefore, to determine at which point there is 'enough' information to produce robust conclusions.

This inevitably means that there will be uncertainty associated with key assumptions and variables underlying the whole risk reduction procedure. A decision should thus be made as to whether it is worth the additional expenditure to reduce the uncertainty associated with key variables.

With this in mind, 'value of information' analysis (VOI) can be used as part of an economic appraisal to determine whether or not it is more appropriate to base a decision on incomplete or inaccurate information or to delay any actions until sufficient data have been collected to minimise (or at least reduce) the key uncertainties. Value of information analysis allows additional data collection with a commitment to consider taking a decision in the future as an additional, viable option within the decision making process. For more detailed discussions see, for example, Stigler (1961), Weitzman (1979), Lippman & McCall (1982), Laffont (1989), Feeney & Greives (eds, 1994), Parker (1994), Thompson & Graham (1996) and Lawrence (1999).

The value of this additional information is derived by considering the expected outcomes with and without the information. The approach relies upon the use of conditional probabilities and expected values. If the additional cost of gaining the extra information is outweighed by its expected health or environmental benefits (in terms of improving the decision, *e.g.* decreasing the costs of risk reduction or increasing health and environmental benefits), then it is worthwhile obtaining the additional information.

It may be the case that an expert group is set up to determine how much 'better' (in terms of actually reducing the risks) one risk reduction option is over another. Such a group would then score each option with a view to deriving its overall chance of success in reducing risks.

Of course, there may be good reasons why some gaps in the data and hence uncertainties exist, for example, owing to the costs of obtaining the data, time constraints, political reasons, and so on. The political dimension may be particularly important when decision makers are considering delaying a risk reduction measure in order to gain additional information. Pressure from political parties, lobby groups, the media and stakeholders in general may create significant pressure for decision makers (and society) to accept the uncertainties in favour of short-term implementation.

3.3.7 Portfolio Analysis and Robustness Analysis

Although the above techniques may be those most commonly used in the context of chemical risk analysis, techniques used in other fields may also be of value, in particular, complementary tools such as risk-benefit plotting, portfolio analysis and robustness analysis.

Risk-Benefit Plotting

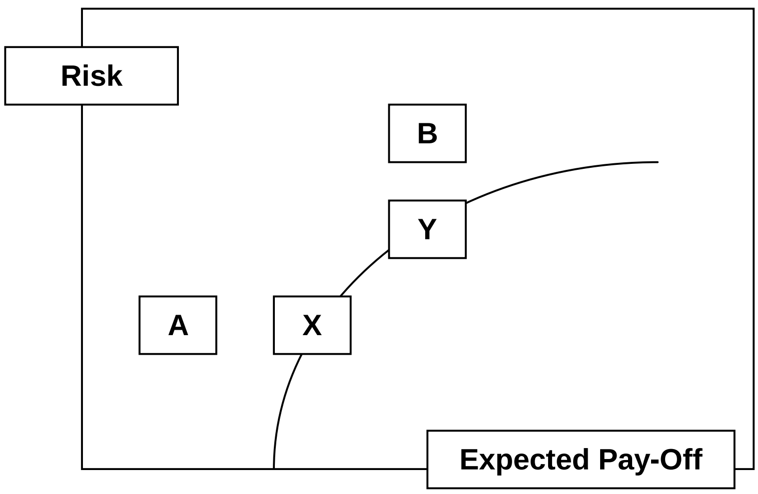
In spite of its limitations, use of expected values is the most convenient way of choosing a single figure to describe the impacts of adopting a given option. But, as it does not incorporate any indication of associated risk involved, additional measures are often required. This can be achieved by:

- taking the worst thing that could happen if that option is chosen – the downside risk or worst case outcome;
- examining the **range** of things that could happen; and/or

– considering the **distribution** of things that could happen, using statistical measures such as variance.

In other words, the risks and expected benefits of all the options must be brought together if we want a full picture of the uncertainty affecting a decision. One simple way of doing this is through the use of a risk-benefit graph as shown in Figure 3.1.

Figure 3.1: Risk-Benefit Plotting



The top left hand part of the graph represents the area to be avoided, where options have low expected pay-offs, and run high risks. The bottom right-hand part of the graph includes those extremely attractive but elusive options that give a high expected pay-off, and involve little risk. Somewhere between these two extremes will lie a line representing combinations of risk and expected pay-off which are the best that can be hoped for in a particular decision. Options X and Y in Figure 3.1 are contained in this set - sometimes called the efficient frontier. Any option that is positioned on the line dominates any other option which lies beyond the line towards the top left hand part of the graph. So option X dominates option A (X gives a better pay-off for the same risk) and option Y dominates option B (Y has a lower risk for the same pay-off).

By plotting the various alternatives as described above, a risk-benefit graph allows us to assess the risks associated with an option, either in isolation, or by comparing its risk-benefit characteristics against other options. On the basis of such information, we may then be able to make a judgement as to the ‘best’ option.

Portfolio Analysis

In some cases, however, we may want information that goes beyond consideration of the risks arising solely from a particular chemical or family of chemicals. Decisions may instead be concerned with the combined risk-benefit position of decisions taken across a range of different chemicals that are all used in the same production processes (as substitutes for one another or as complementary chemicals). In particular, if there is concern that a number of fairly high risk options have already been adopted, then decision makers may want to ensure that another high risk decision is avoided. Conversely, a series of low risk but low benefit decisions may have been adopted and decision makers might look favourably on taking a lower cost but higher risk decision. Expressed in the language of the financial investor, in such cases the basis for making a decision should be based on an examination of the total portfolio of risk management decisions.

The expected benefit from a portfolio is simply the sum of the expected benefits from the individual options or activities that comprise it. The portfolio's degree of risk, however, will depend on three factors:

- the degree of risk associated with each individual activity/option in the portfolio;
- the amount invested in each activity/option; and
- the degree of correlation or connection between the activities/options.

This last point is important. Suppose two activities are subject to exactly the same set of uncertain and uncontrollable factors. If the pay-off from one activity declines, then so will the pay-off from the other – both eggs are in the same basket. But, if the uncontrollable factors in the decision influence the two activities in opposite directions, then a reduction in the pay-off from one activity will be accompanied by an increase in the pay-off from the other. So the contribution of an additional investment to the riskiness of the group of investments will depend on the extent to which the risks are correlated. This can range from:

1. perfect positive correlation – if the uncontrollable factors cause an increase in the pay-off from one system, there will certainly be an increase from the pay-off from the other, and vice versa; to
2. totally uncorrelated – a change in the pay-off from one investment gives no clue about whether the pay-off from the other will increase, decrease or remain unchanged; or
3. perfect negative correlation – if the uncontrollable factors cause an increase in the pay-off from one investment there will necessarily be a decrease in the pay-off from the other and vice-versa.

Robustness Analysis

Robustness analysis is related to sensitivity analysis but is aimed instead at examining how robust a solution is in relation to uncertain developments in other areas (related decision areas), and in particular in relation to unlikely or unexpected events.

Robustness analysis was first developed during the late 1960s to provide a more rigorous measure of the quality of options that makes them more or less likely to work out well in the future. It is based on the observation that, in the face of uncertainty, decisions tend to be deferred as long as possible. Thus, when considering a set of related decisions, it is inevitable that they will not all be taken at the same time - that they will, in fact, fall into a sequence of some sort. As a result, robustness analysis applied to the initial set of choices ('or action set') can be helpful.

Essentially, the analysis is based on the principle that when there is substantial uncertainty, the first decision in a sequence should be taken so as to maximise the number of feasible and satisfactory solutions that are left open. By adopting such an approach, one can expect that the initial choice of options will leave some flexibility for the future. This may obviously be an important concept for chemical risk management, particularly where the aim is to reduce risks from a range of chemicals which can act as substitutes.

Using this definition, the measure of robustness is given by:

$$= \frac{\text{Number of satisfactory full solutions achievable from the initial decision}}{\text{Total number of satisfactory full solutions}}$$

The aim then is to identify the option that maximises the above ratio, as this is the most robust option. Within such an analysis, care must be taken not to include nearly equivalent alternatives among the possible decisions - this will weaken their apparent robustness. The most robust initial decisions can then be used as a set from which to choose - perhaps using other criteria.

This approach can obviously be elaborated to include alternative futures, and the relative desirability of future options. In addition, some quite sophisticated mathematics can be applied, taking probabilities into account - and such methods are used in developing, for example, artificial intelligence.

3.4 Presenting the Results

Uncertainty may affect predictions of both the costs and benefits associated with different risk reduction options. However, information on the likely level of uncertainty affecting a decision may not be needed by all users, while others may seek details of the degree of uncertainty to be given precisely.

It will therefore be important that steps are taken to ensure that the importance of uncertainty to the end decision is effectively communicated. This should include providing:

- an appreciation of the overall degree of uncertainty and variability and of the confidence that can be placed in the analysis and its findings;
- an understanding of the key sources of variability and uncertainty and their impacts on the analysis;
- an understanding of the critical assumptions and their importance to the analysis and findings; this should include details of any such assumptions which relate to the subjective judgements of the analysts performing the analysis;
- an understanding of the unimportant assumptions and why they are unimportant;
- an understanding of the extent to which plausible alternative assumptions could affect any conclusions; and
- an understanding of key scientific controversies related to the assessment and a sense of what difference they might make regarding the conclusion.

The guiding principles when assessing and describing uncertainty are transparency and clarity of presentation. Although the extent to which uncertainty is analysed and presented will vary according to the intended audience, some general minimum requirements can be applied to most economic analyses. Uncertainties may be communicated in a number of ways, including by:

– using '+' or '-' statements;

– stating statistical parameters such as standard deviation or confidence limits; and/or

– graphical means such as PDFs, cumulative frequency or probability distributions.

Confidence intervals are a particularly useful means of describing uncertainty as they can illustrate the precision of estimates and provide bounds to the values used in sensitivity analysis.

At the very minimum, each SEA should set out what the key risks or uncertainties are and how they may affect the overall appraisal. Knowledge of such factors enables decision makers to place confidence (either explicitly or implicitly) in the recommendations from a SEA as part of the process for introducing risk reduction options.

3.5 The ZOZ Case Study and Sensitivity Analysis

Given the nature of the results derived in the ZOZ SEA, it was decided to undertake sensitivity testing on the key parameters. In some cases, this involved taking ranges from probability distributions developed through modelling exercises, in other cases expert opinion was sought to identify the most likely range, while in still others the variation in figures provided by industry gave the basis for the analysis. In some cases, Monte Carlo analyses were run to develop improved information on the likelihood of different outcomes occurring.

The results of some of the sensitivity testing are presented in Table 3.1. These reflect the following variations in assumptions.[72]

– Scenarios 1 and 2: a halving and doubling of environmental benefits respectively to account for uncertainty in the underlying dose-response functions;

– Scenario 3: a doubling of the monetary value of the health benefits to workers given that estimates from the lower end of the literature were adopted (noting that the total health benefits may be even greater if there were also benefits to consumers from reducing exposure to ZOZ in products or to the general public from reducing exposure to ZOZ in the environment); and

– Scenarios 4 and 5: a 50% reduction and a 50% increase respectively in costs to reflect the range of figures coming out of the survey of industry.

As expected, the analysis is sensitive to these changes in assumption. Table 3.1 sets out the impact that these changes have on the calculated net present values. These results were achieved by altering the parameter in question whilst holding the other parameters constant, and it can be seen that for some of the sensitivity tests the net present value for the marketing and use option becomes negative. Examining the table suggests that although the marketing and use option still performs well, it no longer always has the highest net present value. Indeed, the command and control option may be more robust under the various uncertainties.

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72 The decision was made to focus on 'broad-brush' sensitivity tests for this SEA.

Table 3.1: Summary of Sensitivity Testing on Net Present Value Estimates ($ millions)					
Option	**Environmental Benefits**		**Health**	**Costs**	
	Scenario 1	**Scenario 2**	**Scenario 3**	**Scenario 4**	**Scenario 5**
Voluntary Agreement	-25.3	19.7	8.1	19.05	-39.6
Product Labelling	-3.85	2.9	N/a	1.45	-4.6
Training and Equipment	N/a	N/a	20.4	10.2	-6.2
Product Tax	-20.9	25	N/a	12.5	-23.7
Marketing and Use Restrictions	-19.7	175.4	66.5	98.3	-7.7
Command and Control	-11.9	116	N/a	58	3.4

4. KEY ISSUES IN PRESENTATION

4.1 Requirements of Decision Makers and other Stakeholders

Any reports should be clearly written (often for very diverse audiences), as transparent as possible (with key assumptions stated), and provide a logical route through the issues to arrive at a risk reduction proposal. The audience (including decision makers, stakeholders and the general public) should be able to understand both the issues (including social, economic and environmental impacts) and the chosen risk management option(s).

The primary audience for the findings of a SEA is decision makers, who will be seeking information in line with established government appraisal processes. Within this context, decision makers require information on which of the options for addressing the risk management issue is the optimum, given the agreed decision criteria. However, decision makers also need a clear understanding of the trade-offs between different options and the extent of uncertainty associated with the SEA findings. Decision makers may also be concerned about the implications of risk management measures on distribution, trade and competitiveness, as discussed in Part 3.

In most cases, however, decisions on chemical risk management are not made in isolation. Instead, the SEA acts as the basis for discussion between decision makers and stakeholders. Information from the SEA thus needs to be presented in a form that communicates to the various stakeholders and focuses the discussion on the merits (or not) of particular measures. It should enable the development of a shared understanding of the implications of taking action (or not taking action).

Both decision makers and other stakeholders need the key findings of the SEA to be presented in a clear and precise manner, including analysis assumptions, data sources and any uncertainties contained within them. However, between decision makers and stakeholders, there are likely to be differences in the levels of understanding of the process and techniques of SEA, and of the approach to risk management. Decision makers will normally have been involved in setting the original boundaries of the study and the decision criteria, and they should therefore have some understanding of the SEA process. In contrast, the level of understanding held by the range of stakeholders is likely to vary.

The stakeholders likely to be active in the decision - manufacturers and users of the substance for which control measures are being assessed - are likely to have been involved in the preparation of the SEA. They provide a key source of information on current uses of the substance, the potential costs associated with control measures and possibly the exposures associated with its use. There is thus scope for ensuring that these stakeholders understand the approach being taken to the assessment and the control measures being considered. Other groups of stakeholders, such as those who may benefit from certain forms of risk reduction, may not have an understanding of the basis of the SEA, the decision criteria, and the degree of certainty inherent in the techniques used will need to be explained. These stakeholders need to be engaged to the point that they do understand. In general, though, it must be realised that stakeholders are likely to be most concerned with how a risk management decision will impact on them personally, rather than with the overall balance of costs and benefits.

By its nature, chemical risk management has the potential to generate concerns among the general public and, particularly, among those who are, or consider themselves to be, potentially affected by chemical risks. Analysing and presenting information from the SEA in a way that addresses these concerns will be of significant value to stakeholders and to decision makers.

4.2 Key Issues in Presentation

4.2.1 General Principles of Data Presentation

An OECD workshop on risk communication (OECD, 2000) produced some interesting results and conclusions that are relevant for the purposes of this guidance. Some of the key points from the workshop emphasised the need for:

- the use of plain language and the shaping of key messages to suit the target audience;
- the use of short statements on options of risk management decisions;
- the use of quantitative risk comparisons only when these can enhance understanding;
- the description of uncertainties, gaps in the data and the potential for errors; and
- the development of a ‘toolbox’ for effective and efficient risk dialogue (such as the identification of communication tools that are available and an understanding of their relative strengths and weaknesses).

In communicating risk in a SEA, there is a need to differentiate between the intrinsic hazards of a substance and the risk associated with the different products containing that substance in order to differentiate between safe uses of a hazardous substance and those that are not.

4.2.2 Clarity, Transparency and Achieving a Balance in Reporting

Decision makers and stakeholders need the key findings of the analysis to be presented in a clear and precise manner, stating assumptions, data sources and any uncertainties contained within them. It is essential, therefore, that both the analysis and any conclusions reached are transparent. Not only will this help ensure that the results are correctly interpreted, but also that users of the results will have confidence in them and will be able to understand whether there are any significant gaps in the analysis. Reporting also needs to achieve a balance, neither overstating nor understating the costs and benefits of particular options.

Clarity

A number of approaches have been suggested to improve the clarity of risk management reporting for stakeholders (Kamrin *et al*, 1997). These include:

- describing costs, risks and benefits to the stakeholders concerned, not just society in general;
- describing the risks associated with alternatives, as well as the substance itself (or uses or products containing the substance);

- describing how stakeholders can get involved in the decision-making process; and
- providing information to help stakeholders to evaluate the risk.

Helping stakeholders to understand and evaluate risks could include providing answers to the following common questions:

- how much of the substance is the stakeholder actually exposed to (across its full range of uses)?
- what is the likelihood of accidental exposure? What safety measures are in place?
- what is the legal standard for the substance? Is it controversial or widely accepted?
- what health or environmental problems is the standard based on? Should other problems be considered?
- what is the source of the risk information? Is it reputable? Do other sources concur?
- were the studies done on a population similar to the stakeholders?
- what are the benefits of the substance? What are the trade-offs? and
- how does the risk compare with other risks the stakeholder faces?

Care is needed when making comparisons with other risks the stakeholder is facing. When an involuntary risk, such as chemical exposure, is compared with a voluntary risk such as smoking or other lifestyle choices, this tends not to influence people's perceptions. Equally, risks tend to accumulate in people's minds so that however small a new risk is, it is seen as adding to the overall burden of risks. The more useful types of risk comparisons include:

- comparisons of similar and everyday risks, *e.g.* the risk of ill health, the risk of injury in a road accident, *etc*.;[73]
- comparisons of risks with benefits from use of the chemical;
- comparisons with alternative substances;
- comparisons with natural background levels; and
- comparisons with regulatory standards.

Transparency

Transparency as a term simply refers to ensuring that the steps leading up to the conclusions from a SEA are explained in an understandable manner. All assumptions, calculations, uncertainties, base data

[73] For further reading on the perception of risk and the differing interpretations by experts and the public, see Slovic *et al* (1980) and Pidgeon *et al* (1992).

and conclusions from the analysis are set out in a manner that allows the audience to follow an audit-trail throughout the SEA to the recommended risk management measure.

Achieving a Balance

When dealing with such complex issues, there is a real danger of excessive detail, often at the expense of readability, clarity and transparency. It may be important to consider the type of audience for which the SEA is being developed, for example:

- decision makers may wish to only read a summary of the SEA, keeping technical details down to a minimum;
- industry may wish to have a very in-depth presentation of the usage of a particular hazardous substance together with the costs of alternative risk management measures;
- environmental groups may wish to see great detail on the interaction of the hazardous substance (and its substitutes) with the environment and the impact on human health; whilst
- the general public may want to understand how any risk reduction measure impacts upon their lives via the products they consume and/or the prices they pay.

Any presentation of findings, therefore, will need to balance the different requirements of the various audiences; a difficult task to achieve. However, in general terms, clear, concise and transparent presentation will meet the needs of this diverse audience. This includes providing adequate contextual information in both qualitative and quantitative formats.

4.3 Content of Typical SEAs

Guidance on the structure of a SEA report will of course very much depend upon the type of approach for the appraisal. The different appraisal approaches will result in different outputs (*e.g.* cost-effectiveness ratios, net present values, and semi-quantitative/qualitative information) and the structure of the SEA should reflect this. This section sets out a generalised approach to act as a guide for the type of information to be included in a report. Box 4.1 shows the generalised list of contents, with some indication as to the elements contained within each part.

Box 4.1: A Typical Contents List	
Section	**Description**
Executive Summary	The summary should set out in as non-technical way as possible: the problem; the approach; the advantages and disadvantages of risk management options; and the conclusions.
Background to the Problem	This section should set out the concerns of decision makers and stakeholders. It should set out the current baseline conditions and actual risk should be quantified as far as possible, in terms of both the likelihood of exposure and the consequences (for all pathways). Extent of use, markets and industrial organisation should also be set out.
Risk Management Context	The current context should be set out, together with what options are available under current national legislation. Consideration should also be given to instruments applying in other jurisdictions (local and international).
Existing Risk Reduction Measures	It may be the case that the hazardous substance in question is already subject to risk reduction measures. These should be stated together with their current effectiveness.
Available Risk Reduction Options	What further options are available to limit exposure to tolerable limits should be discussed. Options may include voluntary agreements, limit values or phase-outs. A long list should be established together with details of how a short list in the ‘full’ SEA was developed.
Costs of Available Options	All risk reduction measures will usually have costs associated with them. This section should set out what the costs will be and on whom they fall. Special attention should be paid to inequitable distribution of costs and vulnerable groups within society.
Benefits of Available Options	The benefits of each risk reduction option should be considered and reported. Benefits will tend to be human health and/or environment related, but may include technology forcing and other effects.
Sensitivity Analysis	This section should consider assumptions underlying the analysis in an effort to ensure that the SEA as a whole is robust and defensible.
Comparative Assessment of Options	The most preferred option, after taking into account uncertainties, costs and benefits (and their distribution), should be set out. This would also include how risk reduction measures would be implemented in practice.
Recommendations	Usually a short section summarising the preferred way forward.
Appendices containing all specific data	Key data should be included either in the main text or as supporting information within appendices.

In short, the aim of a SEA report is to clearly set out:

- what the risks being addressed are and thus the aims of risk management;
- what risk management options are feasible;
- the advantages and disadvantages of these options; and
- the preferred approach to managing the risks of the hazardous substance(s) in question, taking into account the relative costs and benefits of the different options.

Two other features of a good SEA, which are actually designed to help improve the methodology itself, are:

- making available any new data (*e.g.* from a valuation exercise) to other SEA practitioners; and
- the use of *ex-post* appraisal to examine flaws in previous SEAs.

With regard to the latter, recommendations on how to undertake *ex-post* or retrospective analyses are provided in OECD (1999).

4.4 Summary

To summarise, the findings of a SEA should be presented in a manner that is:

- readable;
- understandable;
- clear;
- transparent;
- disseminated appropriately; as well as
- robust and defensible.

On the whole, the message is that socio-economic analysis will be most effective in the risk management process when it is communicated effectively and there is a great deal of trust amongst all participants. To facilitate the success of the overall process, trust can be improved through openness, transparency and accountability.

These are clearly fundamental points to be borne in mind - throughout the SEA process - by all stakeholders, from decision makers to analysts and the general public.

5. REFERENCES

Benaire M (1988): Delphi - and Delphlike Approaches with Special Regard to Environmental Standard Setting. Technological Forecasting and Social Change. Vol. 33, pp. 149-158.

Brown B (1968): Delphi Process: A Methodology Used for the Elicitation of Opinions of Experts. Santa Monica, The RAND Corporation.

Cicarelli J (1984): *The Future of Economics: A Delphi Study*. Technological Forecasting and Social Change. Vol. 25, pp. 139-157.

Cline A (1997): Prioritization Process Using Delphi Technique, available from www.carolla.com/wp-delph.htm

Dalkey N & Helmer O (1962): An Experimental Application of the Delphi Method to the Use of Experts. Santa Monica: the RAND Corporation.

Dalkey N (1967): Delphi. Santa Monica: the RAND Corporation.

Drummond MF *et al* (1997): Methods for the Economic Evaluation of Health Care Programmes, Oxford Medical Publications, Oxford: Oxford University Press.

Feeney M & Grieves (eds) (1994): The Value and Impact of Information. London: Bowker Saur.

Gold MR *et al* (1996): Cost-Effectiveness in Health and Medicine, Oxford: Oxford University Press.

Hammersely JM & Handscomb (1964): Monte Carlo Methods. New York: John Wiley & Sons.

Jones-Lee MW and Loomes G (1995): *Discounting and Safety*, Oxford Economic Papers, Vol.47, pp.501-512.

Kalos MH & Whitlock PA (1986): Monte Carlo Methods, Volume 1: Basics. New York: John Wiley & Sons.

Kamrin MA *et al* (1997): A Journalist's Handbook on Environmental Risk Assessment, Pasedena, California: Foundation for American Communications.

Krahn M & Grafni A (1993): Discounting in the Economic Evaluation of Health Care Interventions, in Medical Care, Vol.31, pp.403-418.

Laffont J-J (1989): The Economics of Uncertainty and Information. Cambridge: MIT Press.

Lawrence DB (1999): The Economic Value of Information. New York: Springer Verlag.

Linstone H & Turoff M (eds) (1975): The Delphi Method: Techniques and Applications. London: Addision-Welsey.

Lippman S & McCall J (1982): *The Economics of Uncertainty - Selected Topics and Probabilistic Methods*, in Arrow K & Intrilligator (eds) (1982): Handbook of Mathematical Economics. Vol. 2. Ch. 6. North Holland Press.

Morgan MG & Henrion (1990): Uncertainty - A Guide to Dealing with Uncertainty in Qualitative Risk and Policy Analysis. Cambridge: Cambridge University Press

OECD (1992): Project and Policy Appraisal: Integrating Economics and Environment, Paris: Organisation for Economic Co-operation and Development.

OECD (1999): Guidance for Conducting Retrospective Studies on Socio-Economic Analysis, OECD Environmental Health and Safety Publications, No 11, Paris: Organisation for Economic Co-operation and Development.

OECD (2000): Proceedings of the Workshop on the Convergence of risk assessments and Socio-Economic Analyses to Better Inform Chemical Risk Management Decisions, May 1-2, Arlington, Virginia.

Parker J & Houghton J (1994): *The Value of Information: Paradigms and Perspectives* in Proceedings of the 57th Annual Meeting of the American Society for Information Science.

Pearce D *et al* (1989): Blueprint for a Green Economy, London: Earthscan Publications.

Pidgeon N *et al* (1992): *Risk Perception*, in Risk: Analysis, Perception and Management, pp.89-134, London, The Royal Society.

Pindyck RS (2000): *Irreversibilities and the Timing of Environmental Policy*, Resource and Energy Economics, Vol. 22, No. 3, pp23-259.

Postle *et al* (1984): *The Worth of Monitoring at Landfills*: *A Case Study*, Research Bulletin Number 687, Massachusetts Agricultural Experiment Station, Amherst: University of Massachusetts at Amherst.

Ripley BD (1987): Stochastic Simulation. New York: John Wiley & Sons.

Rowe G & Wright G (1999): *The Delphi Technique as a Forecasting Tool: Issues and Analysis*. International Journal of Forecasting. Vol. 15. No. 4. pp. 353-75.

Slovic P, Fischhoff B & Lichtenstein S (1980): *Facts and Fears: Understanding Perceived Risk*, in Societal Risk Assessment, Schwing R & Albers WA eds., pp-181-124, New York: Plenum.

Stigler G (1961): *The Economics of Information* in Journal of Political Economy, No. 69. pp. 1 to 225.

Thompson & Graham (1996): *Going Beyond the Single Number: Using Probabilistic Risk Assessment to Improve Risk Management*, in Human and Ecological Risk Assessment, Vol.2, No.4, pp.1008-1034.

Turner RK (1994): Environmental Economics. London: Harvester Wheatsheaf.

US EPA (1999**):** Guidelines for Preparing Economic Analyses, Washington DC: United States Environmental Protection Agency.

Viscusi WK & Moore MJ (1989): *Rates of Time Preference and Valuation of the Duration of Life*, Journal of Public Economics, Vol.38, No.3, pp.297-317.

Weitzman M (1979): *Optimal Search for the Best Alternative*, Econometrica, Vol. 47. No. 3. pp. 654-.

Woudenberg F (1991): An Evaluation of Delphi in Technological Forecasting and Social Change, No. 40, pp. 131-150.

ANNEX TO PART 4: DECISION MAKING UNDER UNCERTAINTY

A1. INTRODUCTION

Many risk management decisions will have to be made under conditions of uncertainty. Not only does this mean that analysts need to pay particular attention to reporting the nature of such uncertainties, but also as to how decision makers may wish to take such uncertainties into account within their own decision making processes.

In practice, when faced with a decision that is clearly to be made under conditions of uncertainty, there are generally three ways to proceed:

1. to ignore the uncertainty, in other words to make the best possible guess on the strength of the information available and to carry on with the assessment as though uncertainty did not exist;

2. to proceed as above, but make allowances for the possibility that the estimates may be optimistic or pessimistic; or

3. to explicitly incorporate uncertainty into the assessment process.

The discussion that follows starts by identifying simple forms of decision techniques and moving on to the more complex techniques.

The simpler techniques are those that would normally only be used in cases where the decision is characterised by only a few key uncertainties. These essentially make allowances for the possibility that the analysis results may be either pessimistic or optimistic. The more complex ones are those that explicitly incorporate uncertainty into the assessment results. The latter are more commonly used as part of chemical risk SEAs.

The main types of techniques that might be adopted by decision makers in the face of uncertainty are:

- **Hedging and flexing**: these assume that either the worst (or the best) outcome will occur for each option and then to choose the option which gives the least bad outcome (or the best possible outcome);

- **Regret, minimax and maximin**: under a **regret** approach, the decision maker selects either the option that would cause the minimum embarrassment in the future; in other words, the option which minimises the maximum regret is selected. In practice, the decision rules used under a regret based approach are those of *maximin* and *minimax.*

- **Expected values**: in contrast to the above, the expected value directly incorporates estimates of the likelihood of a particular outcome occurring into the analysis results. In other words, estimates of costs and of benefits would be presented in probabilistic terms, taking into account the range of uncertainties underlying them.

The first two types of techniques are, in general, only likely to be applicable in cases of 'near' certainty, where there are few uncertainties surrounding the choice of options. As a result, it is likely that expected values and other probabilistic decision criteria will be called upon for many SEAs.

A2. SIMPLE CHOICE TECHNIQUES

A2.1 Hedging and Flexing

The most straightforward way of dealing with uncertainty is to either assume that the best (or the worst) outcome will occur for each option and then to choose the option which gives the best possible outcome (or the least bad outcome). In other words, the decision maker can adopt an optimistic (or pessimistic) approach towards decision making.

Where a decision maker has adopted a pessimistic stance, this is sometimes referred to as **hedging** because one is foregoing the best outcome in order to avoid the worst (as in 'hedging your bets'). An alternative to hedging is that of **flexing**. Under this type of approach, the option which would give the best possible results is chosen, but methods are explored that would enable the decision to be modified if the worst outcome did happen, *i.e.* the environment is monitored to detect any signs of the worst outcome occurring.

Adopting a flexing approach depends on:

- the uncertainty factors that are developing slowly relative to the time needed to modify the decision;
- it being feasible to modify the options; and
- the costs of modifying the options (*i.e.* the decision) being less than the costs of choosing an alternative, less uncertain option.

Such conditions will not hold for many decisions, including decisions with regard to chemical risks where the environmental and health costs will exist in the present and may be significant.

A2.2 Regret, Maximax and Maximin, Minimin and Minimax

A more sophisticated (hedging based) approach has been recommended as a result. Under a pure **regret** approach, the decision maker selects the option that would cause the minimum embarrassment in the future. In other words, the option which minimises the maximum regret is selected. This approach (which many decision makers find intuitively attractive) is based on a simple but extremely valuable question:

If we decide on one particular option, then with hindsight, how much would we regret not having chosen what turns out to be the best option for a particular set of circumstances?

The question being asked here is closely related to the economist's concept of 'opportunity cost'; that is, by choosing one alternative course of action, what opportunity are we foregoing by not choosing another course of action? Because the regret approach is illustrative of a moderate attitude towards risk, it is necessary to present both opposite ends of the conservatism/risk spectrum.

In practice, the decision rules used are those of maximax and maximin when outcomes are expressed in terms of returns or benefits. Alternatively, in the context of outcomes expressed in terms of costs, the decision rules used are of minimin and minimax. The principles underlying each of these decision rules are set out in Table A1.

Table A1: Regret Based Decision Rules

Context of Return or Benefits	Context of Costs
Maximin: alternatives should be evaluated according to the minimum return that each provides under the worst conceivable outcome.	**Minimax**: alternatives should be evaluated according to the maximum cost that each provides under the worst conceivable outcome.
Maximax: alternatives should be evaluated according to the maximum return that each provides under the best conceivable outcome.	**Minimin**: alternatives should be evaluated according to the minimum cost that each provides under the best conceivable outcome.
Regret: for each of the options, the difference in amount from the best result (return or benefit) is calculated and used as the loss incurred for not having chosen the best option.	**Regret**: for each of the options, the difference in amount from the best result (lowest cost) is calculated and used as the loss incurred for not having chosen the best option.

These principles provide an indication of the attitude toward risk. Maximax and minimin are generally associated with optimistic people or risk taker attitudes. At the other extreme, maximin and minimax are characteristic of pessimistic people or risk averse (very conservative) people. While both methods represent opposite ends of the conservatism/risk spectrum, the regret criterion is considered as being between both extremes (*i.e.* moderate conservative). These concepts are easier to understand when shown in the following two contextual examples.

Context of Benefits

Consider a situation where it has been determined that there are three possible policy options with two simple outcomes (such as the exposure to a substance results in premature death). There is a range of benefits associated with the different options and resulting states (*i.e.* exposure or no exposure); these are summarised in Table A2. The three rows (A, B and C) are the three policy options and the two columns are the 'states of nature' (*i.e.* whether or not a premature death occurs) and the monetary figures refer to benefits and not costs.

Table A2: Benefits Associated with a Range of Outcomes

A	$130,000	$400,000
B	$140,000	$260,000
C	$80,000	$90,000
Policy	**Premature Death**	**No Premature Death**

Clearly the decision maker is faced with a choice under uncertainty. Whether the decision maker is overly conservative or not will determine the use of either the maximin or maximax decision rule. In the case of maximin, the decision maker examines each policy and selects the minimum value from each row. Then, the decision maker selects the maximum of these minimum values ($140,000). By doing so, he/she adopts a pessimistic view of the outcome of a policy. The chosen option in this case will be B. Table A3 sets out this process.

Table A3: Benefits Associated with a Range of Outcomes

Policy	**Premature Death**	**No Premature Death**	**Row Minima**	**Maximin**
A	$130,000	$400,000	$130,000	
B	$140,000	$260,000	$140,000	**$140,000**
C	$80,000	$90,000	$80,000	

However, this approach clearly suffers from being overly conservative with the potential loss of $140,000 (if there is no premature death - the difference between options A and B). This is the ‘cost of playing safe’ with it resulting in the decision maker losing the opportunity of gaining $140,000 more if the ‘best’ should happen.

At the opposite end, an optimistic view can be illustrated by considering only the maximum of the maximum returns (thus its maximax name). Table A4 shows how the option is selected under this decision rule. The decision maker considers first the maximum returns of each policy (identified by the “Row Maxima” column in the table). Then he/she selects the policy that provides the maximum return of the “Row Maxima” column. In this case, the policy with a return of $400,000 would be the choice (*i.e.* policy A will be selected).

Table A4: Benefits Associated with a Range of Outcomes

Policy	**Premature Death**	**No Premature Death**	**Row Maxima**	**Maximax**
A	$130,000	$400,000	$400,000	**$400,000**
B	$140,000	$260,000	$260,000	
C	$80,000	$90,000	$90,000	

The two methods represent opposite ends of the conservatism/risk spectrum. The maximax approach is exceedingly optimistic, whereas the maximin approach is exceedingly cautious. Both approaches share the disadvantage of not considering the likelihood of different states, so any potential extra return may be foregone to avoid an unlikely loss. In addition, both can also be influenced by irrelevant considerations. In order to minimise any benefit forgone, the regret approach may be adopted.

The regret approach is obtained by calculating for both outcomes the benefits forgone (or losses). For instance, if the outcome is some premature death, the best outcome is $140,000 obtained by the policy B. If however, policy C was selected instead, the loss of having taken this non-optimal policy will be $60,000, that is the difference between the best return ($140,000) and the return of the selected policy (C), all under the premature death outcome. This loss (or benefit forgone) is called regret. Similarly, regrets should be constructed for the outcome of no premature death. In that case, the best return is $400,000. Selecting policy B under the outcome of no premature death leads to a regret of $140,000 (*i.e.* that is $400,000 less $260,000).

In practice, and due to the organisation of the table of benefits, the regret calculation is based on working by column (*i.e*. under each outcome). The result of the regret is a new table (called table or matrix of regrets, see Table A5). From this new table, the decision maker looks only at the regret maximum for each outcome (these are respectively $60,000 and $310,000). The decision rule is to select the minimum of these regret maxima, with this being policy C having a regret of $60,000.

These two final steps explain why this method is also known as the minimax regret criterion. It should be noted that whilst this approach is also affected by irrelevant factors, it does have the advantage of not being distorted by changes which would not affect the comparative benefits.

Table A5: Matrix of Regrets			
Policy	**Premature Death**	**No Premature Death**	**Regret Minima**
A	$140,000 - $130,000 = **$10,000**	$400,000 - $400,000 = **$0**	
B	$140,000 - $140,000 = **$0**	$400,000 - $260,000 = **$140,000**	
C	$140,000 - $80,000 = **$60,000****	$400,000 - $90,000 = **$310,000****	**$60,000**
**** = Regret maximum of the outcome**			

Context of Costs

The previous example dealt with returns or benefits. It might be possible in some situations for costs (in the context of these guidelines negative benefits) to be the main area of uncertainty. The corresponding approaches to be used are of minimin, minimax and regrets. Table A6 shows an example with four possible policy options and two simple outcomes (negative health and environmental impacts). This example implicitly suggests that both outcomes are mutually exclusive in their projected impacts.

Table A6: Costs Associated with a Range of Outcomes		
Policy	**Health Impact**	**Environmental Impact**
A	$130,000	$400,000
B	$260,000	$140,000
C	$90,000	$100,000
D	$110,000	$80,000

Again the decision maker is faced with a choice under uncertainty. Whether the decision maker is overly conservative or not will determine the use of either the minimax or minimin decision rule. In the case of minimax, the decision maker examines each policy and identifies the maximum value from each row. This provides the pessimistic view of the resulting policy, based on costs. The decision maker then selects the policy which minimises this maximum value ($100,000). In this case, policy C will be chosen, as indicated in Table A7.

Table A7: Costs Associated with a Range of Outcomes				
Policy	**Health Impact**	**Environmental Impact**	**Row Maxima**	**Minimax**
A	$130,000	$400,000	$400,000	
B	$260,000	$140,000	$260,000	
C	$90,000	$100,000	$100,000	**$100,000**
D	$110,000	$80,000	$110,000	

The minimax approach also suffers from being overly conservative. At the opposite end, an optimistic view can be illustrated by considering only the minimum of the minimum (thus its minimin name) of the costs. The decision-maker considers first the minimum costs of each policy (identified by the "Row Minima" column in Table A8). Then he/she selects the policy that provides the minimum costs of the "Row Minima" column. In this case, the policy with a cost of $80,000 would be the choice (*i.e.* policy D will be selected).

Table A8: Costs associated with a Range of Outcomes				
Policy	**Health Impact**	**Environmental Impact**	**Row Minima**	**Minimin**
A	$130,000	$400,000	$130,000	
B	$260,000	$140,000	$140,000	
C	$90,000	$100,000	$90,000	
D	$110,000	$80,000	$80,000	**$80,000**

The disadvantages mentioned in the previous example made in the context of returns remain valid for these two extreme approaches. The regret can also be viewed here as a moderate conservative approach. In this particular context, the regret represents the "extra" cost (or negative benefits) incurred for not having chosen the best policy.

The regret approach is almost similar as for the previous context. The regret is obtained by calculating for both outcomes the cost increase (or increase in negative benefits). For instance, if the outcome is health impacts, the best result is $90,000 obtained by the policy C. If however, policy B was selected instead, the loss from having taken this non optimal policy will be $170,000, that is the difference between the cost of the policy B ($260,000) and the optimal cost achieved with policy C ($90,000), with regard to the health impact outcome. This loss (or cost increase in this case) is the regret. Similarly, regrets should be constructed for the outcome of environmental impacts. In this case, the best result is $80,000 in costs (or negative benefits). Selecting policy B under the outcome of environmental impacts leads to a regret of $60,000 (*i.e.* that is $140,000 less $80,000).

The matrix of regrets (Table A9) is constructed in almost the same way as given earlier. The decision-maker looks only at the regret maximum of each outcome (with these being $170,000 and $320,000 respectively). The decision rule is again to select the minimum of these regret maxima. The selection will be the policy that provides the regret of $170,000, *i.e.* policy B.

Table A9: Matrix of Regrets			
Policy	**Health Impact**	**Environmental Impact**	**Regret Minima**
A	$130,000 - $90,000 = $**40,000**	$400,000 - $80,000 = $**320,000****	
B	$260,000 - $90,000 = **$170,000****	$140,000 - $80,000 = **$60,000**	**$170,000**
C	$90,000 - $90,000 = **$0**	$100,000 - $80,000 = **$20,000**	
D	$110,000 - $90,000 = **$20,000**	$80,000 - $80,000 = **$0**	
**** = Regret maximum of the outcome**			

A3. PROBABILISTIC DECISION CRITERIA

A3.1 Expected Values

The above approaches go some way towards clarifying the significance of uncertainty to the end decision, but they do not take into account estimates of the likelihood of a particular outcome occurring. The incorporation of such estimates provides a powerful means of managing uncertainty in decision making.

How can the likelihood of a particular outcome occurring be taken into account? In order to examine how this can be achieved, it is useful to start by assuming that most people are risk neutral. Under this assumption, the expected value of an effect can be estimated, taking into account the range of uncertain values. The expected value of an effect (in this case a benefit) resulting from a proposed measure in a given year is:

$$Eb = \varphi\, p_i b_i = p_1 b_1 + p_2 b_2 + \ldots p_m b_m$$

Where p_i represents the probabilities (adding up to 1) of different values of b occurring in i = 1….m scenarios. Expected costs (Ec) would be estimated in a similar way.

Given risk neutrality, the project with the highest expected value should be preferred. For example, suppose the choice is between two risk reduction measures:

- Measure A has an expected value of £2 million with a 50% chance of a NPV of £5 million and a 50% chance of a NPV of –£1 million; and
- Measure B has a certain NPV of £1.5 million.

Taking an expected value approach would lead to the selection of Measure A, based on the assumption that decision makers (and society) should be risk neutral (or indeed a risk taker).

The assumption that most individuals are risk neutral does not generally hold in reality. Instead, most individuals tend to be risk averse.[74] However, it is frequently argued that for many public sector decisions this risk averseness can be ignored because the risks to individuals are small and because of risk pooling and risk distribution. Risk pooling relates to governments being able to spread risks across a large number of projects; where one policy/project under-performs, it is expected to be offset by other policies/projects over-performing. As a result, decision making on the basis of risk neutrality and using expected values will produce the highest aggregate return to society. In addition, where the government

[74] One of the oldest and most important points about uncertain outcomes is the idea that the value people place on such outcomes depend not only on the mean or the expected value of that outcome but also on the variability about that mean. In particular, most people are risk averse, meaning that they value an uncertain outcome less than they do a certain outcome with the same mean. Although problematical at the individual level, this is not so at the societal level. However, as Section 3.2.7 of Part 3 has discussed, there may be a need to adjust willingness to pay values with regards to health impacts to take into account the effect of risk aversion.

bears most of the costs and benefits of the project, the impact on any individual will be small and thus capable of being ignored.

These assumptions do not necessarily hold in the case of chemical risk management, where some portion of the costs are likely to be borne by the private sector and where particular individuals may experience significant benefits (or costs).

Although use of expected values is the most convenient way of developing a single figure to describe the outcomes for a proposed measure, it does not incorporate any indication of the degree of associated uncertainty. For this reason, expected values are normally accompanied by further information on the overall probability distribution of end outcomes. This includes information on confidence limits and standard deviations.