Induced and Opportunity Cost and Benefit Patterns in the Context of Cost-Benefit Analysis in the Field of Environment

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by

Risk & Policy Analysts Limited,
Farthing Green House, 1 Beccles Road, Loddon, Norfolk, NR14 6LT, UK
Tel: +44 1508 528465 Fax: +44 1508 520758
email: post@rpaltd.demon.co.uk

and

Metroeconomica
108 Bloomfield Road, Bath, Avon, BA2 2AR, UK

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1. INTRODUCTION

1.1 Key Characteristics of Environmental Policies

Governments need to make choices and when faced with the protection of either the environment or human health, two truisms must be recognised. The first is that such protection does not come 'free', and the second is that such choices are constrained by limited resources. Decisions makers, therefore, need to balance the benefits of protection against the cost of such protection, especially when resources could be allocated to other policy areas (such as health, education, transport, etc.), which may be able to provide comparable levels of protection or other gains.

What are the costs and benefits associated with environmental policies? In general, they are the same range of effects which may be of concern in any policy appraisal, with the potential characteristics of importance being:

- the scale and relative magnitude of the direct policy effects, and hence potential for such effects to lead to further indirect and secondary effects;
- the level of integration of the affected industry sectors and hence the potential for impacts on related markets;
- the degree to which equity considerations are of concern given the likely divergence between those who would gain and those who would lose;
- related to the above, the degree to which the policy affects all EU countries equally and countries outside the EU;
- the relative significance of both environmental and human health effects, where this includes both the negative and positive effects of the policy;
- the inter-relationship with other policies and policy areas; and
- the timing and duration of policy measures.

The first two characteristics are important in that they help set the boundaries for any policy appraisal. Firstly, they establish a role in establishing the degree to which more than just the direct compliance costs of the policy will need to be taken into account. Secondly, and related to the above, they determine the type of appraisal required. For example, if a policy results in non-marginal changes, then 'conventional' Cost-benefit analysis (CBA), which is based on partial equilibrium analysis, is no longer an appropriate appraisal framework. In such circumstances, a general equilibrium type of approach is required. This type of approach automatically considers the so-called 'wider' effects.

As will be discussed later in this report, where a regulation affects a product acting as a minor input to a production process the analysis is likely to require the consideration of only the direct costs to industry and the direct environmental and human health effects associated with the adoption of an alternative product. However, where a policy affects a product which accounts for a significant share of production costs, and/or affects an industry sector which is highly integrated with other sectors of the economy, then the analysis may need to go wider. In this case, the analysis may need to consider the effects of the policy on the structure and functioning of the whole economy (including direct, indirect and possible secondary effects. Examples abound of both types of environmental policy, with many hazardous chemical management policies falling into the first category and environmental taxes falling into the second category.

With regard to the other characteristics, however, it is more difficult to make any generalisations. The implications of these will vary in importance across policies on a case-by-case basis. Indeed, the first two characteristics are likely to determine the importance of the other aspects.

It is also useful to consider whether policies aimed at protecting the different environmental media (air, water, land) or at protecting human health may have fundamentally different characteristics and thus appraisal requirements. Consider a series of policies:

- a policy aimed at reducing sewage effluent discharges into the aquatic environment by requiring secondary and tertiary treatment;

- a policy aimed at reducing damages associated with acidifying emissions from industrial and power plant sources; and

- a policy aimed at reducing health effects to workers by placing restrictions on the use of a widely used chemical.

The first policy will obviously have direct effects on the associated utilities and if sewage treatment costs increase significantly may also have indirect effects on other industrial sectors. It will also have direct benefits to the aquatic environment, but contrastingly may result in negative effects to the other environmental media. For example, the generation of additional sewage sludge will require some form of disposal, whether to land, through landfill, composting or the use of incineration. Similarly, a policy aimed at reducing acidification related effects will result in direct and indirect effects on industry, but may also have wider macro-economic effects should it affect the energy prices paid by consumers for example. Although it may provide benefits across the range of environmental media (and to a number of different impact categories) and to human health, there will also be negative effects associated with particular treatment technologies and the need to dispose of their residuals (e.g. ash). With regard to the chemical restrictions, as they would affect a widely used chemical, there may be direct effects on a range of industry sectors, but the degree to which these result in indirect effects would depend on the change in costs faced by those sectors. Although the policy should generate direct health benefits for the relevant workers, indirect damages may arise from the introduction of substitute chemicals which may pose their own health or environmental risks.

1.2 The Appraisal Tools

The issues which must be examined within any policy appraisal may therefore be complex, requiring consideration of potential effects at a number of different levels. One of the tools commonly used for this purpose is that of social CBA, which is aimed at determining whether or not the benefits of introducing a policy outweigh its costs. In its fullest sense, it requires the assessment of all the effects of a policy, whether positive or negative and, through the use of market data or specialist valuation techniques, the expression of these effects in a common unit, typically money, to facilitate comparison. The use of CBA is not without its criticisms, however. The first, and often repeated criticism, concerns the need to convert all information on costs and benefits to a monetary value. A number of methodological issues arise in trying to value the environmental and human health effects arising from environmental policies. These issues have two main implications: a) they impact the degree to which such valuation can be achieved; and b) they put into question the reliability which may be placed on such estimates at the current time. Some analysts also object to the assumptions underlying the valuation process on ethical or other grounds. In light of these uncertainties, what are the implications for such analyses, and for the messages they give, concerning the justification of a particular policy?

A second criticism is that such analyses tend to focus on the direct or first order effects of a policy and may therefore exclude consideration of indirect and secondary effects which may be significant. In some cases, indirect and secondary effects may be considerable relative to the measured direct impacts and, therefore, could have a significant influence on the true social costs and benefits stemming from a policy. For example, it is usual practice for CBAs of environmental policies to exclude employment effects (both direct and indirect) generated by the policy - the assumption being that the labour market is flexible and unemployed labour will find employment elsewhere in the economy (hence it is treated as a transfer payment). However, policies which have a significant effect on a particular industry sector (e.g. through constraints placed on particular activities or changes in core input prices) or which result in shifts in activity between sectors, may have significant employment related impacts. Similarly, where a policy would have (non-marginal) impacts on the costs faced by a wide number of industry sectors, there may be macro-economic effects stemming from changes in aggregate supply and demand. If policy appraisal methods are to fully reflect society’s preferences and provide a truer indication of the full net social value of a policy, then these additional impacts may need to be taken into account.

A third criticism is that such analyses tend to be undertaken on an isolated, case-by-case basis. The analysis typically considers a particular policy measure and possibly, alternative options for its implementation. Alternative policy measures may not be considered. As a result, questions have arisen as to the ‘opportunity costs’ associated with particular policies, where these relate to the benefits which would have been generated had the resource inputs to a particular policy allocated elsewhere. With respect to policies with the same objective, this then leads to questions concerning the relative cost-effectiveness of policies that protect the environment and human health. Could greater gains for the environment or human health and safety be achieved for the same investment? Or could a set improvement for the environment or human health and safety be achieved at lower resource costs?

This report tries to address these questions by examining the manner in which the assessment of costs and benefits is undertaken through the various economic appraisal tools available to policy analysts. Although the focus is on the use of 'conventional' CBA, the potential for combining this type of 'bottom-up' approach with the use of 'top-down' modelling techniques, such as input-output analysis and general equilibrium models, is also highlighted.

1.3 Scope of the Report

This study has been commissioned by DGIII (Industry) of the European Commission to examine the cost and benefit patterns of environmental legislation. The Specification requires that the study provides an overview of the main types of direct, indirect and secondary costs and benefits arising from environmental policies and their assessment within a CBA framework. The second aspect of the study concerns questions over the opportunity costs associated with the adoption of one type of policy measure versus the other.

The research undertaken as part of this study has been broken into three different stages:

- the first stage involved a review of the existing literature concerning the limits, in both theory and practice, of the various appraisal tools;
- the second stage then examined three different CBA-type appraisals of environmental policies, with the aim of determining whether or not it was possible to consider the full range of impacts within a 'conventional' CBA framework and to highlight issues related to the opportunity costs question; while
- the third stage considered the question of opportunity costs and how these may be taken into account within current appraisal practice in more detail.

1.4 Structure of the Report

This report presents the findings of all three stages (thus superseding the two previous reports - the Preliminary Review and the Mid-Term Review). The organisation of the report, however, reflects the three different stages of work:

- Section 2 sets out the definitions of direct, indirect and secondary effects and other key terms that are used throughout the report;
- Section 3 reviews the use and limitations of CBA (and to a much lesser degree cost-effectiveness analysis) and the 'top-down' modelling techniques respectively;
- Section 4 discusses valuation of the impacts of environmental and health related policies on a range of societal factors, including industry, consumers, employment, mortality and morbidity etc.;

- Sections 5, 6 and 7 present the three case studies, starting with the setting of air quality targets, the definition of waste management strategies and the regulation of a hazardous substance;

- Section 8 brings together the findings of the previous sections to discuss the concept of opportunity costs and its evaluation in environmental policy making; and

- Section 9 presents our conclusions and recommendations for further research.

2. DEFINITIONS

2.1 Introduction

This section outlines the definitions to be used throughout this report. It was believed necessary to have such a section given the wide variation in understanding and use of economic terms and in order to avoid any confusion in later sections or stages of the study. The terms referred to here are usually related to cost-benefit analysis, but they may be used in reference to cost-effectiveness analysis as well. As the focus of this report is cost-benefit analysis in relation to environmental regulation, the definitions have been chosen to reflect this.

Definitions are provided for the following terms:

- externalities;
- cost-benefit analysis;
- cost-effectiveness analysis;
- direct effects;
- indirect and induced effects;
- secondary effects; and
- opportunity costs and benefits.

Before proceeding however, it is necessary to make a distinction between ‘economic’ effects and ‘environmental’ effects. The definitions presented in this section, in particular those relating to direct, indirect and secondary effects, concern solely the former. This distinction is necessary because environmental effects may also be classified as direct or secondary (and even tertiary, quaternary effects), depending on the endpoint on the cause-effect chain one is referring to. However, it is not possible to generalise that a direct ‘economic’ effect will correspond exactly to a direct ‘environmental’ effect. For example, fish kills may be considered a tertiary impact on the cause-effect chain for SO2, yet the value of reduced fish kills may be taken as one of the direct benefits of a programme to limit SO2 emissions.

2.2 Externalities

A collection of definitions from the literature review is provided below as to what comprises an externality:

- “..[an external effect is] a direct effect on another’s profit or welfare arising as an incidental by-product of some other person’s or firm’s legitimate activity...” (Mishan, 1988);

- “..[external effects] are experienced by other groups in society...” (ODA, 1988);

- “..externalities are variously known as external effects, external economies and diseconomies, spillovers and neighbourhood effects...” (Pearce (ed.), 1992);

- "..An externality can arise when two conditions are present: condition 1 for any two or more economic agents *j* and *k*, an externality is present whenever agent *j*'s utility or production relationship includes variables whose magnitudes are chosen by the other agent, *k*, without regard to *j*'s own preferences; condition 2 the *j*th individual or firm has no control over the variables chosen by *k* because the variables have no explicit exchange value, i.e. no markets (or imperfect markets) exist for the variables entering *j*'s objective function...." (Hartwick & Olewiler, 1986, p383);

- "..an externality exists whenever the welfare of some agent, either a firm or household, depends on his or her activities and on activities under the control of some agent as well..." (Tietenberg, 1992);

- "..the actions of one individual or firm affect the well-being of others..." (Turner *et al*, 1994); and

- "..the actions of a firm cause costs to other firms and/or loss of welfare to households..." (Johansson, 1991).

A positive externality (external economy or benefit) is considered here to exist when the actions of one agent benefit another party; a negative externality (external diseconomy or cost) exists when one agent's actions harm the other party. In each case, the affected party has no control over the actions of the agent generating the external effect. Externalities arise because of technological interdependencies among consumers or firms that persist because of the failure of markets to price these external effects.

2.3 Cost-Benefit Analysis

A collection of definitions for cost-benefit analysis from the literature review is provided below:

- "..considers all gains (benefits) and losses (costs) regardless of to whom they accrue (although usually confined to inhabitants of one nation..." (Pearce (ed.), 1992);

- "..an estimation and evaluation of net benefits associated with alternatives for achieving defined public goals..." (Sassone & Schaffer, 1978);

- "..goes beyond the idea of an individual's balancing of costs and benefits to society's balancing of costs and benefits..." (Turner *et al*, 1994);

- "..the fundamental role of CBA is to establish principles by which the costs and benefits of any public programme are measured..." (Hartwick & Olewiler, 1986, p 426);

- "..CBA is designed to show whether the total benefits of a policy or project exceed the costs, including environmental benefits and costs…as far as possible, all effects are measured as the persons affected would measure them..." (Abelson, 1997, p 15);

- "..decisions are made by decision makers, and benefit-cost analysis is properly regarded as an aid to decision making, and not the decision itself..." (Zerbe & Dively, 1994, p 2);

- "..general premise that benefits and costs of actions should be weighted prior to deciding on a policy choice..." (Tietenberg, 1992); and

- "..process of identifying, quantifying, weighing up, and reporting costs and benefits of the measures which are proposed to implement a policy..." (DoE, 1991).

This Report examines appraisal from a policy level, i.e. cost-benefit analysis in this case weighs up the costs and benefits of a proposed policy. Traditionally, the boundaries of such an approach are set at a national level, i.e. the CBA considers the (net) impact of the policy on 'national' welfare.

Although not necessarily reflected in the above definitions, the term cost-benefit analysis (or assessment) is used differently by policy-makers. In some cases, it is used to refer to any form of assessment, whether qualitative or quantitative in nature, which examines the costs and benefits of a policy. In other cases, it specifically refers to a type of economic analysis which is based on the measurement of as many impacts as possible in monetary terms. It is the latter definition of cost-benefit analysis which is adopted here.

Consequently, cost-benefit analysis as used here refers to a set of procedures for measuring and comparing costs, and in this sense, is a method for organising and analysing data as an aid to decision making; it does not represent the decision itself.

2.4 Cost-Effectiveness Analysis

A collection of definitions from the literature review is provided below:

- "..maximising physical benefits subject to a cost constraint...or...minimising costs for a given level of physical benefits..." (Sassone & Schaffer, 1978);

- "..identifying the cheapest manner of achieving an objective..." (Tietenberg, 1992);

- "..a method that finds the option that meets a predefined objective at minimum costs..." (DoE, 1991);

- "..aids choice between options but cannot answer the question whether or not any of the options are worth doing..." (Pearce (ed.), 1992); and

- "..shows how we can maximise the number of lives saved for a given budget..." (Turner *et al*, 1994).

As stated in the previous section, this Report focuses on the policy level of decision making, i.e. the costs of meeting the pre-set criteria of a policy. All of the above definitions are appropriate in this regard.

2.5 Direct Effects

A collection of definitions from the literature review as to what composes direct effects is provided below:

- "..costs [or benefits] that vary directly with the rate of output..." (Bannock *et al*, 1987);

- "..effects that have directly measurable productivity changes and that can be valued using market prices..." (Dixon *et al*, 1986);

- "..primary [direct] effects comprise local value added as a direct result of the project….this in turn comprises local factor incomes resulting from incremental spending associated with the project..." (Schofield, 1987, p 179);

- "..a direct benefit of a project is simply defined as an increased real value of output associated with the project..." (Sassone & Schaffer, 1978);

- "...the primary [direct] effect of cleaning a lake will be an increase in recreational uses of the lake..." (Tietenberg, 1992); and

- "...internal effects are those experienced by parties directly involved in the project...direct effects are automatically included in the arithmetic of economic analysis..." (ODA, 1988).

For the purposes of this report, direct effects shall refer to those effects and impacts that can be primarily attributed (i.e. of the first order) to a proposed policy/project. Direct effects are associated with first round changes in demand. It is such effects that are generally the focus of cost-benefit analysis, hence they are the more 'traditional' appraisal impacts.

Consider the restriction of a hazardous substance used by industry. The restriction will require users of this substance either to greatly tighten safety procedures or move to a less hazardous substance. Direct effects in this case will include (but will not be limited to):

- the incremental institutional reform or administrative costs of implementing tightened safety procedures in the 'user' industries, if this course of action is chosen;
- the incremental costs (process modification, product reformulation) that will be incurred by the move to the new, less hazardous substance, if this course of action is chosen;
- the change in income for a company producing the restricted substance, if no other buyers exist;
- the economic value of reductions in work related illnesses; and
- the economic value of reductions in local pollution due to lower discharges to the environment.

2.6 Indirect and Induced Effects

Definitions of indirect and induced effects found in the literature review are as follows:

- "...reflect the impact of the project on the rest of the economy..." (Eckstein, 1958);
- "...in a market economy a project that impinges only on one or a small number of markets can have indirect effects which are much more widespread…such indirect effects arise out of relationships of complementarity or substitutability between the demand or supply of one good and the demand or supply of another.." (Sugden & Williams, 1990, p 134 and p 137);
- "...the increased incomes of various producers...that stem from...projects..." (McKean, 1958);
- "...the indirect utility function is a function of prices and income, not of the amounts of commodities consumed..." (Johansson, 1991); and
- "...accrue to people one or more steps removed from the users of the project output..." (Schmid, 1989).

With regard to the environment more specifically:

- "...indirect use values correspond to the ecologist's concept of 'ecological functions'..." (Pearce, 1993).

In the literature, it appears that there is confusion regarding a precise definition of what indirect and induced effects are. In particular, some consider 'secondary' and 'indirect' effects as a form of 'induced' effect. To avoid such confusion in this study, the term

'induced' effect will be omitted from the discussion; we will restrict ourselves to indirect and secondary effects (discussed below). Indirect effects for the purposes of this report shall refer to changes in output (or employment) in related sectors of the economy through backward and forward production linkages with the policy, that is effects involving second- or third-round responses associated with inter-industry demand (e.g. the increased demand for factor inputs and related services induced by the first-round environmental expenditures). A diagrammatic representation of this is provided in Figure 2.1 at the end of this section.
Consider the hazardous substance example, in this case such indirect effects would include (again, this list is not meant to be exhaustive):

- the value of increased demand for factor inputs (throughout the economy) to the businesses that supply the substitutes;

- the net value of additional employment resulting from increased demand for the factor inputs (throughout the economy) required by the businesses that supply the substitutes; and

- the reverse of the above in relation to factor inputs used by the company producing the restricted substance.

2.7 Secondary Effects

A collection of definitions for secondary effects from the literature review is provided below:

- "...cost benefit calculations that take no account of these secondary income and employment effects [in the form of multipliers] will underestimate the net benefits of the projects involved..." (Mishan, 1988);

- "...the true economic efficiency benefits of this type [secondary] comprise the differential in income generated by the project as compared with some other use of the resources embodied in the project….secondary benefits may flow from income multiplier effects..." (Hufschmidt *et al*, 1990, p 36);

- "...secondary effects refer to increases in local income which are induced by multiplier processes following the creation of value added by the project in question..." (Schofield, 1987, p 180);

- "...benefits from 'multiplier' effects are sometimes claimed when short-run increases in income are generated and surplus capacity in an economy is activated by additional rounds of spending resulting from the investment..." (ODA, 1988);

- "...secondary benefits are consequences of the primary benefits of a project..." (Abelson, 1997, p 21);

- "...economic multipliers have been used occasionally to estimate secondary benefits..." (Sassone & Schaffer, 1978);

- "...immobile resources create problems...this shift [of labour] cancels out in national income accounting, but makes a big difference to the owners of the fixed assets involved..." (Schmid, 1989); and

- "...secondary employment benefits should be counted in high unemployment areas or when particular skills demanded are underemployed at the time the project is commenced..." (Tietenberg, 1992).

Secondary (effects) benefits as used in this study arise when policy expenditures and surpluses generate demands for commodities that, in turn, lead to a secondary increase in output. A diagrammatic representation of this is provided in Figure 2.1 at the end of this section.

Using our example of the hazardous substance restriction, secondary effects may include:

- the value of any additional expenditure by employees affected either directly, or indirectly by the policy (if they now receive higher wages than they would have in the absence of the hazardous substance restriction); and

- the value of any additional expenditures by businesses that provide inputs, directly or indirectly, to the policy (if they make higher profits than they otherwise would).

2.8 Opportunity Costs and Benefits

The term 'opportunity cost' can refer to effects at different levels, with definitions from the literature review being as follows:

- "...it is...useful to speak of 'private' opportunity cost in cases where one is looking at the foregone private benefits of an action; and social opportunity cost where one looks at the much wider range of foregone benefits..." (Pearce (ed.), 1992);

- "...in perfectly competitive markets, the market price of an input will equal its supply price…the supply price will reflect the opportunity cost, or the value of the resources used to produce the input in their next best alternative use ..." (Perkins, 1994, p 25);

- "...the value to society of the good or service in its best alternative use (other than the project under examination)..." (ODA, 1988);

- "...the opportunity cost of using units of an input in one project is the sacrifice of the benefits of whatever use they would otherwise have been put..." (Sugden and Williams, 1990, p 75);

- "...each input has an opportunity cost, and should contribute in output to the project at least as much as it could produce in the next best alternative use..." (OECD, 1995b);

- "...the opportunity cost of unpriced or unmarketed uses of resource...can be estimated by using the foregone income from other uses of the same resource as a proxy..." (Dixon *et al*, 1986);
- "...the opportunity cost of capital...is obtained by looking at the rate of return on the best investment of similar risk that is displaced as a result of a particular project being undertaken..." (Turner *et al*, 1994); and

- "...for environmental services their opportunity cost is the net benefit forgone because the resources providing the service can no longer be used in their next beneficial use..." (Tietenberg, 1992).

As evident from the above set of definitions, there are two distinct types of opportunity costs in economic analysis: the opportunity cost of a policy's inputs (i.e. the true economic cost of the individual resources utilised); and the opportunity cost of pursing one course of action over another. One of the objectives of this study is to investigate the opportunity cost of allocating limited resources to one policy area in contrast to another(s). Therefore, for the purposes of this report, we are concerned with the second use of the term opportunity costs. A diagrammatic representation of this is provided in Figure 2.1 at the end of this section; in this case, the opportunity cost of implementing the chosen policy is given by the foregone net benefits of not implementing the next best alternative policy.

A related concept is that concerning the cost-effectiveness of policies. This relates to the comparative cost of implementing alternative policies that would result in equivalent levels of protection to the environment or human health (in the absence of estimates of the value of the level of protection offered).

Consider our hazardous substance example, the opportunity cost of such a restriction would relate to whether or not equivalent reductions in environmental degradation and human health impacts could be achieved via another means, such as via an education process on the safe manufacture, use and disposal of the restricted substance.

This concept can be extended (and will be during the project) to examine policies that can achieve similar results for a lower or higher cost. For example, if the cost of saving a life via the restriction of the hazardous substance is ECU 2 million but a life can be saved for ECU 200,000 by investing in other policy areas such as health care, then it could be argued that the health care policy provides better value for money.

3. A FRAMEWORK FOR COST-BENEFIT ANALYSIS

3.1 Introduction

3.1.1 Overview

Economics can be divided into two areas of positive economics and normative economics. Positive economics attempts to explain and predict actual economic activity, the aim is to represent the 'facts' and not to judge whether they are 'good' or 'bad'. In contrast, normative economics explicitly introduces value judgements, i.e. its aim is to assess the desirability of various economic states. Economic appraisal techniques aim to tell policy makers which options are 'better' and such a process falls firmly in normative economics.

Cost-benefit analysis and cost-effectiveness analysis provide the key normative approaches for determining the economic impacts of the introduction of new regulations. This section provides an overview of the key concepts underlying the application of these techniques to environmental policy appraisal.

3.1.2 Cost-Benefit Analysis and Cost-Effectiveness Analysis

Although both of these approaches are very familiar, it is useful to define what is meant by these two forms of analysis:

- Cost-effectiveness analysis (CEA) is aimed at determining the costs associated with different ways of meeting given criteria; in such cases, no monetary assessment of benefits is required. When criteria are not already specified, CEA can be used to determine the implicit value which would have to be placed on a particular outcome in order for an action to be justified (e.g. the implied value of a statistical life). As part of environmental policy making, the criteria generally relate to targets set so as to minimise environmental risk or human health risks; and

- Cost-benefit analysis (CBA) is aimed at determining whether or not a project or policy is worthwhile from a social welfare perspective. An action is considered worthwhile or justified if the social benefits outweigh the costs. CBA can be used to examine the justification of a single action in terms of the relative costs and benefits, or can be used to compare the relative advantages and disadvantages of a series of options.

Under the 'conventional' application of both approaches, the micro-economic implications of adopting a new regulation are considered in detail. As a result, CBA and CEA are often classified as 'bottom-up' approaches to the assessment of policies. In this context, they tend to focus on estimating the direct effects of a change in regulatory policy, although indirect effects in a few related markets may also be taken into account in such analyses. Aggregation to a national level however, frequently requires the specification of sometimes questionable assumptions. In some cases, the effects of a policy on the structure and functioning of the economy may be so large as to render invalid the assumptions of *ceteris paribus*, thereby making it not possible to utilise 'conventional' CBA or bottom-up approaches. In these cases general equilibrium or ' top-down' types of approaches need to be adopted. That is, a more sectoral or macro-economic approach to modelling is required, whereby the interactions between different economic agents in the economy are taken into account. However, technical detail is sacrificed for greater spatial scope. These methods are discussed further in Section 5.

The relationship between these two approaches is discussed in more detail in Section 3.2 below. First, however, it is useful to consider in more detail the key principles underlying CEA and CBA.

3.1.3 Key Principles

Both CEA and CBA are based on the principles of neo-classical welfare economics, which 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 is met requires that the social opportunity costs of resource use 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.

Some of the key principles underlying these two techniques, and CBA in particular are as follows.

Individual and Societal Welfare

Welfare economics is based on a number of key assumptions including:

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

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

- societal welfare is the sum of individual welfare.

Pareto and Economic Efficiency

Welfare economics requires that for an option (or policy decision) to be justified, it should result in a 'Potential Pareto Improvement', where those who would gain from an action would be able to compensate those who would lose and still be better off. This principle underlies CBA and the decision criteria used within it which state that an action is justified if the total social benefits outweigh the total social costs.

As a result, economic efficiency is measured without regard to whom the benefits and costs accrue and irrespective of whether society considers the prevailing distribution of income to be desirable.

Incremental Costs

The key concern in such analyses with regard to policy appraisal is not the value of environmental quality *ab initio*, but the value of the incremental changes in quality which will occur as a result of the policy. As highlighted above, the concept of opportunity costs underlies the manner in which changes in welfare are measured within welfare economics, with market prices indicating the opportunity cost associated with the use of an input as reflected by individuals willingness to pay for it.

3.2 Boundary Issues in CBA

3.2.1 Introduction

Before reviewing the techniques used in the valuation of different types of costs and benefits, it is useful to first examine how the boundaries for a 'conventional' CBA are set in terms of the inclusion/exclusion of specific impacts.

To begin with, it is useful to consider a key question which is likely to determine the scope and hence nature of the CBA. That is:

> *How many markets must be examined in order to ascertain the total net impact of a policy on social welfare?*

The discussion provided below sets out the factors which should be considered in answering this question. As will be seen from this discussion, in those cases where 'conventional' CBA is judged to be inappropriate, the top-down approaches discussed in Section 5 can often be utilised to provide the necessary information.

3.2.2 Direct and Indirect Costs on Target Markets

CBAs will usually focus on the direct costs associated with the implementation of a policy measure 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 polluting behaviour is the primary subject of the proposed policy, for example, the 'operator' of a fossil fuel power station who is required to install FGD, the 'owner' of a vehicle that is now subjected to a more stringent inspection and maintenance programme, or the 'industrialist' who is required to switch from an oil-fired boiler to a natural gas fired-boiler, etc.

3.2.3 Direct and Indirect Costs in Related Markets

Following on from consideration of the effects on the target market, is the question of whether or not impacts on related markets should be considered. 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[1]:

- markets which 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 the 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 **<u>not</u>** change (the supply curve is perfectly elastic)[2]; and
  - social and private costs of the activities involved are **<u>equal</u>**.

- 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

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1 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 & Williams (1978, p.134) or Arnold (1995, p.84).

2 If, for example, a fall in the price of a good in the directly affected market shifts the demand curve for a substitute good to the left, and thereby reduces consumer surplus attached to the substitute good, this reduction is not to be counted. Provided that the supply price of the substitute good remains constant, the reduction in consumer surplus is simply the consequence of consumers improving their welfare levels by switching from the substitute to the now relatively cheaper first good; and this welfare gain is measured in terms of the increase in consumers surplus on the first good. The possibility of a change in the economic rent in connection with the second good does not rise.

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.

The smaller the quantity change and the smaller the price change in the related market, the smaller will be the related market's contribution to the total welfare costs of the policy intervention. In such cases, the related markets' contribution to the total welfare costs of a policy intervention are not likely to be sufficiently significant to worry about. That is, partial equilibrium calculations of the costs and benefits of the policy in question will provide reasonably accurate approximations of the policy's net benefits. In contrast, when the effects of a policy are non-marginal, and the targeted industry sector is highly integrated with other sectors of the economy, it is likely that related market effects will need to be considered. Potential examples where such considerations may be important are multi-pollutant/multi-effect policies (such as those on acidification which involve significant emission reductions) which may affect key factor prices such as electricity and may disrupt the demand for particular goods, such as coal and gas.

3.2.4 Direct and Indirect Environmental and Health Effects

The direct aim of environmental regulations are to reduce the damages to either ecosystems, resources or people more generally arising from human activity. However, the degree to which such direct and any associated indirect benefits (or costs) can be valued and hence directly incorporated into a CBA will depend on the following data:

- 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 partial CBA or a cost-effectiveness analysis. Either of these approaches, however, may fail to indicate whether a policy is actually justified from an economic efficiency perspective or there are significant variations in the 'value' of the environmental and health gains stemming from a policy (with this being a particular issue where a policy concerns multi-pollutant effects and different proposed measures would result in different levels of reduction and potentially geographic scales).

Where exposure-effect and population at risk data exist, however, valuation should be possible, either through the use of specific willingness to pay values for the policy of concern or through the use of benefit transfer techniques. The major constraints here are likely to be ones related to time and resources as valuation exercises are themselves costly and take several months in elapsed time to complete.

3.2.5 Social Value of Employment

In neo-classical welfare economics, no social cost is normally associated with unemployment for theoretical reasons. The presumption is that the economy is effectively fully employed, and that any measured unemployment is the result of the need to match changing demand for labour to a changing supply. In a well functioning and stable market, individuals can anticipate periods when they will be out of work, as they leave one job and move to another. Consequently, the terms of labour employment contracts, as well as the terms of unemployment benefits, will reflect the presence of such periods, and there will be no cost to society from the existence of a pool of such unemployed workers. For these reasons, a CBA will usually not consider the social employment costs (or benefits) arising from a given policy.

However, the conditions stated above are far from the reality in most countries indicating that consideration of employment related costs and benefits associated with some policies may be important. This is particularly true given the explicit obligation which has been placed on the EU to consider employment aspects in the development of policies under the Treaty of Amsterdam (COM(97)592/4).

It may be important, therefore, for CBAs to be expanded for some policies to examine the the potential siginficance of such effects.

3.2.6 Secondary Effects (Benefits)

As defined in Section 2, secondary effects refer to increases in income which result from multiplier effects. These effects comprise increased income in related sectors of the economy arising from backward and forward production linkages which are affected by a policy. They also comprise increased income through additional spending induced by increases in income or surplus capacity which results in additional rounds of spending, e.g. any second order effects arising from expenditure by additional workers employed as a result of a policy, e.g. to install any required capital equipment.

The failure to include multiplier effects in an appraisal can relate to significant underestimates of the real benefits of adopting a policy. However, they should only be included within an appraisal when the economy is not at full employment or there are no other constraints affecting the supply of labour. If all resources are currently in use, then using them in an alternative way (as part of a multiplier effect) will entail opportunity costs. For example, if a person is already employed, then taking up employment elsewhere will simply represent a shift in the economy and not an increase in employment; in such cases, there is no net benefit for the economy as a whole (with the same arguments following for shifting the use of capital assets from one mode of production to another).

Thus, if a region has full employment, then for that region, the multiplier effects resulting from a policy would be zero. Some secondary gains may occur if new jobs are higher paying but these will not take the full value of the multiplier. Where capital projects are undertaken in depressed urban areas or regions (and where ‘leakages’ are minimal), however, secondary effects may assume considerable importance.

Three broad approaches may be used to estimate secondary (multiplier) effects:

- economic base multipliers;
- Keynesian multipliers; or
- multipliers derived from input-output tables.

Of these, the economic base multiplier is the crudest, although it does implicitly capture both inter-sectoral linkage and induced spending effects. The Keynesian multiplier fails to capture the full effect of inter-sectoral production linkages but can be specified to include comprehensive induced spending effects. The input-output approach offers the opportunity to combine both production linkage and induced spending effects, while also providing industry-specific multipliers, for which the other approaches are less suitable. These approaches are discussed further in Section 5.

Note that if the analyst is also interested in the (secondary) employment-creating potential of a policy, employment multipliers can be computed from the input-output tables using estimated employment-output relationships (industry occupation matrices) for each sector. Alternatively, employment rather than monetary values could be used as the unit of measurement in building the basic input-output table.

In both cases, the information can be brought into a CBA to expand the scope of the analysis.

3.2.7 Distributional Issues

Changes in regulatory policy may give rise to distributional equity issues between people of different incomes, ages, health states and skills. Although most guidelines concerning the application of CBA identify these issues, little guidance is given concerning the inclusion of their assessment within the analysis. The main reasons for the exclusion of such issues from CBAs are the theoretical assumptions as to what constitutes economic efficiency, as described above.

There are examples of CBAs which have tried explicitly to account for distributional effects through the use of weighting systems. These weighting systems assign the value of an ECU to a person as a function of his/her relative income. The aim of these systems is to reflect government policies towards assisting certain segments of the society, and thus they give greater weight to impacts on certain income groups than to others. Although these are used, it is not undertaken as widely as was hoped.

3.2.8 Trade, Competitiveness and Productivity

In addition to employment impacts, at an EU-wide level, the issue of competitiveness is at the heart of the single market. 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, which contributes to, or increases sectoral output and incomes. These indicators include: income per head; 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, environmental policy may have implications for competitiveness if it imposes costs on some firms, which are not imposed on its competitors. It may not always be the case, however, that environmental policy imposes costs on firms; in fact, in some cases, the policy may generate benefits for the firm 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 the policy imposes costs on the firm, 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 and Tietenberg (1985)[3] identify five reasons why environmental policy 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 plant may prolong the life of older, less productive, plant;
- pollution control equipment requires labour to operate and maintain with no contribution to saleable output;
- compliance with environmental 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)[4] 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 an environmental policy will affect the competitiveness of a firm also depends on the incidence of the compliance costs. In general, the incidence

3 Christainsen & Tietenberg (1985): *Distributional and Macroeconomic Aspects of Environmental Policy* in Kneese & Sweeney (eds.) Handbook of Natural Resource and Energy Economics, Amsterdam: Elsevier Science Publishers, pp. 372-3.

4 Dean (1992): *Trade and the Environment: a Survey of the Literature* in P. Low (ed.) International Trade and the Environment, World Bank Discussion Paper, Washington: World Bank, pp. 15-28.

of compliance costs (i.e. whether the burden is borne by producers, passed onto 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 measures 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, 1995, p. 156). 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 under cut the higher prices.

Secondly, it is necessary to distinguish between short-run incidence and long-run incidence, as the effects differ. "…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, 1995, p. 163). In the short-run, it is also 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, 1995, p. 157). In this case, the compliance cost burden is borne by the producers; 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 achieve environmental benefits may yield net financial as well as environmental gains, and so can be justified in terms of financial return irrespective of environmental considerations. Smart (1992)[5] gives five reasons why firms can benefit by moving 'beyond compliance' with environmental regulations:

- preventing pollution 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;

[5] Smart (1992): **Beyond Compliance: a New Industry View of the Environment**, World Resources Institute, Washington, p. 3.

- 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 in to 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 'knock-on' effects.

If environmental policy makes some sectors uncompetitive, however, the economy will tend to restructure over time to replace the uncompetitive sectors although this will be at a cost. For example, new firms may not be as productive as the displaced ones. If economically important sectors lose their competitiveness, this could lead to substantial transactions costs for the economy and, in some instances, a higher equilibrium rate of unemployment. In economic terms, any restructuring could be painful. It may also be the case that affected firms may move to countries, which have less stringent, or no environmental polices.

Reviewing the reported effects of environmental policy on economic growth and employment, an OECD (1985)[6] study concludes however,

> "…that the effect on growth is indeterminate, being positive in some studies and negative in others. The main conclusion which emerges is that the macroeconomic effect of environmental policies is relatively small. Most of the figures reported…are in the range of a few tenths of a percentage point per year…".

This conclusion has not changed with time, with there being little evidence of environmental policy having had a negative effect on the competitiveness of even the most affected sectors (Pearce, 1992, p.27; Jaffe *et al*'s, 1995, p.157; OECD, 1996, p.45).

In summary, it should be clear that effects on competitiveness are only likely to arise if environmental policy in different countries imposes different levels of 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 environmental policy,

6 OECD (1985): **The Macroeconomic Impact of Environmental Expenditure**, Paris: OECD, p. 88.

and the regulations it gives rise to, impacts on competitiveness, therefore, depends on the extent to which its implementation is harmonised across all firms operating in the same 'global' market. In theory, EU environmental policy should not result in significant effects on competitiveness within the EU, as the potential for the harmonisation of regulations is relatively high.

Where the EU implements an environmental policy which is more stringent than policies adopted elsewhere globally, impacts on the competitiveness of EU industry may arise. For example, banning the use of a substance in a particular process or in an end-product may increase the production costs faced by EU companies as a result of the need to adopt more costly (or less efficient) alternatives. In cases where the cost increase is significant and must (to a large part) be passed on to buyers, the increase in costs may be significant enough that buyers will instead seek other global suppliers who are able to provide the goods at the previous, lower cost.

Impacts on global competitiveness have been a concern with regard to restrictions on a number of hazardous chemicals. For example, appraisals undertaken in the UK concerning bans on the use of tributyltin (TBT) anti-fouling paints on sea-going ships highlighted the potential impact which such bans would have on dry-dock activities within the UK. At the time, the alternatives were less efficient (requiring more frequent dry-docking) and more costly and there was concern that banning use in the UK would lead ship owners to shift to dockyards in Eastern Europe and Asia where use of TBT paints was still allowed.

It must be recognised, however, that where a CBA indicates the benefits of adopting an environmental policy outweigh the costs (to industry, consumers, the wider economy, etc.), then from society's perspective (in economic efficiency terms) there is a net welfare gain arising from the increased costs placed on individual firms (whether this results in just higher variable and capital costs or the closure of the productive capacity which is generating the environmental impacts). Although such costs may have impacts beyond a firm and affect local communities, the policy is justified in economic efficiency terms as the benefits exceed the costs. Obviously though, political and other considerations may over-rule such a strict economic criterion and judge the impacts to industry, local communities and specific regions as too great given equity, self-sufficiency, competition and other considerations.

3.3 Key Analytical Issues in CBA

3.3.1 Accuracy of Appraisals

Even if the calculation of the costs and benefits within the appraisal suggests a particular course of action, it may not necessarily be the case that the original appraisal provides the necessary accuracy for determining policy implementation (such as industry overstating the costs of compliance). Areas of concern include (Morgenstern & Landy, 1997) limitations in the analysis itself and difficulties in and accuracy of cost and benefit estimation.

Limitations in the Analysis Itself

In his review of all 61 appraisals carried out by the US EPA between 1990 and mid-1995, Hahn (1996) observed “wide variation from very poor to very good” in the technical quality of analyses. However Morgenstern & Landy (1997), commenting on a sample of analyses which admittedly was not selected to be representative, concluded that “*per se* [the] economic analyses are fundamentally sound” with no gross errors such as double counting, confusion of costs and benefits, equating of transfer payments with economic costs or benefits or failure to discount. They concluded that “...the studies are generally credible”.

The US EPA has faced a number of criticisms, however, with many of these relating to the appropriateness of the analysis itself. Criticisms include:

- lack of consideration of alternative regulatory options;
- lack of consideration of all categories of costs and benefits;
- failure to adequately discuss the uncertainties of their analyses;
- presenting a single estimate of costs and benefits, rather than a range of values that would reflect potential uncertainties;
- not considering distributional effects;
- not adequately considering the impact of the regulation on employment; and
- not including overall economic trends in the analyses.

The Accuracy of Cost and Benefit Estimation

Morgenstern & Landy (1997) identified three cases where there was sufficient information to judge the accuracy of *ex-ante* estimates. For control of CFCs, whilst early analysis considerably over-stated marginal control costs, estimates developed for the final RIA proved to be quite accurate and may even have underestimated costs somewhat. For the phase-down of lead in gasoline, the decline in sales of leaded gasoline proceeded much more quickly than anticipated. Banking of lead rights by refiners was more popular than expected. Overall, the costs of the regulatory change were lower than predicted, based on the fact that refiners reduced their production of leaded gasoline so much more rapidly than anticipated. For reformulated gasoline, the US EPAs predicted cost differences were in line with those observed in the market.

Another study assessing the economic consequences of amendments to the US Clean Air Act highlighted significant deviations in the estimated costs of reducing sulphur dioxide emissions to the air. At the time of implementation, the total costs of these emissions were estimated to be between $4 and $5 billion by 2010. A recalculation at the time of the amendments in 1990 estimated the total costs to be between $2 and $4.9 billion.

A particular source of inaccuracy in cost estimates arises from the difficulties analysts have in predicting how industry will respond to a new regulation and thus the costs that it will face. This has led to significant errors in evaluating both costs and benefits. These problems arise from two main sources:

- a lack of baseline information on industry structure or use of substances to be regulated; and
- the impacts of technological change.

With regard to the first, when the UK introduced regulations to meet its obligations under the EU Directive on Packaging and Packaging Waste in 1997, it was estimated that around 9,000 businesses would be required to take action under the regulations (DETR, 1998). The Compliance Cost Assessment for the regulations was based on that figure.

In 1998 the regulations were reviewed by the Advisory Committee on Packaging, which found that in fact some 19,000 businesses would be covered by the regulations by 2000. The increase was thought to be partly due to changes in the structure of the economy and economic growth. It was also noted that figures were not certain because of the difficulty in predicting what type of industries use packaging. Although the costs per business were thought to be largely unchanged from the original assessment, the total cost was increased to take account of the revised number of companies affected.

Analysis of several case studies[7] has shown that the ability of management to develop the least cost means of compliance tends to result in lower costs than originally estimated. When companies are faced with increasing compliance costs, there is a strong incentive on managers to try and reduce those costs, while still complying with the regulation.

In terms of the costs of meeting Montreal Protocol commitments to control of ozone-depleting chemicals, a general finding is that the costs have been smaller than expected (see UNEP, 1994; Cook, 1996; and Hoerner, 1995 as referenced in Hammitt, 1997). This trend has been attributed to some extent to unanticipated technical change, perhaps spurred on by market-based regulatory instruments. However, Hammitt (1997) finds that comparison of *ex ante* and *ex post* forecasts suggests "...more tempered conclusions...". Previous estimates by the USEPA are generally accurate but may have underestimated aggregate marginal control costs.

In contrast, the USEPA (referenced in Caulkins and Sessions, 1997) found that in all six cases it examined, total costs experienced by industry had been significantly underestimated. The main reason for this was that the economic appraisals had assumed that certain compliance strategies would be adopted by companies which were not followed in practice and that only incremental costs to these strategies were estimated

3.3.2 Discounting

The element of time is crucial to CBAs and CEAs and is dealt with by a process called discounting. Throughout the life of a policy, costs and benefits will occur at different times. Discounting provides a method for dealing with such differences and attempts to reflect individuals time preferences, i.e. most individuals would rather have money in

[7] Robinson (1995): **The Impact of Environmental and Occupational Health Regulation on Productivity Growth in U.S. Manufacturing**, Yale Journal on Regulation, Vol. 12, p.387-434.

their pockets now rather than some time in the future. Discounting reflects this preference by giving costs and benefits in the future a lower ‘weight’.

The main reasons behind individuals so-called time preferences relate to capital productivity and individual impatience:

- Capital productivity: money today can be used for productive purposes (i.e. capital) to generate additional income. Therefore, the £1 taken and used to buy capital may be worth £1.20 in a years time; and

- Impatience: individuals may not want to delay the chance of having money in the current period, with underlying reasons being the risk of death, uncertainty over the future, and diminishing marginal utility of money in the future due to individuals’ expectations of being better off in the future.

In perfect markets, these two factors would lead to the same rate through the equality of supply and demand for capital. However, both of these factors imply rates that are higher than is socially desirable (i.e. it does not reflect society's social time preference). Hence, government departments appraising public policies and projects, especially those with a long time gestation, use a rate that is lower than the risk free market rate.

Given that discounting automatically places less weight on costs and benefits which occur in the future, it is viewed by many as acting against protection of the environment, particularly in cases where the benefits of protection would not occur until well into the future.

4. IMPACTS AND THEIR VALUATION

4.1 Introduction

The previous section has set out basic CBA framework required to appraise environmental and health related policies. This section discusses the valuation of the impacts generated by such policies. In general, the impacts have been categorised into:

X impacts on industry and consumers;
X administrative costs;
X employment;
X environmental effects;
X mortality impacts; and
X morbidity impacts.

These are discussed in the following sections.

4.2 Estimation of Impacts on Industry and Consumers

4.2.1 Overview

The introduction of environmental regulations may impact on producers (i.e. industry) and consumers both directly and indirectly. These impacts may include:

X changes in capital and operating costs to industry arising from changes in production processes, reductions in or treatment of emissions, adoption of substitute chemicals, monitoring and any other actions required directly as a result of the regulation; and

X any increases in the costs of end products to consumers associated with changes in the costs to industry of producing the regulated goods.

Annex 1 provides a checklist of the types of cost items which may need to be considered when estimating changes in capital and operating costs.

4.2.2 Producer and Consumer Surplus

In theoretical terms, estimating the marginal value in opportunity cost terms of any impacts on industry and consumers is based on examination of changes in producer and consumer surplus. Producer surplus can be defined as the excess of revenue received by the supplier of a good over the minimum amount he would be willing to accept to maintain the same level of supply; while consumer surplus is the excess of the price which a consumer would be willing to pay for a good over that which he actually has to pay.

Each of these measures provides an approximation of the changes in welfare associated with a change in prices. They are fundamental concepts within neo-classical economics, as they indicate the minimum and maximum price that either the producer is willing to accept or the consumer is willing to offer.

In terms of producer surplus, a private sector company will face a variety of options in determining how to produce their outputs or services and are assumed to chose the one that is most efficient. The imposition of a restriction on a particular activity or the use of a particular substance will, therefore, tend to raise the production costs of the company. 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.

The importance of such impacts will depend upon the nature of the product under consideration:

X products that act as 'intermediate goods', 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, only changes in producer surplus need to be considered; or

X where a product accounts for a large proportion of end product costs, then there are likely to be impacts on both producer surplus and consumer surplus.

Many of the products regulated through environmental legislation, such as potentially hazardous chemicals, will fall within the first category acting as 'intermediate products' which are insignificant inputs relative to the overall costs of a product. For example, they may constitute only a small percentage, say 1 to 5%, of end product costs. As a result, increases in the costs of such goods will only result in small increases in the costs of the end good. For example, if the regulation of a product led to a 10% increase in its cost, and the product constituted 5% of end product costs, then the resultant price increase would be around only 0.5%. In most cases, such increases in price, would in result in insignificant losses in consumer surplus (assuming price is relatively elastic to demand and that there are substitute goods for the end-product). In such cases, therefore, the main impact which the CBA needs to consider is the increase in costs to industry and any resultant losses in producer surplus.

Where the product to be regulated accounts for a large proportion of end product costs, it will be more important to consider changes in consumer surplus. This requires information on the relationship between changes in price and changes in demand (i.e. the price elasticity of demand). Unfortunately, for many of the end-products likely to be of concern, this type of information may not be readily available. However, where a regulation impacts upon widely consumed and more essential goods, such as electricity or other forms of energy, then data on the associated elasticities of demand are likely to be available, allowing calculation of changes in consumer surplus.

4.2.3 Substitutes

The above analysis is based on the assumption that the regulation of the product does not lead to any deterioration in the quality of the final goods produced, and that there exists sufficient flexibility in technology for the companies involved to make alternative arrangements to substitute for the controlled product (albeit more expensively) where appropriate. This may not be a valid assumption in all cases as:

- drop-in substitutes may not be available;
- substitutes may have a lower efficacy; and
- substitutes may require changes in the production processes used and, potentially, the nature of the end product.

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).

Substitutes may also pose their own human health and environmental risks, which should be taken into account in the overall assessment. The European Commission has identified a number of questions which should be asked when assessing the impacts of an environmental risk reduction measure concerning the introduction of substitutes (European Commission, 1996). These are presented in the Box 4.1 below.

Box 4.1: Assessment of Substitute Chemicals as part of Regulatory Analysis

What substances might be used in place of the substance in question? What are their market situations?

Do these substitutes 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? What are the associated costs?

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

What research and development are necessary in order to switch to the substitutes? Will such activities require significant expenditure? Will retraining of personnel on use of the substitutes be required?

Will the consumer have the same level of satisfaction with the substitute?

Will some products disappear due to unsubstitutability?

In an ideal world, a CBA would consider the range of potential substitutes to determine the risks, costs and benefits associated with each of their use. In practice, however, time, data and resource constraints will limit the degree to which a detailed examination can take place. There is an inherent danger in this as it implicitly assumes that the risks associated with new substances will be less than those arising from the existing activities/substance. This may well not be the case and substitution may merely result in a new set of risks or the continuation of the risks of concern.

4.2.4 Productive Capital and Residual Value

A key issue for industry in many regulatory assessments concerns the treatment of lost productive capital. This issue arises in cases where a regulation would result in the cessation of particular production processes, either due to the direct environmental risks which the production process itself poses or to restrictions placed on the use of the end product (e.g. a certain chemical or family of chemicals). Industry has often argued that the residual value of the lost capital assets should be included as a cost of the regulatory measure and this argument has merit (from a strictly financial accounting perspective).

The failure to consider the opportunity costs associated with the loss of these assets, by an analysis instead treating them as 'sunk' costs, may result in the true costs of a policy being underestimated.

4.3 Administrative Costs

The introduction of new environmental policies is likely to either change or give rise to a range of new administrative costs associated with the implementation of the associated regulations. The types of cost which may change as a result of regulation include:

- administrative costs associated with, for example, licensing an activity;
- inspection and monitoring costs;
- costs of scientific sampling and testing;
- enforcement costs; and
- income stemming from changes in taxed activities (care must be taken in including such losses as they represent transfer payments from industry to the regulator).

Such costs may relate to both the need for investment in new capital equipment (e.g. monitoring equipment) and to revenue costs (such as man-power requirements). Impacts may be experienced by both industry and regulators, although measures which are voluntarily adopted by industry may present little change in costs to regulators and impact only on some of the costs faced by industry.

Although these costs are generally included in CBAs and CEAs where they stem directly from the regulation, they may not be considered where they arise in a more indirect manner. For example, if a policy required the introduction of limits on effluent discharges from sewage treatment plants, the direct costs associated with increased sampling and monitoring to both the treatment plant operator and enforcement authorities are likely to be considered in the analysis. However, any sampling or other administrative costs which fall on dischargers to the treatment plant as a result of the new limit values may fail to be considered.

4.4 Employment

Where examination of employment effects is likely to be important owing to the significance of a policy for either a particular sector or a number of sectors, the derivation of a physical measure for determining the impacts of changes in employment is required before the social value of that employment can be estimated. In general, the number of people likely to find short-, medium-, or long-term employment, by sector and occupational type, directly or indirectly from environmental investments, may be assessed using one of two approaches:

- micro-economic and sectoral approaches, including partial equilibrium supply-or demand-side models; and

- macro-economic approaches, including Keynesian econometric models and computable general equilibrium models.

If a policy creates a job, this has a benefit to society to the extent that the person employed would otherwise not have been employed[8]. In other words, the benefits of employment are equal to the social costs of the unemployment avoided as a result of the policy. These benefits will depend primarily on:

- the period that a person is employed;
- what state support is offered during any period of unemployment; and
- what opportunities there are for informal activities that generate income in cash or kind.

8 The same logic can be applied to the loss of a job; the arguments are simply in reverse.

In addition, unemployment is known to create health problems, which have to be considered as part of the social cost.

Once a physical measure of the net employment effects associated with a given policy has been determined, it is possible to place some money value on them. The welfare gain of an additional job is traditionally defined as:

a) the gain of net income as a result of the new job, after allowing for any unemployment benefit, informal employment, work-related expenses (i.e. the net financial gain to the 'newly' employed person), etc., minus

b) the value of the additional time that the person has at his or her disposal as a result of being unemployed, which is lost as a result of being employed, plus

c) the value of any health related consequences of being unemployed that are no longer incurred.

To calculate the social benefits (the unemployment avoided as a result of the policy), one has to multiply the welfare cost (a) minus (b) plus (c) by the period of employment created by the policy. There may be a case for including any foregone losses in human capital (e.g. loss of knowledge, communication skills, etc. that a person may experience as a result of prolonged periods of unemployment) as part of the welfare gain of creating an additional job.

Gain of Income

The gain of income will depend on the new net of tax wage, and how much unemployment and other benefits are available. Data on average earnings by occupation are available for many countries and have to be used for this calculation. Adjustment for personal taxes should be made, and this is often complicated. For those working in large enterprises, tax deductions are relatively clear, but for the informal sector there is very little information available on what taxes are paid.

Replacement Earnings

Replacement earnings are the earnings received during the period of unemployment, in the form of unemployment benefit and other forms of support. The structure of these benefits is complex. Some countries have no benefits; others have a limited amount; and others a more complex system, with benefits falling after some period. In addition, the unemployed receive some social benefits, depending on their previous work history and their qualifications. They are also permitted some part time earnings while claiming unemployment.

Value of Any Lost Leisure

In moving from unemployment to employment, an individual faces a loss of leisure, which has some value. A source of estimates of non-working time is the transport literature, where savings in travel time are valued at approximately 30-50 percent of the gross wage. This estimate is derived from considerations of tax rates as well limited work opportunities for any time saved. However, for large scale enforced non-working, such as that associated with unemployment, it is likely to be too high.

For a set of recently conducted case studies looking at the indirect costs and benefits of greenhouse gas limitation, non-working time was valued at 15 percent of the gross wage, reflecting **some** limited alternative earning opportunities9.

Health Related Impacts

It has long been known that, on average, people in employment are healthier and have greater life expectancy than those who are unemployed. This is despite the fact that many jobs involve work-related hazards, both accidents and occupational diseases (for example, long-term exposure to carcinogens at work). The generally better health of people in employment is known in occupational epidemiology as the 'healthy worker effect' (HWE). It arises at least in part because the selection of persons for employment, and the continued employment thereafter, depends on being healthy (Fox and Collier (1976) as referenced in Markandya, 1998). Thus, it is not unusual to find mortality studies of industrial workers which show standardised mortality ratios (SMRs) of 80 or thereabouts; meaning that they have age-specific death rates which are 20 percent less than the general population.

Recently, however, some real evidence has been collected which shows that health-related selection for work only explains part of the difference between employed and unemployed people; and that unemployment *per se* is also detrimental to health. To investigate this, it is necessary to separate out the effects of unemployment as such from the effects of health related selection, and that in turn requires longitudinal (cohort) studies. A selection of such studies are reviewed in Markandya (1998)10, where the author concludes that the excess mortality from unemployment in men of employable age may be taken as 75 per cent, with a range from 45 to 110 per cent11. Therefore, if the mortality rate of men of working age is 6 deaths per 1,000 men, then the excess (or additional) mortality rate of the unemployed is 4 deaths per 1,000 men. The valuation of such changes in mortality are discussed in detail in Section 4.

4.5 Environmental Effects

4.5.1 The Valuation Techniques

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9 Markandya, A. (1998): **The Indirect Costs and Benefits of Greenhouse Gas Limitation**. A report prepared for the UNCCEE, Roskilde, DK.

10 Markandya, A. (1998) **The Indirect Costs and Benefits of Greenhouse Gas Limitation**. A report prepared for the UNCCEE, Roskilde, DK.

11 This conclusion is based on an unpublished review of the literature by Dr. F.Hurley of the Institute of Occupational Medicine, Edinburgh.

One of the key requirements of a social CBA is consideration of the full social impacts of a decision, where this includes environmental and human health effects (with valuation of the latter discussed in the next section). Because such external effects fall outside the marketplace, they are not traded in the same way as other goods and services. As a result, their economic value has to be imputed through some other means.

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 (willingness to accept - WTA) an environmental loss] as revealed in the marketplace, through individuals' actions or as directly expressed through surveys. The general aim of these 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:

X conventional market price or effect on production approaches;
X household production function approaches;
X hedonic pricing methods; and
X experimental markets.

It is beyond the scope of this report to provide detailed discussion on how these different approaches are applied in practice. It is important though that the basic principles underlying each of these techniques are understood. These principles are summarised below together with the manner in which they are applied.

Market Price/Effect on Production Approaches

These approaches rely on the use of market prices to value the costs/benefits associated with changes in environmental quality. One approach in this category is 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.

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).

Household Production Function Approaches

In these approaches, expenditure on activities or goods which are substitutes for, or complements to, an environmental good are used to value changes in the level of the environmental related good.

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 measures which mitigate impacts. So, for example, the installation of double glazing on windows is a substitute for reduced noise impacts, and expenditure on this provides an indication of individuals' willingness to pay for policies aimed at reducing noise levels.

One of the key difficulties in applying this method, however, is decomposing the reason for the expenditure (for example, is 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.

The **travel cost method** (TCM) 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, any entry fees, environmental characteristics and the availability of substitute sites. In practice, a number of issues surround the application of this approach related to, for example: the inclusion of costs associated with the actual time spent travelling; trips which may involve visits to more than one site; difficulties in accounting for varying qualities of alternative sites and thus their affect on the demand for a given site; and accounting for visitors who travel to the site by modes other than private car.

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 willingness to accept an increased risk. These methods determine an implicit price for a good by examining the 'real' markets in which the asset is effectively traded.

Hedonic property (land) prices have been used in the valuation of characteristics such as air quality, noise, fishery quality 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 which are not readily perceptible in physical terms. A number of studies, for example, have found no relationship between increases in property values and differing standards of chemical water quality.

Experimental (Hypothetical) Markets

The two key techniques which involve the use of experimental or hypothetical markets are the contingent valuation method and the contingent ranking method. 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. Difficulties with this approach include: problems in understanding the concept of risk and in particular marginal changes in risk, and individuals acting strategically when responding to questions (or indeed respondents giving random answers in that numbers are pulled out of the air).

The **contingent ranking (or stated preferences method)** 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.

4.5.2 Total Economic Value

In deriving an economic value for environmental and other non-market goods and services, it is essential to consider the total economic value (TEV) of the asset under consideration. This is the sum of what economists call 'use' and 'non-use' or 'passive use' values. Use values are those associated with the benefits gained from actual use (or 'consumption') 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 values) 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 themselves or others.

Total economic value therefore is the summation of use and non-use values. However, a particular problem with defining 'value' in this 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.

4.5.3 Application of Techniques in Practice

The applicability of the different valuation techniques varies across different types of environmental impacts, as illustrated by Table 4.1. The market price, household production function and hedonic pricing techniques are restricted in their application to valuation of costs and 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 which is revealed through actions undertaken by individuals or in the marketplace.

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. Dose-response 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.

Table 4.1: Environmental Impacts and Valuation Techniques

Impacts	**Valuation Technique**					
	Financial Payments	**Dose-Response**	**Replacement/Avertive**	**TCM**	**CVM**	**HPM**
Water Quality/Quantity				T	T	T
Recreation	T			T	T	
Landscape					T	T
Heritage			T	T	T	
Habitat/Ecosystems	T		T		T	
Wildlife			T		T	
Noise		T	T		T	T
Health		T	T		T	T
Fisheries	T	T		T	T	

Aesthetics					T	T
Forestry	T	T		T	T	T

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 hypothetical 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 which can be introduced into the survey which 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.

4.5.4 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 *ab initio*. 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 issue, 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. There are three different approaches which might be adopted in benefit transfer (OECD, 1993):

X the transference of mean unit values;
X the transference of adjusted unit values; and
X 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 which 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, 1996). The advantages of this type of approach are that calculated benefits are based on information on use and unit values which 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.

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 damages to the various 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 and Kaoru, 1990; Bateman, 1996), 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 which 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 which 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). 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*, 1990). Several different approaches have been suggested, such as adjustments according to relative income, according to purchasing power and/or environmental awareness. However, 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. As a result, 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 - e.g. 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.

4.6 The Valuation of Mortality Impacts

4.6.1 Overview

Government expenditure on improving public health employs scarce funds. The decision as to whether or not to implement a health-improving project or policy, therefore, requires that a judgement is made as to whether or not the value of the lower risk of mortality (and morbidity) justifies its cost. This, in turn, necessitates that some method is found for measuring the benefits of the project in terms of the number of lives that a project is expected to save over a given period.

4.6.2 General Approach

Within a CBA, the aim is to develop a monetary estimate for mortality which provides a measure of an individual's preference for safety in terms of the amount that he or she would be willing to pay for it. By so doing, an indication is given as to how a person values safety relative to other potential goods/services, but also with regard to his or her ability to pay (and hence linked to the ability of society more generally to pay for safety).

The derivation of WTP with regard to mortality (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). These amounts are then aggregated over all affected individuals to derive a total value for the risk reduction measure or safety improvement under consideration. The resultant figure indicates what the risk reduction measure 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.

It is important to recongise that what is being estimated is not the value attached to a particular invididual's life, but the value across all of those who might be affected by reducing the risk of premature death, where the probability of death is below one. By determining the total sum which all of those who may be at risk would be willing to pay to reduce that risk, it is possible to value the benefits associated with small changes in risk.

If we assume that a group of N individuals (where N is a large number) is each willing to pay an average of X ECU to reduce the probability of death of one member of the group. The total sum that the group is then willing to pay to avoid one statistical death is equal to XN. If the mean value expressed by the group of individuals to reduce the risk of death by 1 chance in 1 million is 1 ECU, then the value of a statistical life (VSL) is equal to 1 million ECU. The calculations therefore provide an indication of the amount of money which individual's would be willing to pay to achieve marginal reductions in risk across whole populations.

This way of conceptualising the willingness to pay for a change in the risk of death has many assumptions, primary among them being the 'linearity' between risk and payment. For example, a risk of death of 1/1000 would then be valued at 1 million ECU/1000, or 1000 ECU using the VSL approach. Within a small range of the risk of death at which the VSL is established this may not be a bad assumption, but it is probably indefensible for risk levels very different from the one used in obtaining the original estimate.

In addition to an individual's willingness to pay for a reduction in risk, the concept of an individual's willingness to accept (WTA) compensation for an increase in risk is also highly relevant in the valuation of mortality 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.
Estimates of the WTP for a reduction in risk or the WTA compensation for an increase in risk have been made by:

X looking at the increased compensation individuals need, other things being equal, to work in occupations where the risk of death at work is higher (this provides an estimate of the WTA);

X through the CVM method, where individuals are questioned directly about their WTP for measures that reduce the risk of death from certain activities (e.g. driving); or their WTA for measures that, conceivably, increase it (e.g. increased road traffic in a given area); and

X by examining actual voluntary expenditures on items that reduce the risk of death from certain activities, such as purchasing air bags for cars.

The main issues that arise in estimation of the value of a statistical life are the following:

X the validity of the methods used in estimating the value of a statistical life;
X the transfer of risk estimates from different probability ranges;
X the decision context and characteristics of the risk;
X the treatment of acute versus chronic mortality; and

X the treatment of age dependent mortality and whether an approach based on VSL or on the value of life years lost is more appropriate.

4.6.3 Validity of the Valuation Methods

All three methods of valuing a statistical life have been subject to criticism. The wage-risk method relies on the assumption that there is enough labour mobility to permit individuals to choose their occupations to reflect all of their preferences, one of which is the preference for a level of risk and the level of compensation required to accept that level. In economies suffering from long-standing structural imbalances in the labour markets, this is at best a questionable assumption. Secondly, in using these methods, it is difficult to distinguish between individual's implied WTA for mortality risks as opposed to morbidity risks. Thirdly, the WTA will depend on perceived probabilities of death. Almost all studies, however, use a measure of the long-run frequency of death as a measure of risk, raising questions over the reliability of the results. In addition, the probabilities for which the risks are measured are generally higher than those faced in most other situations of interest. This point is returned to below, but a related factor is that high risk occupations generally involve individuals who are 'risk takers' and thus whose WTA an increase in the risk of death is not typical of the population at large (e.g. steeplejacks)12. The net impact of all these factors is difficult to gauge but it is <u>likely</u> that the estimated WTA will be lower than the true WTA, and of the WTA held by the wider population.

The consumer expenditure approach is subject to the difficulties that perceived probabilities are very different from objective probabilities, and that the effects of the expenditures are to reduce the risk of death as well as of illness following an accident. As a result, it is difficult to separate out the two impacts in the results produced by such studies.

The predominant approach for deriving the VSL is the contingent valuation method. As noted in the previous section, this approach relies upon direct questioning of individual's concerning their WTP for improvements which are contingent upon the existence of a hypothetical market for such goods. It asks people directly, through the use of a survey, how much they would be willing to pay for risk reduction in specified circumstances. Within these surveys, individuals are typically provided with information on the nature and level of a risk, the type of intervention proposed, the method of payment (e.g. through an increase in taxes, a one-off payment, or an increased user fee, etc.), thus creating the hypothetical market. 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.

The CVM method is subject to a range of criticisms within this context. These include the following (Ball *et al*, 1998):

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12 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.

- 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 repsondents 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 valutaion 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).

Despite the above, CVM remains the preferred approach, with many researchers arguing that it is the best method available and that most of the problems can be dealt with through better questionnaire design.

4.6.4 The Transfer of Estimates from Different Probability Ranges

An additional issue concerns the probability range over which estimation of the VSL is carried out and over which it is applied. Typically one is dealing with much lower probabilities of death from the situations of interest (of the order of 10^{-6} and lower), whereas the studies on which the estimated value of a statistical life is based are dealing with probabilities of between 10^{-1} to10^{-5}. Furthermore, as the survey by Fisher, Chestnut and Violette has pointed out, the results from studies at the higher end of the probability range are less reliable. As mentioned earlier, theoretical models would tend to predict that the WTA for lower risks should be lower but, if anything, the empirical literature shows the opposite. Partly this is due to the fact that the groups are not homogeneous. The issue remains unresolved and there is little that can be done about this problem at this stage. In the medium term, research on the theoretical and empirical aspects of the problem is needed.

Table 4.2 below summarises from ExternE (CEC, 1995) the European studies covering these three categories, and gives the central estimated VSL (and range if available) in £. All values from the studies were converted into 1990 prices. The European studies show a range of values 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 which reflect actual expenditures.

By comparison, Table 4.3 gives a summary of results from US studies, converted into £. The range (taking the average in each category) is £1.4 - £1.9 million for these studies as compared to £1.8 - £3.1 million for the European studies. Given the higher per capita income in the US compared with that of the European countries where the studies were carried out, this is somewhat surprising. It is principally the result of higher values in the European CVM studies, particularly the earlier Jones-Lee study (1976) and the Frankel Study. Eliminating these two studies would produce a range and an average for the CVM group of £1.5 - £2.1 million and £1.9 million for the EU and the US, respectively. Taking those figures would give a mean value of life of £1.8 million, which is probably the best central estimate from the European studies. In 1997 prices this is equivalent to £2 million (allowing for changes in the exchange rates).

Table 4.2: European Empirical Estimates of the Value of a Statistical Life (VSL)

Wage Risk Studies			
Country	**Author**	**Year**	**VSL (£1990 million)**
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

Consumer Market 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
Sources: See original studies			

Table 4.3: Summary Table for VSL (£ 1990 million)		
Study Type	**European**	**USA**
Wage-risk	2.0 - 2.5	2.5 - 3.9
CVM	2.9 - 4.5	1.0 - 1.8
Market	0.5 - 2.4	0.7 - 0.8
Average	1.8 - 3.1	1.4 - 2.1
Source: Adapted from Pearce *et al*, 1992.		

As discussed earlier, all of the studies are likely to be biased, with the wage-risk studies producing values that are too low and the CVM studies values that are too high. Taking an average is averaging unknown errors and one cannot say what the final impact will be. However, one can draw some comfort from the fact that the values are, in broad terms, consistent and in a plausible range.

4.6.5 Risk Context and Characteristics

A review of the literature reveals substantial across-study variation in estimates of VSL, regardless of the valuation technique adopted. These variations are in part attributable to inconsistences and differences in survey design, methodological approach, data analysis, and survey population. Kidholm (1996) also notes that differences arise between countries as income levels vary and 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.

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. 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.

Psychometric research (see for example Slovic, 1987) has identified a series of other factors related to the characteristics of the risk or risk generating activity itself which may also affect people's preferences for risk reduction. These factors relate to context, scale, age and temporal variations in the risk activity suggesting that a person's willingness to pay is influenced on whether it is:

X a risk which is observable or unobservable;
X known or unkown to those exposed;
X a risk associated with immediate or delayed effects;
X a novel or a familiar risk;
X a risk which is known or unknown to science;
X a controllable or uncontrollable risk;
X a dreaded risk (e.g. cancer) or not dreaded risk;
X associated with fatal or non-fatal consequences;
X equitable in terms of the distribution of risks and benefits, including over time;
X a voluntary or involuntary risk;
X a result of human failure or arises from natural causes; and
X is managed by a trusted and respected party.

Although all of these factors may influence people's willingness to pay for reductions in the risk of death or injury, it has not been possible to isolate their individual importance. 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. For example, Mendeloff and 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.

Horowitz (1994) found that consumers had distinct and consistent preferences for regulation of pesticide residues when compared with automobile exhaust controls when both options costed 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). The data from this work was analysed in two ways. An analysis which pooled all data together suggested that the perceived level of risk, the perceived exposure level and socio-economic characteristics all influenced expressed WTP for safety across a range of hazards. The 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 use of VSL versus VLYL.

There is certainly strong 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}; in contrast Litai (1980) argues that the difference could be as much as 100 times. Interestingly, for risks of low probability, 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.

From consideration of the above studies, some researchers have drawn the conclusion that while contextual issues do appear to have implications for expressed WTP, their effects may be relatively modest when averaged over populations, and when compared to the variance in the WTP estimates which have been derived through past studies. Others, however, do not believe this is the case as is discussed further below.

There are also arguments against the use of different VSLs based on egalitarian grounds that all lives are equal. If this rationale is accepted, the value of preventing the death of an elderly person should be equal to that of preventing the death of a young person. Similarly, the value of preventing a death now should be equal to the value of a preventing a death in the future. It could also be argued on this basis that differences in the characteristics of the risk should not be used as a basis for establishing differing VSLs because 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.

4.6.6 The Treatment of Age Dependent Mortality, Ill Health and Latency Effects

In terms of the types of adjustments which may be need to be made to VSL values to account for differences in context and risk characteristics, the key questions over what types of adjustments may be appropriate tend to be reduced to:

(a) should an adjustment be made for the risk context?
(b) should we adjust the VSL values for the fact that many of those affected are old?
(c) should some adjustment be made for their state of health? and
(d) 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' sets out a series of adjustment factors for deriving a relative value for mortalities associated with air pollution using a road accident VSL figure. The adjustments relate to differences in the risk context, income and age of the affected population, health state, the level or risk and other costs not captured by the roads willingness to pay estimates. The adjustments underlying the need to make such adjustments are examined further below.

Age Dependence for VSL

The issue of age has arisen because some of the studies, and much of the clinical evidence, suggest that pollution disproportionately affects the elderly. For example, Schwartz and Dockery (1992) report a relative risk for under 65s as 1.049 per 100$\mu g/M^3$ of PM_{10} and for over 65s as 1.166. Other studies that look at age as a distinct variable also find this effect for air pollution.

The literature on age and VSL 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 is supported by both theoretical and empirical studies. Thus, for example, within the framework of a discounted expected utility model of the type developed by Maddison (1997), the VSL will either be a monotonically declining function of age or will follow an 'inverted-U' life cycle, depending upon the particular assumptions made concerning borrowing and lending opportunities.

In turn, several empirical studies have produced evidence of a significant inverse relationship between the VSL and age, with perhaps the most marked example being the pronounced inverted-U life-cycle for the roads VSL which emerged from the data generated by a nationally representative sample survey employing the contingent valuation (CV) approach carried out in 1982 and reported in Jones-Lee (1989).

The results from that study are summarised in Table 4.4. These are reinforced by the findings of a recent (December 1997) UK Government inter-departmental study, reported by Jones-Lee, which point toward a roads VSL that is slightly lower than the figures quoted in Table 4.4. at between 55% and 75% of its mean value for 75 year olds and between 28% and 40% for 85 year olds.

Table 4.4: Mean Estimates of VSL for Different Ages as Percentage of VSL at Age												
Age	20	25	30	35	40	45	50	55	60	65	70	75
VSL at age 40	68	79	88	95	100	103	104	102	99	94	86	77
Source: Jones-Lee *et al* (1985)												

The ‘raw’ figures quoted in this study, however, mask a number of complex effects. One is that income increases with age up to a certain point and declines thereafter. Hence, if VSL is related to income one would expect 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*.

The above findings suggest that the baseline (average across the population) VSL value should be adjusted for age if the VSL valuations used in an appraisal are to provide a robust indication of willingness to pay and hence opportunity costs.

Impact of Health Impairment

Apart from the effects of age, one might expect VSL to vary with the state of health. There are two dimensions to this. One is the effect of pure health impairment and the other is the effect of shortening of life span. If a person’s quality of life is poor this may effect 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.

Adoption of the baseline VSL representing the average across the entire population provides no adjustment for pure health impairment. Nor does it include an adjustment for reductions in life expectancy. For environmental quality issues such as air pollution this may be particularly important, as there is a lot of clinical experience to suggest that the life expectancy of those who die from such exposure is already 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. Taking the data reported in Jones-Lee (1989), Markandya (1997) regressed the VSL against age **and** life expectancy. The results indicate a log linear model provides a good fit, with VSL increasing with age (reflecting increased risk averseness with age) and decreasing with life expectancy. The log-linear model used takes the reduced form:

$$VSL = A.AGE^{\alpha}LE^{\beta}$$

where LE is life expectancy, and alpha and beta are parameters of the VSL function. This implies that the valuations of different individuals is age dependent and dependent on life expectancy. This allows one to replace the 'normal' life expectancy with the reduced life expectancy that is characteristic of environmental pollution. The above equation was estimated for data on European males, using a VSL age profile as given in Jones-Lee (1989).13 The data are given in Table 4.5.

Table 4.5: VSL, Age and Life Expectancy for EU Males

Age	Life Expectancy	VSL (£m)	VSL Predicted (£m)
20	69.9	1.66	1.6
30	40.5	2.16	1.93
40	31.2	2.42	2.77
50	22.5	2.38	2.94
60	14.7	2.08	2.31
65	11.4	1.84	1.85
70	8.4	1.52	1.32
75	5.9	1.12	0.86
80	3.9	0.68	0.49
85	2.4	0.16	0.24

Notes: 1) VSL is take from Jones-Lee (1989) and is based on a mean value of £2.0 million.
2) VSL Predicted is calculated from the above estimated equation.
3) Life expectancy is computed from EU male survival probabilities.

The estimated equation indicates that VSL increases with age and withlife expectancy. Being log-linear, the coefficients can be treated as elasticities. A one percent increase in age, therefore, raises the VSL by about 2.9 percent, while a one percent decrease in life expectancy reduces it by about 1.8 percent. Replacing the actual life expectancy for a given age with the life expectancy of a typical person who is affected by, say, particulate

13The estimates from the equation are as follows: constant (A) = 15.519; alpha = 2.882; and beta = 1.793.

pollution (0.75 years)14, yields the estimates given in Table 4.6. These values correspond to a mean value of £2.0 million for someone aged 40. It should be noted that this approach is essentially deriving estimates of life years lost that vary with age.

Assuming that for particulate pollution we are dealing with persons over 65, the corresponding value of mortality would be between £24,000 and £51,000. That is, preventing the death of a 65 year old eprson from particular pollution may be valued at about £24,000.

Table 4.6: Estimates of VSL for a Person with 0.75 Years to Live

Age	30	40	50	60	65	70	75	80	85
VSL £	2,350	5,800	11,043	18,680	23,520	29,20	35,553	42,800	50,970

Source: Markandya (1997)

These findings highlight the potential importance of adjusting VSL for reduced life expectancy in addition to age.

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 accepted way to deal with such latency is to discount future risks, so that if the WTP for an immediate reduction in risk is £X, then the WTP for a reduction in a risk with a latency of T years is x $(1+r)^{-T}$. The key question, of course, is what value should T take?

There is a case for relatively high discount rates (around 11%), as well as one for low rates, in the region of 3%. Given the lack of agreement among economists as to which is the appropriate rate, it is recommended that calculations be done with both rates and the resulting range of values reported.

4.6.7 Value of Life Years Lost

In the valuation of the mortality effects of chemicals in the environment, concern has been expressed over the use of a VOSL based on traffic accident or 'wage risk differential' studies. Such methods of assessing the willingness to pay for a reduction in the risk of death deal with individuals for whom death would imply a loss of life expectancy of 40 years or thereabouts. Across different forms of chemical risk, the expected loss of life expectancy may vary greatly from this. As noted above, in the case of air pollution, experienced clinicians suggest that the loss of life expectancy is between 9 and 15 months. Furthermore, as the above discussion indicates, it is unreasonable that

14This means that the average life expectancy of a person susceptible to death from particular pollution is 9 months, i.e. they ae likely to die in 9 months in any event. Hence, the average loss of life is 0.75 years.

the same value should apply in both cases; as Table 4.6 demonstrates, the presumption must be that a lower value applies for mortality effects where the loss of life expectancy is considerably shorter than it is in traffic accidents, for example.

In relation to the valuation of mortality effects from air pollution, the alternative approach which researchers are considering is to analyse changes in the risk of death in terms of the value which would be assigned to each of the future life years which would be lost as a result of premature death. This is the 'value of life years lost' (VLYL) approach.

There are two aspects to this approach; one is conceptual and the other is practical. The central conceptual issue is that premature death matters because life is shortened and the amount of the shortening is material. 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 the 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, we should carry out studies of WTP for people with different survival probabilities and, from those, see how much the value would vary by life expectancy. In practice this is difficult, but not impossible.

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 experession:

$$VSL_a(\{P\},r)=VLYL.\sum_{i=a}^{i=T} P_i(1+r)^{-i}$$

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-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 one to get an initial estimate for the kind of changes in survival probabilities expected to be found in the area of air pollution.

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, as 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 which have been so adjusted, such as those presented in Table 4.4 above or through research aimed at directly eliciting such values.

A more recent study that takes a different approach in calculating the VLYL is that of Johannesson and Johansson (1996). They use the contingent valuation method to look at the WTP of different respondents aged 18-69 for a device that will increase life expectancy by one year at age 75. 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?"*

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)15. 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, which may require checking. 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.

4.6.8 Current Practice and the Application of VSL and VLYL

As has been discussed in this section, the limitations of the current practice on valuing mortality 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 CVM 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 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.

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15 The normal assumption is that VLYLs decline over time at a constant discount rate and it is this assumption that is challenged.

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 very few exceptions, CVM surveys about mortality risk have focused on the risk of accidental death. These surveys concern risks for which the average loss of life expectancy is about 40 years. Because acute mortality risks, 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 air pollution.

The recently published report by the UK Department of Health (Ad-Hoc Group on Economic Appraisal of Health Effects of Air Pollution, 1998) notes, for example, that currently "…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. 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 finding also underlines one of the report's conclusion that empirical work is urgently needed to properly develop a 'VLYL-type' approach, and thereby (directly) derive values for reduced risk of latent, chronic and future mortality in the appropriate context. This approach has the clear presentational advantage of clearly addressing the risk of premature death and is preferred by clinicians as it relates better to the manner in which they prefer to produce risk data. It also brings the WTP approach closer to the QALY approach which is used throughout the health care sector for allocation of funds across different treatments.

Given the above, there are obviously advantages and disadvantages associated with the use of either a VSL or a VLYL approach. So, which measure of value should decision makers seek information on? Firstly, it must be remembered that the VLYL is generally derived from a VSL value and thus the two approaches should be consistent. At this point in time, however, more effort has been placed on the development of the type of age and health state adjustments needed for application of existing VSL values to environmental policy questions such as air pollution. Assuming that such adjustments are undertaken and are justified, then this may be the more robust approach in the immediate future.

However, given that the VLYL approach more appropriately address the premature death risks associated with many environmental issues, decision makers should push for further research in this area to provide the basic data necessary to develop more reliable values specific to the risk contexts of concern.

An example of the type of research which should assist in both regards is given in Annex 2 of this report. The description provided in the Annex will provide the reader with some insight into current elicitation practices and suggestions as to how these may be improved.

4.7 Morbidity Effects

4.7.1 General Approach

Epidemiological data has identified a relationship between certain health ‘endpoints’ and environmental pollution (mainly air). The following are the endpoints for which some valuation is, therefore sought:

I Bronchodilator use in asthmatics;
II Cough in asthmatics;
III Lower respiratory symptoms in asthmatics (wheeze);
IV Prevalence of child bronchitis;
V Prevalence of child chronic cough;
VI Restricted activity days;
VII Chronic bronchitis in adults;
VIII Hospital admissions for congestive heart failure;
IX Chronic admissions for ischaemic heart disease;
X Respiratory hospital admissions; and
XI Cerebrovascular hospital admissions.

Discussed below are the issues arising in the valuation of these endpoints.

The full cost for an illness is composed of the following parts: (a) the value of the time lost because of the illness, (b) the value of the lost utility because of the pain and suffering, and (c) the costs of any expenditures on averting and/or mitigating the effects of the illness. The last category includes both expenditures on prophylactics, as well as on the treatment of the illness once it has occurred. To value these components, researchers have estimated the costs of illness, and used CVM methods as well as models of avertive behaviour.

The costs of illness (COI) are the easiest to measure, 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 costs of lost time are typically valued at the post-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 post-tax wage. Complications arise when the worker can work but is not performing at his full capacity. In that case an estimate of the productivity loss has to be made.

It is important to note that COI is only one component of the total cost and, furthermore, that it is not necessarily a part of the WTP to avoid an illness. For example, if a person's medical costs are paid for through general taxation, the stated WTP to avoid a particular health 'endpoint' will not include such costs. Hence the relationship between COI and WTP is complex, and one cannot add the two items 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. To arrive at the total cost of an illness, however, one should take WTP, **plus** the part of COI that is not reflected in WTP. This will be the component that is paid for through taxation and, possibly, through insurance.

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.

As indicated above, the WTP for different health endpoints can be measured either through the CVM approach, or through models of avertive behaviour. The latter 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. The difficulty is in estimating the production function, as many 'inputs' may provide more than one service (e.g. bottled water, air conditioners); in addition, changes in consumption as a function of the state of illness are difficult to estimate. There are few estimates of health endpoints based on such models.

4.8 Key Issues

4.8.1 Use of Standard Values in Policy Appraisals

Many of the CBAs and CEAs undertaken to assess the impacts of environmental regulations will depend on the use of standard values to allow aggregation from single industrial company/consumer effects to national (or international) effects. Such procedures usually rely on either:

X the use of average costs which are then used to aggregate across all companies within a sector; a common example of this practice is the use of agricultural price data on the value per tonne of output; or

X the use of case studies which develop estimates for a range of 'typical' firms within an industry sector (or sectors), for example, case studies may be developed for different size companies, or companies with different production or treatment processes; aggregation then relies on multiplying case study estimates by the number of firms falling within each category and summing across categories.

Although it is difficult to see how a regulatory appraisal could be undertaken in the absence of either approach, the dangers inherent in such simplifications should be realised. This is particularly true at an international level where the characteristics of any given industry sector are likely to vary considerably over different countries. Key factors which may be at variance include the adoption of new technologies, rates of investing in new equipment, existing levels of environmental protection, employee to output ratios, etc.

Similarly, many national governments use standard values for morbidity or mortality effects within appraisals to assist in making policy and project decisions. For example, the UK government has defined VSL figures for use in road transport assessments and in setting worker health and safety regulations. Industry also uses standard values within its own decision making, for example, some oil companies assume VSLs of US $2 million and higher when determining emergency response and worker safety requirements (Fleischman, 1992).
The adoption of such standard values can help in ensuring that CBAs are more consistent across applications. Although such an approach introduces uncertainty into the analysis as the values which would be generated by problem specific valuation studies might vary considerably from the standard values, it is likely that the standard values (as long as they are based on sound assumptions) will provide reasonable approximations of economic costs or benefits for comparative purposes (GECB, 1995). There is also no reason why varying assumptions on the appropriate VSL, extended life factor or environmental transfer value cannot be examined to determine how sensitive the end results are to the transfer values.

Given that it is unlikely that either the financial resources or the time will be available to undertake problem specific valuation studies, therefore, the use of benefit transfer and standard values provides a way forward toward the direct incorporation of human health and environmental costs and benefits within the regulation of hazardous activities and products. It must be remembered that the end decision will in any event place a value on such effects. In the absence of explicit valuation, the values will be determined implicitly and may not be consistent across different regulations.

4.8.2 Actual versus Hypothetical

However, the key question regarding the use of willingness to pay estimates derived from contingent valuation studies concerns whether respondents would actually pay their bid amounts in reality. It is argued that the elimination of so-called ‘zero bids’ or ‘protest bids’ (or problems such as embedding) ensures that the most reliable estimate is obtained. However, regardless of the sophistication of the statistical technique or the detail of the questionnaire there are two key questions to be addressed:

X do respondents know what they are valuing? and
X even if they do, would they be actually be willing to pay?

In response to the first question, if one looks at the results of willingness to pay studies and attempts to aggregate the associated values for environmental assets (as attempted by Constanza *et al*, 1997) the result is a value in the trillions. It may be the case that environmental assets are indeed worth this value, however, it is unclear if respondents are valuing the specific environmental (or health related) asset or rather valuing the 'environment' (or good health generally) as a 'whole'. If respondents cannot separate out the parts of the environmental whole, then the results of such valuation studies will be (and may be) rendered meaningless.

If it is accepted that respondents are aware of what they are valuing, it still does not answer the question regarding their actual willingness to pay. The construction of a hypothetical market means that, in reality, there is no request for payment, which in turn may mean that respondents may 'bid' higher than they would normally when payments would in effect take place. The implication of this is that the calculated benefits (and to a lesser extent costs) of an environmental policy may be overstated given that they will not be realised in a 'traditional' market.

The problem of the non-realisation of many environmental and health benefits in a traditional market is a fundamental flaw within CBA and the need (at present) to value all impacts in monetary terms. Of course, if impacts are not valued they tend to be given less weight within the decision making process. In effect, a Catch-22 situation arises:

X monetised benefits may not be believed (either due to great uncertainty or the hypothetical basis for the valuations); and

X yet, as demonstrated by the case studies, non-monetised benefits are effectively excluded from further consideration in the appraisal given the nature of CBA.

4.8.3 Equity in Valuation

As noted above, the basic philosophy underlying CBA is that individual preferences can be measured through the use of willingness to pay techniques. Assuming that individuals are able to a express a willingness to pay for a gain in welfare, the sum of willingness to pay across all individuals provides the measure of social welfare. Within such calculations, no special weight is given to any particular group.

The first objection to this approach is with the use of WTP in that it is 'income constrained'. Since you cannot pay what you do not have, a less well-off person's WTP tends to be lower than that of a well-off person, all other things being equal. This occurs most forcefully in connection with the valuation of a statistical life where the WTP to avoid an increase in the risk of death is measured in terms of a VSL. In general, one would expect the VSL for a less well-off person to be less than that of a well-off person (given the differing abilities to 'pay'). But this may be no more or less objectionable than saying that a well-off person can and does spend more on health protection than a less well-off person; or that individuals of higher social 'status' and wealth live longer on average than person of lower 'status'; or that well-off neighbourhoods will spend more on environmental protection than less well-off neighbourhoods. The basic inequalities in society result in different values being put on the environment by different people.

One may object to these inequalities, and make a strong case to change them but, as long as they are there, one has to accept their implications. However, there are arguments that income-constrained VSL values are inequitable (see for example, NERA/CASPAR, 1997). The issue being that it is not equitable to treat those that are less well off as having 'a life worth less' than those that a more well-off. Given this, there is a limit to the extent to which the present methods of valuation should differentiate between people on the basis of their WTP. Policy-makers prefer to adopt a single average WTP to a group, rather than to use a higher value for people with higher incomes. This convention, to a significant extent, answers the criticisms made above.

4.8.4 Absence of Dose-Response Data

Essentially, valuation of changes in risk to human health or the environment is based on four sets of data: pollution concentrations (both background levels and mean activity related levels); dose-exposure/response relationships; unit valuation estimates; and the population or environmental stock at risk. Where such data are not available then valuation is also not possible. For many proposed environmental regulations, information on exposure or dose-response relationships for different receptor groups or environmental targets and on the population or environmental stock at risk will not be available.

To some extent then, the degree to which valuation can take place depends upon the information available from risk and environmental assessments:

X where the output of the assessment is in the form of a risk quotient, insufficient information will be available to place monetary values on any changes in risk; the assessment, therefore, will have to be qualitative or rely on the use of some other form of quantification; and

X where it is possible to translate the risk quotient to predictions of the frequency of a specified consequence(s), then monetary valuation may be possible.

In addition, estimates are required not just for the worst-case scenario, but also for other potential scenarios or outcomes. Economic theory does not argue for decisions based on the expected risks (or benefits) but argues, instead, for separating the distribution of risks to the population from estimates of risk aversion. In this case, the risk assessors use of 'worst case' estimates provides only one point in the risk distribution.

5. COMPLIMENTARY ASSESSMENT METHODS

5.1 Introduction

Section 3 reviewed the application of partial equilibrium CBA to estimate the costs and benefits associated with meeting new environmental policies. In both approaches, pollution abatement measures are represented in great technical detail and changes induced by a policy intervention are assumed to be marginal. The use of CBA techniques in this way represents a kind of 'bottom-up' approach to policy appraisal. However, in some cases, the actual affects of a policy on the structure and functioning of the whole economy may be non-marginal in nature, and should therefore be taken into account. In such cases, the use of so-called 'top-down' methods are required, in which a more sectoral or macro-economic approach to modelling is adopted, whereby the interactions between economic agents in the economy are explicitly taken into account. However, technical detail is sacrificed for greater spatial scope.

Top-down approaches recognise that the implementation of new legislation 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 new environmental legislation at the micro-level will also have an impact at the sectoral-level, the next highest economic accounting level to which individual companies belong. Sectors also act in a dual capacity, i.e. as 'buyers' and 'sellers'. Consequently, the introduction of new legislation at the micro- or sectoral-level will affect the interactions between sectors, and ultimately affect the functioning of the entire economy.

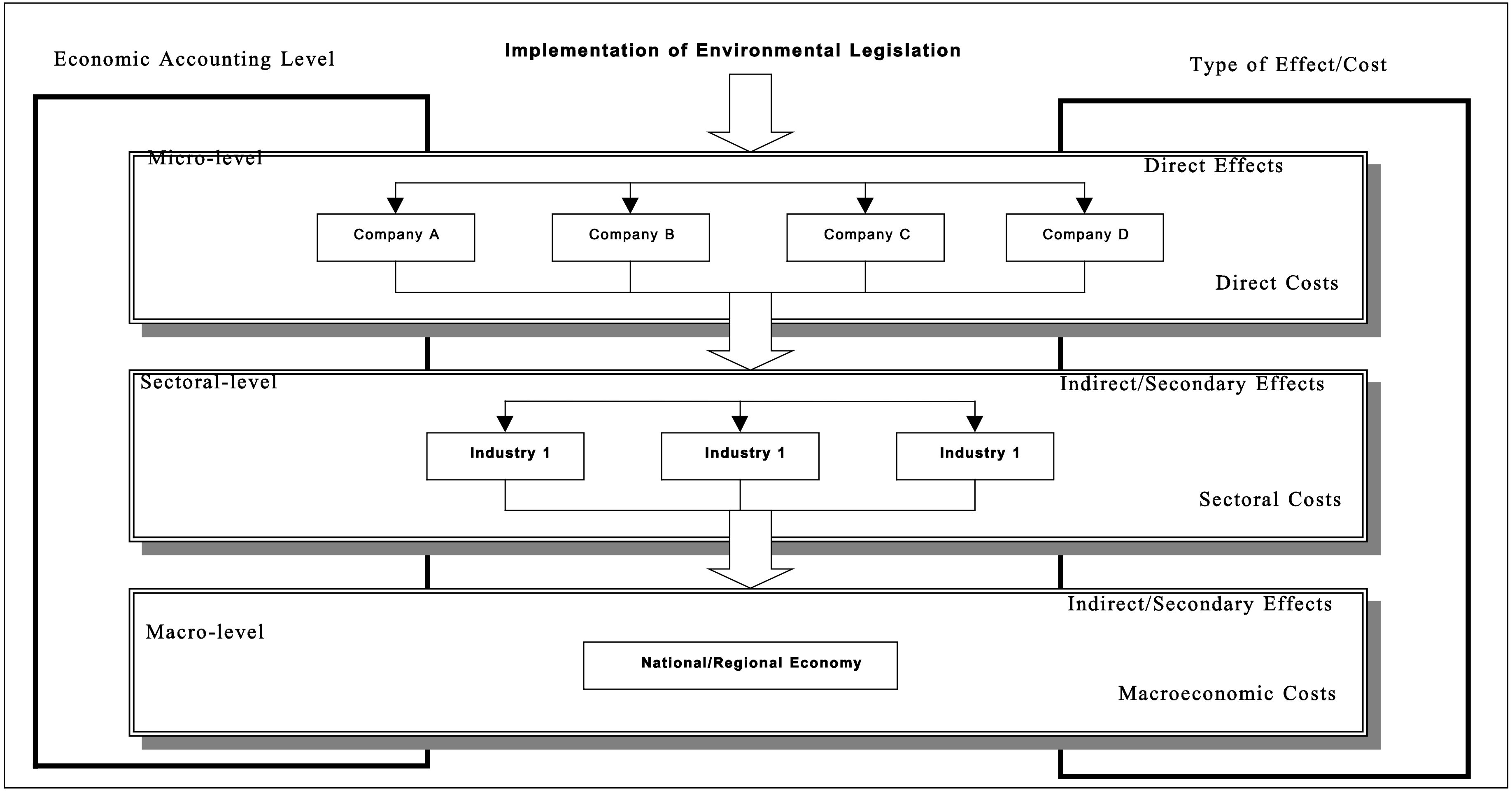
The intra-actions between companies in the same sector and companies in another sector, and the inter-actions between various sectors, encompass the direct and indirect effects of implementing new environmental legislation.

Figure 5.1 summarises the linkages between the different economic accounting levels. Quantifying the direct and indirect effects of implementing new environmental legislation at the sectoral- or macro-level requires the linkages between the various economic agents in an economy to be explicitly specified. This requires the use of more sophisticated appraisal techniques to those typically used to assess marginal (direct) effects[16]. Techniques which are capable of modelling the linkages between different economic

16 With respect to employment, standard bottom-up approaches do not capture the indirect employment effects of environmental policies, for example, due to:

a. the demand of intermediate goods and services induced by the environmental sector;
b. multiplier effects through increased wage incomes of those employed in the environment sector; and
c. accelerator effects through increased investment of the environment sector (OECD, 1997a).

agents in an economy, and therefore quantifying the ‘general equilibrium effects’ of implementing new environmental standards, include computable general equilibrium models and input-output models.



Source: Adapted from FSO (unpublished)

Both of these techniques are considered briefly in this section, with the aim of illustrating how they may be used to assess the 'wider' or 'macro-economic' effects of policy interventions. It is not the purpose of this section to provide a detailed discussion of these methods. Multipliers are also briefly considered as a method of measuring secondary benefits.

5.2 General Equilibrium Models

5.2.1 Introduction

Applied, or computable, general equilibrium (GE) models are the most sophisticated type of top-down approach used to evaluate the net benefits/costs of implementing a proposed environmental policy. They are capable of quantifying direct and indirect effects of environmental policies on the economic structure and product mix, economic growth, the allocation of resources and the distribution of income (FSO, unpublished). Moreover, as GE 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 all economic variables.

GE models still provide a measure of policy worthiness (i.e. the net benefits of a policy intervention), which may be used as an input to the decision making process.

5.2.2 Partial Versus General Equilibrium Analysis[17]

To illustrate the distinction between the two types of analysis, suppose a carbon tax is imposed on petrol. It is likely that the imposition of such a tax will have impacts beyond the petrol market[18]. Firstly, the tax services to raise the price of petrol, which in turn will induce shifts in demand curves in other markets. Secondly, the prices of other goods and services whose supply curves are upward sloping, will change, inducing second-round effects on the demand for petrol. Thirdly, primary inputs will be reallocated across the economy as the production of goods and services changes. This, in turn, will affect the incomes to different factors of production (e.g. labour, capital, etc.). Finally, since different agents in the economy may not have the same marginal propensity to save/consume, when redistributing the proceeds of the tax across different economic

[17] Based on an example given in Zerbe & Dively (1994).

[18] As noted in Section 2, these indirect effects arise out of relationships of complementarity or substitutability between the demand or supply of petrol and the demand or supply or other goods, e.g. automobiles. 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 (Sugden & Williams, 1990).

agents, the government may change the pattern of relative demands for different goods and services. These changes result in a new vector of consumption and productive prices, which directly affects the rate of productive capital formation, technological innovation, labour supply, and therefore the economy's dynamic growth path.

When these effects are non-marginal, partial equilibrium calculations of the costs and benefits of the tax will give a very poor approximation of the overall value of the tax policy[19]. It is worth noting that "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..." (Mishan, 1994, p.188).

This is, partial equilibrium analysis focuses on the gains and losses of a policy on a single market, or a few markets. It is also fair to say that most bottom-up approaches to modelling the costs of environmental policies are a type of partial equilibrium analysis.

However, if the policy under investigation instigates changes that are so large (non-marginal) as to render invalid the assumption of *ceteris paribus*, then it is not possible to proceed with the marginal approach, i.e. conventional CBA or bottom-up approaches. If other prices are expected to change as a result of the policy, then a general equilibrium type of approach is needed. General equilibrium analysis is a comprehensive method of analysis in which the *ceteris paribus* assumption is discarded and all the economy's interrelationships are taken into account.[20] Conceptually the technique is quite simple:

"... each market has a supply curve, a demand curve, and a corresponding equilibrium condition that equates the quantities supplied and demanded at a given price. But prices and quantities in each market are shift parameters in the supply and demand curves in other markets. Thus equilibrium must be system wide ... general equilibrium is a comprehensive, simultaneously determined equilibrium in all markets ..." (Cal & Holahan, 1983, p.416).

Hence, when a policy induces non-marginal changes, GE models, because they explicitly model the interactions between markets and thus account for the effects that a change in one market has on another, give a relatively more accurate estimate of the overall impact of a policy, than would be obtained through conventional (partial equilibrium) CBA.

5.2.3 The Standard GE Model

[19] Some 'rules-of-thumb' regarding the applicability of partial equilibrium CBA in assessing the direct and indirect effects of a policy in related markets are presented elsewhere in this Report.

[20] Walrus, L. (1874): **Elements d'Economique Pure**, Lausanne, Switzerland: F. Rouge.

GE models essentially simulate markets for production factors, products, etc. with systems of equations specifying supply and demand behaviour across all markets. Although there are many examples of GE models, Zerbe & Dively (1994) suggest that the best 'thought-out' models will have the following elements.

- 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 GE analysis is conducted, although most analyses involve the following basic steps (Gramlich, 1990):

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;
the proposed policy change is then modelled by shifting the supply and demand curves appropriately;
the model is re-solved, yielding a new vector of production and consumption prices; and
finally, the overall net benefit/cost of the proposed policy is determined by examining the difference pre- and post-policy vectors of prices.

As noted, general equilibrium approaches compare two distinct states of the economy; pre-policy versus post-policy. The difference between the two 'states' represents the net (economic) benefit/cost of implementing the policy in question. GE models therefore still provide a measure of policy worthiness, which may be used as an input to the decision making process.

5.2.4 Examples of GE Models

Various applied GE models have been used to assess the implementation of environmental policies. The potential of a number of these to assess direct and indirect employment effects are reviewed in OECD (1997a) and listed in Table 5.1 under the three broad headings adopted by the report's authors.

Table 5.1: Some Examples of Applied GE Models: Employment effects

Type of Model	**Model/Architect**
Optimisation of planning models:	Manne and Richels's GLOBAL 2100[1] Rutherford's Carbon Right Trade Model
Econometric Models:	Jorgenson and Wilcoxen's model[1] Hazilla and Kopp's model

	McKibbin and Sachs's G-Cubed Model
Calibrated GE models:	Whalley and Wigle's model[1] OECD's GREEN Model[1] Goulder's model The Dutch Central Planning Bureau's MIMIC Model Bergman's model Beaumais and Schubert's model

[1] **Note:** These models are also reviewed in Cline (1992)
Sources: OECD (1997a)

Several applied GE models have also been specifically designed to assess the overall economic impact of addressing the enhanced greenhouse effect. Some of the leading models are reviewed in Cline (1992); in addition to those models noted in Table 5.1, these include: Edmonds and Reilly's model; Nordhaus and Yohe's model; and Blitzer, Eckaus, Lahiri and Meerus's model. Other important surveys of top-down economic models for abatement cost assessment can be found in Boero *et al* (1991), Darmstadter and Plantinga (1991), Edmonds & Barns (1990) and Hoeller *et al* (1990).

In addition to applied GE models, there are other economic modelling approaches for assessing the direct and indirect effects of environmental policies, although these are not as sophisticated as the GE models. A number of traditional econometric models used to assess the medium-term economic effects of environmental policies are also reviewed in OECD (1997a), including: HERMES (European Commission); DRI (D.R.I.); MDM (Cambridge Econometrics); QUEST (European Commission); and ATHENA (The Dutch Central Planning Bureau). Further examples in detail are discussed below.

The DICE Model

The 'DICE' model has been developed by William Nordhaus of Yale University as part of his studies of climate change. The acronym stands for 'Dynamic Integrated Climate Economy model' which attempts to integrate the economic costs and benefits of greenhouse gas reductions with a dynamic representation of the scientific links of emissions, concentrations, and climate change.

DICE constructs a model of optimal economic growth with an additional climate sector and it estimates the 'optimal' path for both capital accumulation and greenhouse gas emissions reductions. The calculated trajectory can then be read as one of two states:

- the most efficient path for slowing climate change; or as
- the competitive equilibrium among market economics where the externalities are internalised using the appropriate social shadow prices for greenhouse gases.

The structure of the model can be broken down into a number of parts:

- the time horizon is 400 years;
- an inital stock of capital;
- an initial stock of labour;
- an initial stock of technology;
- industries behaving competitively;

- output is produced in accordance with a Cobb-Douglas production[21];
- population growth is exogenous;
- technological change is exogenous;
- capital accumulation is determined by optimising the flow of consumption over time;
- an emissions equation;
- a concentrations equation;
- a climate equation;
- a climate-damage function;
- a greenhouse gas reduction cost function;
- a conventional investment policy variable; and
- a rate of emissions reduction policy variable.

[21] Such functions take the form $Y=A.L^{\alpha}.C^{\beta}$, where Y=output, L= labour, C=capital.

The two key components of the model are the climate-damage function and the greenhouse gas reduction cost function[22]. The outputs from such a model make recommendations as to the most efficient way forward to curb climate change. The results of alternative approaches are shown in Table 5.2.

Table 5.2: Results of Alternative Policy Approaches via DICE

Run	Control Rate (%)*	Carbon Tax ($/tC)	Annualised Global Impact ($bn)
Optimal policy**	8.8	5.24	16.39
20% cut in emissions from 1990 levels	30.8	55.55	-762.50
Tax with wasteful spending	0.3	0.02	-0.56
Tax recycled by lowering burdensome taxes	31.7	59.00	205.97

* reduction of greenhouse gas emissions below baseline as percentage of baseline
** assumes that revenues are returned through lump-sum or nondistortionary rebates

It is interesting to note that the policy with the greatest 'benefits' is a carbon tax of $59 per ton of CO_2 equivalent with revenue recycling (which refers to reductions in taxes on capital, leisure or consumption). This is in stark contrast to a situation where such revenues are 'wasted' resulting in costs of $0.56bn.

William Cline, an economist at the Institute for International Economics, has highlighted four problems with the DICE model:

12. The results as given in Table 5.2 do not take into account a probabilistic distribution of lower and upper bound warming. DICE applies a 3°C rise for benchmark carbon-dioxide-equivalent doubling and a damage of 1.33% of GDP for that level. If the DICE model were applied to upper-bound warming (4.5°C for doubling), all damages are 2.25 times higher.

13. The DICE model uses a 50-year thermal lag, this means that using DICE global warming will be 3°C by 2100, however, the IPCC estimate that warming by this year will be 4.2°C. The DICE model may therefore underestimate warming.
14. The nature of the abatement costs curve may overestimate such costs. As explained above, for the initial cuts in greenhouse gases costs will be low, but for greater reductions the costs will be very high. This curve does not take technological change into account, therefore even though new technology may be available to make reductions possible at little cost, it is excluded from the model.

22 This function assumes an efficient programme could achieve the first 10% reduction at little cost, while a 50% reduction would cost about $200 billion per annum.

15. The DICE model applies a logarithmic utility function and a discount rate for 'pure' time preference. This results in a higher level of discounting as applied by Cline (1992) and as discussed in Section 3, such discounting gives lower weight to impacts far in the future.

As a general criticism of DICE, Cline believes that at present it would underestimate the damages and overestimate the costs, leading to conclusions that may not be borne out in reality.

The Danish Rational Economic Agents Model

The model (called DREAM for short) is a dynamic computational general equilibrium model of the Danish economy under development at Statistics Denmark. It is at its earliest stages and at present is simple and all markets are competitive. The model consists of four sectors:

- a corporate sector;
- a household sector;
- a government sector; and
- a foreign sector.

Firms seek to maximise the value of the outstanding stock of shares and investments are financed by an exogenous combination of debt and retained profits. The households sector consists of overlapping generations of households with a finite and deterministic time horizon. Income for consumers arises from six sources: wages, unemployment benefits, age-dependent income transfers from the public sector, lump-sum transfers from the public sector, bonds and shares, and inheritance left by the parent household. The labour market is assumed to be competitive and unemployment is therefore considered to be voluntary.

The government collect taxes, distribute income transfers and purchases the domestically produced good. Income transfers are assumed to be age and gender specific, the data for which were obtained from a project undertaken for the Danish Ministry of Finance and EPRU on generational accounting.

The model has been used to estimate the impact of shifting the tax burden from labour income onto consumption of goods (which is similar to how environmental taxes would be levied). As a result, aggregate consumption is initially depressed as a consequence of the reduced consumption of pensioners and the increased propensity to save of younger generations. However, consumption recovers after 15 years and increases due to the increased stock of wealth in the economy. Utility declines for most generations living in period 1 but increases for the youngest generations and for all the future generations. From the standpoint of utility alone, given that utility of the youngest and future generations increases, it can be argued that on welfare grounds such a tax reform is acceptable. However, given that the costs of implementing such a policy are not known, it would not be correct to say that the policies benefits outweigh the costs. Indeed, given the discounting procedure included within the cost-benefit appraisal framework, such benefits to future generations may be too small to offset costs.

The Wageningen Applied General Equilibrium Model (WAGEM) for the Netherlands

The WAGEM is an applied general equilibrium model of the Netherlands used for assessing agricultural and environmental policy changes on Dutch agriculture and the economy as a whole. In the model, an industry can produce more than one commodity and one commodity can be produced by different industries. The aggregate output is composed of two hypothetical aggregate inputs: an aggregate intermediate input (composed of several commodities) and primary factor input (labour and capital). Assumptions for modelling trade are used which state that commodities imported or exported are imperfect substitutes of domestically produced and used commodities.

A two-stage procedure has been adopted to represent consumer preferences:

- private household income is distributed over household expenditures and savings according to a Cobb-Douglas function; and
- the household (the one 'representative' household) has linearly homogenous constant elasticity of substitution demand function.

Total gross savings in the economy is equal to the sum of private and government saving, capital depreciation and the (fixed) surplus on the balance of trade. Investment demand is modelled using Leontief input demand functions.

Trade and transportation services (market margins) are produced by different industries. The use of these services is incorporated in the buyers' prices of each commodity at three levels in the model:

- export (export margins);
- total domestic use (wholesale margins); and
- household demand (retail margins).

The authors of WAGEM have highlighted its advantages, but an example of a policy run-through has yet to be demonstrated:

- the model incorporates accounting consistency, e.g. budget constraints are taken into account as is market balance, in addition to basic macro-economic identities such as the equality of saving and investment;
- the model is theoretically consistent which makes the interpretation of the results relatively easy in spite of the fact that the model is rather large;
- all inter-industry effects are explicitly modelled, there is no need to make a choice regarding which linkages are important enough to model as in partial equilibrium models; and
- in partial equilibrium models some results are quite obvious, e.g. introducing supply quotas reduces welfare. In a general equilibrium model this is no longer always the case because the effect on the rest of the economy are also taken into account.

5.2.5 Limitations of GE Models

GE Models start at the 'top', with a representation of what should happen if the economy in question conformed to the assumptions of the model. As a result, some commentators take the view that these models are too abstract for the real world rather than the more traditional 'bottom-up' approaches that take a set of observations relating to what is actually happening (CEC, 1996). In addition, most GE models start from the assumption that there is no unemployment, i.e. the labour market is in equilibrium. Consequently, any change in employment levels is a result of voluntary decisions on the part of the workforce. This aspect of GE models causes studies to reach different conclusions regarding the impact on employment of implementing environmental policies, and subsequently leads the OECD (1997a) to advise that the results of studies using models should be considered with reservations. As with input-output models (see below), the inherent complexity of GE models means that the amount of time and effort required to collect the basic data, and build a suitable model, is often prohibitive.

5.3 Input-Output Models[23]

5.3.1 Introduction

Wassily Leontief developed the first set of basic input-output tables in 1936. By 1941, Leontief had produced input-output tables for the US economy for 1919 and 1929.

Input-output analysis is based on the fact that in modern economic systems linkages exist between activities. Basically, each production activity acts both as a 'supplier' and a 'buyer'. In its capacity as a supplier, each activity sells its output to other sectors and to final consumers. As a buyer, each activity purchases outputs from other sectors, as well as labour, capital, raw materials, etc. (the so-called primary inputs). Therefore, the total value of output from any one activity, not only comprises the value of intermediary goods and services purchased from other sectors, but also the value of primary input consumed directly in the production process. Input-output analysis is essentially, therefore, a method of systematically quantifying the linkages between various sectors in an economy.

It is relatively straightforward to incorporate discharges of residuals and inputs of environmental resources into input-output models; 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. The general form of an economic-environmental quality model is shown in Figure 5.2.

Figure 5.2 Basic Input-Output Model for Economic-Environmental Quality

[23] As input-output models also simulate economic inter- and intra-sectoral relationships, they are sometimes classified under the same broad heading as GE models (CEC, 1996).

Source: Hufschmidt *et al* (1990)

5.3.2 Basic Structure of an Economic-Environmental Input-Output Model

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 5.3. The three sectors considered are energy, industrial processing and commercial services. Looking at row 1, for example, the table shows that the energy sector produced 22,730 ECU of total output, of which, 5,608 ECU was purchased by the industrial processing sector, 1,658 ECU by the commercial services sector, and 15,464 ECU by final consumers[24]. To produce this output (consider column 2), the energy section required 3,208 ECU and 1,543 ECU worth of inputs from the industrial processing sector and

Table 5.3: Hypothetical Three Sector 'Transactions' Accounting Framework (ECU)

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

24 In the input-output tables for the UK final demand (end consumers) consists of consumers' expenditure, government final consumption and capital formation.

commercial services sectors respectively, and primary inputs costing 17,979 ECU[25]. The total cost of all inputs to a sector equals the total value of output from that sector.

Assuming that each sector uses inputs strictly in fixed proportions[26], production technologies remain constant, no economies of scale apply, marginal and average input ratios are the same, and no input substitution occurs, an input-output model can be constructed from the data in Table 5.3 (Hufschmidt *et al*, 1990). By dividing each entry in Table 5.3 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:

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

where w_{ij} is the total output of the *i*th sector purchased by the *j*th sector in the base year, q_j is the total output of the *j*th sector in the base year, a_{ij} is the amount of output from the *i*th sector used in the production of a unit of output from sector *j* and *n* is the number of sectors in the economy. Table 5.4 displays the direct requirement coefficients derived from the transactions data contained in Table 5.3. For example, 0.1411 ECU and 0.0679 ECU of processed goods and commercial services, respectively, are required directly to produce 1 ECU of energy output.

Table 5.4: Direct Requirements Coefficient Matrix (ECU)

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 ECU, 60,000 ECU and 45,000 ECU respectively, the total output of each sector is determined by solving the following set of simultaneous equations:

$$q_1 = 0.0q_1 + 0.1063q_2 + 0.0377q_3 + 25{,}000$$

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

26 That is, a 1 per cent increase in energy output will result in a 1 per cent increase in all inputs.

$$q_2 = 0.1411q_1 + 0.0q_2 + 0.1796q_3 + 60{,}000 \qquad (2)$$
$$q_3 = 0.0679q_1 + 0.4007q_2 + 0.0q_3 + 45{,}000$$

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 ECU.

As **inter-sector linkages** are modelled, the total output of each sector greatly exceeds the final demands, i.e. 25,000 ECU, 60,000 ECU and 45,000 ECU. This illustrates the capacity of input-output models to account for indirect effects in addition to direct effects. In the above example economy, the value of total (direct) output is less than the value of total (direct and indirect) output, as determined by the model, as backward and forward production linkages are taken into account. In terms of policy evaluation, models may be used to compare two distinct states of the economy, the difference between the two states providing a measure of the 'net benefits' of the policy intervention (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 (see Annex 2). The Leontief coefficients derived for the above three sector model are shown in Table 5.5. These coefficients 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 5.5: Leontief Inverse Matrix: Total Requirements Coefficients

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:

$$q_1 = (1.0231)(25{,}000) + (0.1339)(60{,}000) + (0.0626)(45{,}000) = 36{,}428 \text{ ECU}$$
$$q_2 = (0.1690)(25{,}000) + (1.0997)(60{,}000) + (0.2039)(45{,}000) = 79{,}380 \text{ ECU} \qquad (3)$$
$$q_3 = (0.1372)(25{,}000) + (0.4497)(60{,}000) + (1.0859)(45{,}000) = 79{,}277 \text{ ECU}$$

5.3.3 Incorporating Environmental Quality Effects

If detailed base year information for each sector's interactions with the environment are available[27], these can also be represented in coefficient form and used to assess environmental effects. Table 5.6 provides a hypothetical data set for the three sector model described above.

Table 5.6: Base Year Environmental Quality Parameters			
Production Sector	**Energy**	**Processing**	**Services**
Land (hectares)	25,000	60,000	120,000
Water (m^3/year)	50,000	80,000	20,000
PM_{10} (t/year)	100,000	10,000	5,000

Table 5.7: Matrix of Direct Environmental Quality Coefficients			
Production Sector	**Energy**	**Processing**	**Services**
Land (hectares)	1.0999	1.1373	2.7293
Water (m^3/year)	2.1997	1.5164	0.4549
PM_{10} (t/year)	4.3995	0.1896	0.1137

Direct coefficients are derived for each entry in Table 5.6 by the corresponding base year total output level for each sector. Coefficients for land, water and particulates are shown in Table 5.7. 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:

$$
\begin{aligned}
r_1 &= e_{11}q_1 + e_{12}q_2 + e_{13}q_3 + \ldots + e_{1n}q_n \\
r_2 &= e_{21}q_1 + e_{22}q_2 + e_{23}q_3 + \ldots + e_{2n}q_n \\
&. \\
&. \qquad\qquad\qquad\qquad\qquad\qquad\qquad\qquad\qquad\qquad (4) \\
&. \\
r_s &= e_{s1}q_2 + e_{s2}q_2 + e_{s3}q_3 + \ldots e_{sn}q_n.
\end{aligned}
$$

Therefore, assuming final demand for energy, processed products and commercial services in 2000 is the same as that given above, 346,712 hectares of land and 236,565 m^3 of water would be consumed, and 184,324 tonnes of PM_{10} would be emitted. As Figure 5.2 indicated, further analysis could be conducted to translate these residuals into ambient environmental quality, and ultimately into estimates of monetary damage, e.g. with the use of the impact pathway methodology developed as part of the ExternE Project (CEC, 1995).

5.3.4 Incorporating Employment Effects

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

It is also possible to use input-output 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, 1997a). The employment/output coefficients are treated in the same ways as the environmental quality coefficients discussed in the previous sub-section.

5.3.5 Examples of Input-Output Models

National and regional economic-environmental input-output models have been built for several countries. The primary purpose of such models is to forecast residual discharges, mainly air emissions. Examples are listed in Table 5.8.

In addition to forecasting discharges of residuals, input-output models can be used to assess both the direct and indirect effect of controlling flows of residuals from economic activities. The cost of installing new abatement technology, for example, could be added to the primary input costs of an input-output model. The impact on the whole economy could then be estimated with the use of a Leontief inverse matrix, including **direct** and **indirect effects** on final demands, sectoral outputs and primary inputs.

Alternatively, it is possible to alter the matrix coefficients to reflect the implementation of abatement technologies.

In a non-environmental context, input-output models have been applied in various transport-related studies to examine the wider impacts (i.e. direct and indirect effects) of investment in infrastructure on a regional or national economy (see Table 5.3 in CEC, 1996 for a summary of these studies).

5.3.6 Limitations of Economic-Environmental Input-Output Models

In the above example, the economy was only dissaggregated into three sectors. However, a realistic regional model may have thirty or forty sectors, and a national model anywhere between one hundred and three hundred sectors. An obvious drawback of input-output models is, therefore, the amount of time and effort required to collect the basic data and build a suitable model. It is estimated that construction of even a modest regional model would require several man-years' effort, not to mention a great deal of co-operation from industry and government (Hufschmidt *et al*, 1990). Nevertheless, many countries already have input-output tables, e.g. the Netherlands national accounting system which is based on input-output tables, and, as has already been mentioned, such tables do exist for the UK. It may be that these could be usefully extended to include environmental quality effects with relatively less effort than starting from scratch.

Table 5.8: Examples of Economic-Environmental Input-Output Models		
Authors	**Country/Region**	**Residual(s) Modelled**
Leontief and Ford (1972)	United States	Air emissions
Ayres and Gutmanis (1972)	United States	Air emissions/solid wastes
Cumberland and 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 and Laurent (1972)	Charleston (US)	Air emissions/water discharges
Miernyk and Sears (1974)	West Virginia (US)	Air pollution abatement
Koppl *et al* (1996)	Austria	Energy Taxation

The assumptions underlying the construction and operation of input-output (environmental quality) models are subject to criticism, however. For example:

- it is not always the case that fixed coefficients (assumed in input-output models) accurately describe real production relationships or environmental quality effects, especially when non-marginal changes in output are anticipated; and

- the extension of the linearity assumption into diffusion modelling may not adequately handle background concentrations of residuals and threshold effects (Hufschmidt *et al*, 1990). Furthermore, it implies that marginal outputs in an industry require the same composition of inputs as does the average unit of output.

In addition, according to OECD (1997a), input-output models neglect several relevant channels of indirect impact concerning employment effects, including:

- the effects stemming from price and wage adjustments;

- the induced consumption effects of the incremental employment (i.e. the multiplier effects); and
- the induced investment effects (i.e. the accelerator effects).

With the above problems, the huge number of variables considered, and the resulting complexity, input-output models can be cumbersome and frustrating for the decision maker.

Input-output models may be used to compare two distinct states of the economy; pre-policy intervention versus post-policy intervention. Again, the difference between the two 'states' represents the net (economic) benefit/cost of implementing the policy in question (normally expressed in terms of a change in GDP). Input-output models, like GE models, therefore still provide a measure of policy worthiness, which may be used as an input to the decision making process.

5.4 Multipliers

5.4.1 Overview

If the cost-benefit analysis of a proposed environmental policy is based on appropriate shadow prices for factor inputs and policy benefits, the estimated net present value will include all direct costs and benefits (and potentially some indirect). In other words, the measure of net present value includes the direct benefits obtained by labour and companies that service the policy, as well as the surpluses associated with changes in economic rent and prices. The total net value of the policy, i.e. inclusive of direct, indirect and secondary effects, can therefore be expressed as the product of the estimated net present value and an appropriate 'multiplier' (Abelson, 1996):

Total value = Net present value x Multiplier. (5)

Basically, income and output multipliers measure ultimate changes in income and output with respect to changes in some autonomous component of expenditure or direct income created by policies/projects. Employment multipliers relate changes in total employment to changes in autonomously determined employment.

5.4.2 Logic Behind the Multiplier Effect

To understand the workings of the multiplier effect, suppose that autonomous spending (denoted by $\Delta\bar{A}$) increases, perhaps as a result of investment in pollution abatement technologies. Production must expand to meet exactly the increase in demand. Output accordingly expands by $\Delta\bar{A}$. This increase in production gives rise to an equivalent increase in income, which in turn, via the standard Keynesian consumption function, $C=\bar{C}+cY$, results in secondary expenditures equal to $c\Delta\bar{A}$ [28]. Production again expands to meet the increase in spending. The corresponding increase in production, and then

[28] Lower case *c* is the marginal propensity to consume and *Y* is output. The marginal propensity to consume is the increase in consumption per unit increase in income.

income, is $c\Delta\bar{A}$. This gives rise to a further round of secondary spending equal to the marginal propensity to consume times the increase in income, i.e. $c(c\Delta\bar{A}) = c^2\Delta\bar{A}$. Since the marginal propensity to consume is less than 1, the term c^2 is less than c. Secondary expenditures in round three are therefore less than in round two.

The total change in income for successive rounds of increased secondary spending is given by equation 6:

$$\Delta AD = \Delta\bar{A} + c\Delta\bar{A} + c^2\Delta\bar{A} + c^3\Delta\bar{A} + \ldots = (1 + c + c^2 + c^3 + \ldots)\Delta\bar{A} \qquad (6)$$

As c is less than 1, equation 5 is a decreasing geometric series, the sum of which is equal to:

$$\Delta AD = \Delta\bar{A}(1 - c)^{-1} = \Delta Y \qquad (7)$$

Therefore, the cumulative change in aggregate output is equal to a multiple of the increase in autonomous spending. The multiple, $(1 - c)^{-1}$, is the multiplier[29].

It is evident from equation 7 that the size of the multiplier is directly proportional to the size of the marginal propensity to consume. The magnitude of the marginal propensity to consume, and hence the multiplier, will be specific to a given region or country. For example, the multiplier in France is between 1.1 and 1.2 whereas, in the United Kingdom estimates range between 1.1 and 1.8 (OECD, 1997a).

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 case, there are no secondary economic benefits[30].

In general, multipliers are a way of providing ‘order of magnitude’ estimates of the value of ‘growth in income and/or output’ resulting from economic activity that would not otherwise occur in the absence of a certain policy intervention; although with respect to multipliers of this type we are normally talking about expenditures associated with capital projects. In practice, however, they tend not to be included in CBA. As Abelson (1997, p.22) notes:

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29 It is also possible to express the multiplier in terms of the marginal propensity to save, i.e. 1/*s*.

30 This is one of the underpinning reasons for employment not being taken into account in cost-benefit analyses.

"... this is because similar benefits can generally be obtained by alternative uses of the project's resources ... secondary benefits are generally viewed as transfers between communities rather than a net addition to community income ..."

This argument for ignoring secondary benefits does, however, presume that unemployed resources are completely mobile and are distributed evenly throughout the economy. Differences in secondary benefits may thus occur, and may affect total output and incomes, if these assumptions do not hold.

A detailed discussion of 'multipliers' and 'multiplier effects' is beyond the scope of this report; further reading may be found in: Haveman (1976); Armstrong & Taylor (1985); McGuire (1983); Black (1981); and Sinclair & Sutcliffe (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.

6. AIR POLLUTION CASE STUDY

6.1 Policy Overview

This case study examines an appraisal of a new SO_2 air quality objective for the UK[28]. The appraisal was undertaken for the UK Department of the Environment, Transport and the Regions (DETR). Aspects of the appraisal of specific interest to this study include:

- an assessment of the least cost method of meeting the National Air Quality Strategy (NAQS) objective through nationally applied policy measures; and

- an assessment of the benefits of the resulting reductions in SO_2 emissions.

The Environment Act (1995) placed a requirement on the Secretary of State for the Environment "*to prepare and from time-to-time review a strategy for the management and improvement of air quality in the UK*". This management plan is outlined in the UK National Air Quality Strategy Document, and contains a series of proposed objectives and targets for eight priority pollutants. The objectives and targets have been set with due consideration to health effects, current and future ambient concentrations, and the practicality of actions. The result of the strategy has been a proposed regulation to control concentrations of benzene, 1,3-butadiene, carbon monoxide, lead, nitrogen dioxide, ozone, PM_{10}, and sulphur dioxide.

The study conducted for the DETR analysed the costs and benefits of controlling sulphur dioxide (SO_2) concentrations to levels proposed by the Expert Panel on Air Quality Standards (EPAQS), i.e. 100 ppb SO_2 measured over a 15 minute averaging period. The Panel's recommendation was based primarily on exposure studies that investigated the effect of SO_2 on people suffering from asthma, but commented that it is likely that similar effects may be observed in patients with other chronic lung diseases. The NAQS sets the objective of meeting the EPAQS limit by 2005, at the 99.9th percentile annual compliance level.

6.2 Scope of the Analysis

6.2.1 Introduction

While the regulatory appraisal looked at numerous economic and technical aspects of meeting the NAQS objective for SO_2, the two aspects of specific interest to this study

28 It should be noted that only certain aspects of the original appraisal have been drawn upon here; specifically, only those elements deemed necessary to highlight the costs and benefits typically included in the appraisal of air pollution policies. Unfortunately, the emissions baseline, assumptions regarding the reference scenario, the cost data and the emission savings data are 'restricted commercial'. Therefore, only limited data is reported in this case study, and the cost and emission savings data that is reported has been slightly modified to protect its confidentiality. As a result, it is not possible to make direct comparisons between the cost and emission savings of the UK NAQS for SO_2, and similar data pertaining to, for example, the EU Acidification Strategy. Since the benefit valuation data is in the public domain, however, it has not been adjusted. Nonetheless, it is not believed that any omissions or adjustments to the data will bias the conclusions of this section.

relate to the cost assessment and the benefit assessment. The principle features of these assessments are presented below.

6.2.2 Compliance Cost Assessment

This section of the appraisal estimated the least cost method of meeting the limit for SO_2 concentrations proposed by the EPAQS. Predicted maximum 99.9th percentile concentrations and 99.8th percentile concentrations for 2005 throughout the UK were estimated using statistical and atmospheric dispersion models, and possible areas of exceedence of the target limit were identified. Normalised average costs for a range of abatement options were then calculated, which were used in conjunction with a linear programme to calculate the least cost method of meeting the target limits in the identified areas of exceedence.

Abatement Options Considered

The assessment considered a range of abatement options and applied these within a number of industrial sectors, following the industrial classification system employed in the Digest of UK Energy Statistics (i.e. SIC(80)). Where applicable, the options were considered for two fuel types: heavy fuel oil (HFO) and coal, as these fuel types were identified as the major sources of SO_2 in the UK.

The economic sectors included within the assessment and the fuels employed are listed in Table 6.1.

Table 6.1: Economic Sectors and Fuels Considered in the Analysis

Economic Sector	Type of Fuel
Electricity supply industry (ESI)	Coal and oil
Large industry	Coal and oil
Small industry	Coal and oil
Iron and steel industry	Coal and oil
Refineries	Oil only
Domestic	Coal and oil

The abatement options considered within the assessment were:

- various flue gas desulphurisation processes: limestone or lime slurry scrubbing; lime slurry scrubbing with spray drying; dry sorbent injection; and a hybrid process of dry sorbent injection followed by a carbon reactivation stage;
- two modified combustion processes; integrated gasification combined cycle (IGCC) and pressurised fluidised bed combustion (PFBC); and
- various fuel switching options, e.g. 3 per cent sulphur oil to natural gas, 1.25 per cent sulphur coal to natural gas.

The choice of abatement option made within each sector is likely to be affected by regulatory pressures other than those arising from the implementation of the NAQS objective, it was assumed that all measures already agreed were in place[29]. Separate treatment was, however, afforded to the proposed EC Directive on the Sulphur Content of Liquid Fuel (SCLFD). If implemented, the SCLFD would directly affect the quantity of SO_2 emitted from industrial operations and, in turn, impact upon the levels of concentration which formed the focus of the appraisal. Therefore, the two principle scenarios considered were:

- Scenario I: SCLFD in place; and
- Scenario II: SCLFD not in place.

Cost-Effectiveness of Abatement Measures

The study used a 'bottom-up' approach to assess the cost-effectiveness of each measure in reducing SO_2 emissions. This approach involves use of discounted cash flow techniques to reduce the stream of non-recurring (i.e. capital) costs, and recurring (annual operating and maintenance) costs, associated with each measure, to a single present value in a given base year. To facilitate comparison between measures with different operating lives, and to ultimately provide cost estimates that are comparable with the annual benefits of reduced SO_2 emissions, the present value of the total cost stream of each measure was annualised over the forecast period of a plant's operating life. This involved calculating the equivalent annual cost of each measure. An indicator of the cost-effectiveness of each measure was then derived by normalising the equivalent annual cost to the resulting emission reduction, to give an abatement cost in ECU per tonne of SO_2 abated.

Hence, the costing methodology adopted in the study involved the application of equation 1 to selected economic and environmental performance data obtained for each measure under consideration.

(1)

where

= the average cost of measure *i* in abating one tonne of SO_2;

= the non-recurring cost of measure *i* in period *t*;

= the operating and maintenance costs of measure *i* in period *t*;

r = the appropriate discount rate;

T = the operating life of measure *i*; and

29 Estimates of 2005 emissions from power stations were based on the Environment Agency's predictions. For the purpose of predicting emissions from other sources for 2005, the 1995 dissaggregated National Air Emissions Inventory was scaled to approximately match the Department of Trade and Industry's estimates of future emissions for the Central Growth Low Fuel Price scenario with total upgrade, with and without SCLFD. The assumptions over measures in place made for the Central-low case therefore apply in the baseline emission forecasts. Similarly, the baseline scenario takes into account improvements in energy efficiency which are built into the Central-low case.

= the quantity of SO_2 abated by measure *i* in period *t*.

The non-recurring (capital) costs represent the one-off costs incurred to install/implement the abatement measure. Data were not dissagregated between different capital cost components, e.g. between purchased equipment costs, installation costs, etc. In contrast, the annual recurring costs were dissaggregated into the following two categories: (1) annual energy costs (e.g. coal, gas, oil, electricity); and (2) annual non-energy costs (e.g. labour and materials).

The lowest cost abatement measures for each fuel, as estimated by equation 1, are presented in Table 6.2 and Table 6.3 for the 'with SCLFD' and 'without SCLFD' scenarios respectively.

Table 6.2: Least Cost Abatement Measures: with SCLFD

Sector (fuel source)	Abatement Measure	Average Cost (ECU per tonne SO_2 abated)
Industry (coal)	switch to natural gas	830
Industry (oil)	switch to natural gas	870
ESI (coal)	switch to low sulphur coal	140
ESI (oil)	switch to natural gas	870
Refineries (oil)	switch to natural gas	870
Domestic (coal)	switch to low sulphur coal	50
Domestic (oil)	switch to natural gas	1,300

Table 6.3: Least Cost Abatement Measures: without SCLFD

Sector (fuel source)	Abatement Measure	Average Cost (ECU per tonne SO_2 abated)
Industry (coal)	switch to natural gas	50
Industry (oil)	switch to natural gas	1,300
ESI (coal)	switch to low sulphur coal	830
ESI (oil)	switch to natural gas	300
Refineries (oil)	switch to natural gas	140
Domestic (coal)	switch to low sulphur coal	300
Domestic (oil)	switch to natural gas	300

Predicted Exceedences

Exceedances of the 2005 target limit at the 99.9th percentile compliance level were estimated using a statistical model. This model related concentration of SO_2 to the point and area source emission estimates. As a consistency check, concentrations were also predicted using a dispersion model, the ADMS-2 model.

A linear programme was then used to calculate the least cost method of meeting the NAQS objective. This programme assumed that for each source sector, the required abatement could be achieved through the implementation of a single, lowest cost abatement option for each fuel (oil and coal). The lowest cost abatement options, applicable under each scenario, used in the analysis are shown in Tables 6.2 and 6.3. Essentially, the linear programme calculated the required UK-wide abatement in each sector to give the least cost.

The results of the least cost analysis for policies applied nationally to reduce exceedences arising from both point sources and area emissions, are shown in Tables 6.4 and 6.5 for the 'with SCLFD' and 'without SCLFD' scenarios respectively.

Table 6.4: Minimum Compliance Cost with SCLFD

Sector (fuel source)	**Annual SO_2 Abatement Required (tonnes per year)**	**Abatement Measure**	**Annual Compliance Cost (M. ECU per year)**
Large Industry (coal)	36,700	switch to natural gas	30.5
Small Industry (coal)	41,900	switch to natural gas	34.8
Small Industry (oil)	48,500	switch to natural gas	42.2
Total	127,100		107.5

Table 6.5: Minimum Compliance Cost: without SCLFD

Sector (fuel source)	**Annual SO_2 Abatement Required (tonnes per year)**	**Abatement Measure**	**Annual Compliance Cost (M. ECU per year)**
Large industry (coal)	36,800	switch to natural gas	30.5
Large industry (oil)	41,200	switch to natural gas	12.4
Small industry (coal)	17,200	switch to natural gas	14.3
Small industry (oil)	130,000	switch to natural gas	39.0
ESI (oil)	15,500	switch to natural gas	4.7
Refineries (oil)	4,100	switch to natural gas	1.2
Total	244,800		102.1

Without the SCLFD in place, the lowest cost option for achieving much of the required abatement is a fuel switch from oil to gas, thus the linear programme seeks to achieve much of the required abatement by this means. For small industrial oil-burning sources, 130,000 tonnes is abated, whereas for small industrial coal-burning sources, 17,200 tonnes is abated.

With the SCLFD in place the total abatement required from all sources falls from 244,800 tonnes to 127,100 tonnes. The abatement efficiency of measures applied to oil-burning plant falls significantly, and switching fuel from coal to gas becomes more economically attractive than the equivalent switch for oil-burning plant. The abatement cost for a switch from coal to gas for industrial sources is less than for a switch from low-sulphur HFO to gas under this scenario. As a consequence, the linear programme increases the proportion of abatement achieved through a switch from coal to gas. The abatement achieved from small industrial coal-burning sources thus rises to 41,900 tonnes. With the overall fall in total abatement required, and a higher relative cost for switching from oil to gas, the total abatement required from small industrial oil-burning sources falls. The linear programme predicts that only 48,500 tonnes will be abated from this source.

On the basis of a nationally-implemented policy, it was predicted that the necessary abatement could be achieved without requiring additional abatement measures in the domestic sector. Similarly, no additional abatement would be required from coal-burning generating plant within the ESI.

6.2.3 Benefits Assessment

The purpose of the benefit assessment was to ensure that the costs of meeting the new EPAQS were balanced against the benefits of reducing external damage costs associated with emissions of SO_2.

Existing results from the European Commission's ExternE Project: National Implementation: Aggregation Task, were used to derive estimates of external damage costs per tonne of SO_2. This figure was then used to estimate the benefits, in terms of damage costs avoided, of the additional abatement required to meet the new EPAQS.

Impacts Considered in the Study

The damage costs estimated by Krewitt *et al* (1997) relate to impacts on the following three principle receptors;

- human health;
- crops; and
- materials.

These receptors are considered below.

Effects on Human Health

Within this receptor group adverse health effects (both acute and chronic) which have proven causal associations with emissions of SO_2, mainly in the form of sulphate aerosols, were assessed. Table 6.6 lists the acute mortality and morbidity health endpoints considered, while the chronic mortality and morbidity health endpoints considered are given in Table 6.7. Details of the valuation approach and derivation of the unit values are provided in CEC (1995), Metroeconomica (1997) and Hurley and Donnan (1997).

Table 6.6: Receptor: Human Health: Acute Effects

Receptor Sub-group	Impact	Unit Value (ECU)
Acute Mortaility:		
Total population	Acute Years of Life Lost	98,000
Total population	Acute Value of Statistical Life	3,100,000
Acute Morbidity:		
Total population	Cerebrovascular hospital admissions	7,870
Total population	Respiratory hospital admissions	7,870
Total population	ERV[1] for COPD[2]	223
Total population	ERV for asthma	223
Total population	Hospital visit for child croup	223
Above 65 years	Congestive heart failure	7,870
Adults	Restricted Activity Days	75
Adults	Bronchodilator usage	37
Adults	Cough	7
Adults	Lower respiratory symptoms	7.5
Children	Bronchodilator usage	37
Children	Cough	7
Children	Lower respiratory symptoms	7.5

Notes:

1 Emergency room visits

2 Chronic Obstructive Pulmonary Disease

Table 5.7 Receptor: Human Health: Chronic Effects

Receptor Sub-group	Impact	Unit Value (ECU)
Chronic Mortality:		
Adults	Chronic Years of Life Lost	98,000
Acute Morbidity:		
Adults	Chronic bronchitis	105,000
Children	Chronic cough	225
Children	Case of chronic bronchitis	225

Effects on Crops

Within this receptor group, the direct gaseous effects of SO_2 on crop yields, the costs of changing the amount of lime needed to deal with the acidification of agricultural soils, and the benefits of oxidised N deposition acting as a fertiliser, were assessed (see Table 6.8). Damages to crops were valued using international prices. Costs of liming and the benefits of oxidised N deposition were valued using the market prices of lime and fertiliser.

Table 6.8: Receptor: Crops		
Receptor Sub-group	**Impact**	**Unit Value (ECU)**
Barley	Yield loss (dt)	5.9
Potato	Yield loss (dt)	9
Sugar beet	Yield loss (dt)	5.3
All crops	Added lime needed (kg)	0.0171
Rye	Yield loss (dt)	17.1
Oats	Yield loss (dt)	6.2
Wheat	Yield loss (dt)	10.6

Effects on Materials

The corrosive effect of SO_2 on galvanised steel, limestone, mortar, natural stone, paint, rendering, sandstone and zinc were assessed (see Table 6.9). Valuation was performed using repair and maintenance costs. The damage cost estimates are therefore essentially ‘replacement costs’.

Table 6.9 Receptor: Materials		
Receptor Sub-group	**Impact**	**Unit Value (ECU)**
Galvanised steel	Maintenance surface (m^2)	1,015
Limestone	Maintenance surface (m^2)	280
Mortar	Maintenance surface (m^2)	31
Natural stone	Maintenance surface (m^2)	280
Paint	Maintenance surface (m^2)	13
Rendering	Maintenance surface (m^2)	31
Sandstone	Maintenance surface (m^2)	280
Zinc	Maintenance surface (m^2)	25

Total Damage Costs per Unit of SO_2

The estimated annual total damage costs resulting from emissions of SO_2 from the UK’s power sector are presented in Table 6.10. Damages to each of the three receptors considered totalled 20,170 M. ECU; of which 19,130 M. ECU related to health effects, 150 M. ECU to crop damage and 890 M. ECU to material damage. Given that the total damage cost figure is based on annual SO_2 emissions of 2,729 thousand tonnes, average damage costs per unit of SO_2 are equal to 7,390 ECU.

Table 6.10: Aggregate Damage Costs from Emissions of SO_2		
Receptor	**Damages (M. ECU)**	**Percentage of Total**
Human Health		
Acute Effects:		
Mortality	5,900	29.25
Morbidity	580	2.88
Chronic Effects:		
Mortality	11,700	58.01
Morbidity	950	4.71
Sub-total	19,130	94.85
Crops	150	0.74
Materials	890	4.41
Total	20,170	100.00

Benefits of Additional Abatement Required to Meet the New EPAQS

The benefits of meeting the new EPAQS, in terms of damage costs avoided, were then approximated by taking the product of the average damage cost per unit of SO_2, and the additional abatement required to meet the new standard as reported in Tables 6.4 and 6.5. The results for the 'with SCLFD' and 'without SCLFD' are presented in Table 6.11.

Table 6.11 Annual Benefits of Achieving the NAQS Objective			
With SCLFD		**Without SCLFD**	
Additional Abatement (tonnes SO_2 per year)	Benefits of Additional Abatement (M. ECU per year)	Additional Abatement (tonnes SO_2 per year)	Benefits of Additional Abatement (M. ECU per year)
127,100	940	244,800	1,800

Sources of Uncertainty

There are basically two major sources of uncertainty in the above benefit estimates:

- uncertainty relating to the transfer of the damage costs from the power sector to other sectors; and
uncertainty over the damage cost estimates themselves.

In using the average damage cost derived from the ExternE Aggregation Sub-task to estimate the benefits of meeting the new EPAQS, it is implicitly assuming that the damage per tonne of SO_2 emitted from power stations in the UK is equivalent to the damage that would result from a tonne of SO_2 emitted from the manufacturing, refinery and domestic sectors. This is a questionable assumption. For the results to be directly transferable, damage costs must be insensitive to characteristics of the emission which might influence its dispersion and impacts; stack heights, for example, can have a major effect on these factors. Most emissions from electricity generation are from high stacks, whereas, sources of emissions from small industry and the domestic sector are usually closer to the ground. Impacts per tonne of SO_2 from the latter sources are therefore likely to be greater, particularly as the majority of these sources are located in urban areas where the density of receptors (humans and materials) is relatively high.

Furthermore, due to the major changes in the UK generating mix since 1990, damage costs derived from 1990 emission data are not likely to reflect the situation that prevails at present. With the changing fuel mix, emissions of SO_2 from electricity generation today will inevitably differ from those of 1990. As a result, damages per tonne of pollutant based on 1990 emissions may differ from those based on emissions levels today. This is especially the case if the spatial distribution of generating technologies has changed simultaneously with the fuel mix over the period.

Uncertainties associated with the individual damage cost estimates, are discussed elsewhere in this report, and will therefore not be repeated here.

Formal statistical methods may be used to provide an indication of the credibility of damage cost estimates, for example, see Rabl & Spadaro (1997). An alternative approach to integrating uncertainty into the analysis is to rank impact categories by order of perceived uncertainty; once the impacts are ranked, calculated benefits (damages avoided) may be added sequentially, starting with those with the perceived lowest uncertainty, and compared with the policy's estimated costs. This approach was recently applied in a CBA conducted for the UK DETR[30]; the rankings used in the study were derived from surveys of relevant experts and policy makers. Applying the same general rankings to the cost data reported in Table 6.10, the annual benefit estimates of meeting the new EPAQS were recalculated to reflect the perceived uncertainty of the original damage costs. The 'modified' results are shown in Table 6.12.

30 Cost Benefit Analysis of Proposals Under the UN ECE Multi-pollutant, Multi-effect Protocol. An unpublished report prepared for the UK DETR by AEA Technology, Metroeconomica and Eyre Energy Environment.

ÿTable 6.12: Uncertainty of Annual Benefits of Achieving the NAQS Objective

With SCLFD		**Without SCLFD** ÿ	
Impact Categories (in order of increasing uncertainty)	Benefits of Additional Abatement (M. ECU per year)	Impact Categories (in order of increasing uncertainty)	Benefits of Additional Abatement (M. ECU per year)
Materials/Crops	48.41	Materials/Crops	92.70
Acute Morbidity	75.48	Acute Morbidity	144.54
Acute Mortality	350.43	Acute Mortality	671.04
Chronic Morbidity	394.71	Chronic Morbidity	755.82
Chronic Mortality	940.00	Chronic Mortality	1,800.00
All	940.00	All	1,800.00

6.2.4 Comparison of Costs and Benefits

Estimates of the total annual benefits of meeting the new standard range from approximately 48 to 940 M. ECU with the SCLFD to between 93 and 1,800 M. ECU without the SCLFD; compared to the annual costs of compliance of about 108 M. ECU and 102 M. ECU respectively. This suggests that:

- in order for the new EPAQS to generate annual net benefits with the SCLFD, the benefit estimates must at least include crops, materials, acute morbidity and acute mortality benefits; and

- in order for the new EPAQS to generate annual net benefits without the SCLFD, the benefit estimates must at least include crops, materials and acute morbidity benefits.

If uncertainty associated with the benefit estimates is not a concern, then there are significant net benefits associated with implementing the new SO_2 air quality standards. The latter results are summarised in Table 6.13.

Table 6.13: Comparison of the Costs and Benefits of Meeting the NAQS

Scenario	**Compliance Cost (M ECU per yr)**	**Benefits of Abatement (M ECU per yr)**	**Net Benefits (M ECU per yr)**	**Benefit/Cost Ratio**
With SCLFD	108	940	832	8.7
Without SCLFD	102	1,800	1,698	17.6

As noted in the introduction to the section, the principle purpose of the NAQS targets for SO_2 is to reduce adverse human health effects. Taking acute mortality, for example, the damage costs figures given in Table 6.10 are based on 5,937 Years of Life Lost, or 7,916 cases (assuming, on average, a loss of 0.75 years per case). This translates to 2.9 cases per thousand tonnes of SO_2 (i.e. 7,916 cases divided by 2,729 kt SO_2). Hence, the estimated number of acute mortality cases avoided is 369 cases (i.e. 2.9 cases per kt SO_2 times 127.1 kt SO_2) with the SCLFD and 710 without the SCLFD (i.e. 2.9 cases per kt SO_2 times 244.8 kt SO_2). Normalising the compliance costs to the number of acute mortality cases avoided for each scenario, equates to 0.293 M. ECU per 'life saved' and 0.144 M. ECU per 'life saved' with the SCLFD and without the SCLFD, respectively. Note that these measures of cost-effectiveness will decrease considerably if the number of chronic mortality cases avoided are included in the denominator (it was not possible to identify the number of chronic mortality cases avoided from the data set). Of course, the morbidity, crop and material benefits are ignored in this type of cost-effectiveness calculation.

6.3 Comprehensive CBA of the Policy

6.3.1 Introduction

In the previous section, all costs and benefits assessed as part of the original appraisal, were identified. The cost-side of the equation included the incremental capital and operating and maintenance costs associated with the proposed abatement measures. The benefit-side of the equation comprised damages avoided on the following principle receptors: human health; crops; and materials. However, this is not a complete picture of all the costs and benefits that might accrue from the NAQS objective for SO_2. Recall that the aim of an idealised CBA is to determine the 'full' net social value of the policy in question. To this end, it is necessary to identify all parties (consumers and producers) affected by the policy, whether directly or indirectly, and then value the effect of the policy on their welfare as it would be valued in monetary terms by them. Subject to this premise, in this section all potential direct, indirect and secondary effects of the NAQS objective are identified. Where possible, the potential significance of these effects and state-of-the-art regarding their valuation is discussed.

The section concludes by placing the full myriad of possible effects arising from the NAQS objective into a cost-benefit accounting framework, from which its 'full' net social value may be determined. It is this value that should be compared with alternative policies with similar or overlapping objectives.

6.3.2 Cost-Side

Social Welfare Changes in Related Markets

To meet the NAQS objective for SO_2 at least cost would involve a major switch from coal and oil fired plant to natural gas fired plant, across a variety of sectors. An obvious consequence is that the demand for coal and fuel oil would decline, whereas the demand for natural gas would increase. This, in turn, would induce changes in the demand for other factor inputs consumed in the supply of those fuels. In this case, the regulatory costs directly imposed on the sectors listed in Tables 6.4 and 6.5, affect a myriad of other (related) markets.

However, do effects in these other markets need to be examined to ascertain the policy's total impact on social welfare? An answer to this question is useful in defining the scope and boundaries of practical applications of economic analysis to this type of environmental policy.

Relevance of Indirect Effects to Appraisal of NAQS Objective

Whether the omission of (indirect) effects in related markets was appropriate in the appraisal of the NAQS objective for SO_2, is difficult to say in the absence of further, more detailed analysis. However, it is possible to make some crude hypotheses. The impact of increased and decreased demand in the natural gas and refined petroleum products markets respectively, may not be significant enough to warrant consideration; basically, there are other, larger markets for refined petroleum products than those affected by the proposed policy and, furthermore, the price of gas has not increased in recent years despite large increases in demand - possibly indicating the presence of a relatively flat supply curve. One might expect, however, the proposed policy to have significant adverse impacts on the coal industry; the sectors affected by the policy essentially represent the primary markets for domestic (deep-mined) coal production. If these markets were to be closed, the cost to the coal industry, and those employed by it, could be substantial.

As noted elsewhere, each of these energy markets not only acts as a supplier, but also as a buyer. Therefore, for example, downsizing the coal industry may also adversely impact upon suppliers of factor inputs – suppliers of mining equipment, for instance. Input-output calculations represent an appropriate analytical framework for examining all these 'related' impacts simultaneously.

Of possibly greater concern, and a principle reason why analyses of environmental policies should examine related markets, is the presence of pre-existing externalities in those markets. As mentioned, meeting the NAQS objective involves substituting coal and oil for natural gas. In this case, the substitute good harbours less environmental risks than the two goods it replaces; that is, if one accepts that the marginal damage costs estimates for each fuel cycle, calculated as part of the ExternE project, accurately reflect the full life-cycle external costs of each fuel.

Employment Effects

Following the same terminology and definitions used by OECD (1997), and adopted elsewhere in this study, environmental polices can have several employment effects, including:

ÿ positive and negative effects, and
ÿ direct and indirect effects.

Protecting the environment may have positive employment effects through the creation of jobs. Here, switching from coal and oil fired plant to gas fired plant will require the installation of capital equipment, which may create additional employment. These direct employment effects (direct in the sense that they are stimulated by expenditure in the sectors targeted by the policy) will only be realised in the short-term however.

Conversely, environmental policies may also result in the loss of jobs in certain sectors of the economy. In terms of meeting the NAQS objective, it is extremely likely that negative employment effects will be realised in the coal industry, particularly deep-mine collieries. For example, under the SCLFD scenario, required annual SO_2 savings as a result of switching from coal to gas are 78,600 tonnes. This implies that 3.131 million fewer tonnes of coal per annum will be burned in the small and large industrial sector (based on a standard emission factor for industrial (coal burning) plant of 25.1 kg SO_2 per tonne of coal). The average price of coal for all industrial users in 1995 was £37.27 per tonne. Hence, meeting the NAQS objective under the SCLFD scenario would result in a reduction in total sales (gross output) of about £116.71 million. Total gross output in the mining and agglomeration of hard coal sector in 1995 amounted to £2,357.7 million; equivalent to £143,762 per head (total employment in the sector was 16,400, of which 14,000 are operatives and 2,400 are administrative, technical, clerical). Therefore, an 'order of magnitude' estimate of the employment losses in the coal sector as a result of the proposed policy is about 811 jobs, of which 692 and 118 correspond to 'miners' and 'professionals', respectively. Of course, this type of analysis is oversimplified; its purpose is purely illustrative.

Furthermore, the operation of gas-fired boilers is less labour intensive than those using coal or oil as a fuel source. Therefore, job losses may be experienced in those sectors directly, as well as indirectly, affected by the policy. In general however, one might expect positive employment effects to accrue from the operation and maintenance of abatement equipment. These employment effects may be more long-lived.

All of the (least cost) abatement measures considered above involve fuel switching, and thus do not involve the purchase of, for example, end-of-pipe abatement technologies. Consequently, the potential for direct (employment) benefits accruing to the providers of such equipment is not relevant in the context of this case study.

As mentioned above, meeting the NAQS objective may induce changes in the use of intermediate goods and services, e.g. in sub-sectors that supply inputs to the fuel producers, or the suppliers of coal-, oil- or gas-fired plant. Changes in the pattern of inter-industry demands as a result of the policy of fuel switching may, in turn, result in further (indirect) employment effects throughout the economy. In general, such indirect effects take longer to filter their way through the economy.

Clearly, undertaking a full assessment of the (net) employment effects of meeting the NAQS objectives is beyond the scope of this case study. But given the above discussion it may be important in identifying the full implications of the proposed policy.

Impacts on Competitiveness

The implementation of the suggested measures to meet the NAQS objective may produce negative impacts on competitiveness which, depending on their significance, may lead to corporate, sectoral, or national economic decline. A loss of competitiveness may, in turn, impose additional costs not foreseen by the original analysis.

At the level of the company, meeting the NAQS objective may have implications for competitiveness if it imposes costs on some companies which are not imposed on their competitors. However, it may not always be the case that environmental regulations impose costs on companies; even where they do the costs may not be significant enough to affect competitiveness; or the regulation may generate benefits for the company to offset the costs.

Implementation of the NAQS objective may adversely affect the competitiveness of certain sectors and, depending on the severity of the affect, this may be marked by Bankruptcies and job losses in those sectors. If the affected sectors are major export earners, and imports remain constant, then exchange rate depreciation may occur, introducing import-inflation into the economy, with further indirect negative macro-economic effects. This may be interpreted as a decline in national economic competitiveness.

In the long-term however, the UK economy will restructure so that other companies take the place of those made uncompetitive by the new standard. The new companies, however, may not be as productive as those they replaced. Loss of competitiveness of important economic sectors could result in substantial transaction costs and, perhaps, a higher equilibrium rate of unemployment in the economy. Economic restructuring is therefore not without its costs.

Uncertainties in the Compliance Cost Estimates

It should not be forgotten that the estimates of the direct costs of meeting the NAQS objective, may be erroneous. A number of reasons have been hypothesised as to why *ex-ante* estimates of cost may differ from actual out-turn costs, including:

- the importance of the timing and dynamics of regulation, and the implementation of abatement technology by industry. Most companies have short- and long-term financial plans, around which capital investment programmes are formulated. Depending on when a capital investment is made during a company's business cycle, the opportunity cost of capital is likely to vary. This, in turn, will influence the outturn cost of any proposed investment;

- the role of regulatory requirements, and in particular the comparison between a prescribed processes vs. a limit based approach. It is generally accepted that allowing a company flexibility to achieve a pollution reduction target is relatively more cost-effective than prescribing exactly how the company should meet the target; the limit based approach allows companies to find the cheapest way to reduce pollution. For example, some companies will find it cheaper to change their raw material inputs, others may carry out more recycling, others may institute more on-site effluent treatment plant, and so on;

- future developments in abatement technology. This relates to the development of new technologies and changes in the cost of existing technologies over time. With respect to the latter, the literature suggests that for some commodities the marginal cost of production, and thus product price, will decline over time. If capital costs are anticipated to decline with time, the implementation date of a specific technology will therefore influence the cost actually incurred. The decline in production costs is depicted by learning, or progress curves;

- omissions from the cost components. Clearly what is, and what is not, included in the *ex-ante*, non-recurring and recurring cost estimates will affect the final measure of cost-effectiveness. For example, the non-recurring cost estimate of an abatement plant may only include the cost of purchasing the equipment. The cost to the implementing agent, however, will also include the indirect and direct costs associated with installing the equipment;

- standard project cost over-runs; and

- the effect of industrial and regulatory interest in cost estimates during regulatory discussions. That is, strategic bias may exist in the cost estimates. For example, studies in the States have revealed that within the same company, *ex-post* costs may be 10 per cent above, or as much as 90 per cent below, *ex-ante* cost estimates, depending on the source of the figures.

Any one of the above factors may result in mis-specification of the cost-effectiveness of each abatement measure, selection of a ‘sub-optimal’ set of policy measures, and subsequently, under/overestimation of the *ex post* compliance costs.

A study has been commissioned by the by the UK Department of Trade and Industry (DTI) to investigate the influence of these factors on compliance cost estimates.

6.3.3 Benefit-Side

Impacts on Other Receptors

As stated above, the benefits of reduced SO_2 emissions relate to avoided impacts and associated damages, on the following three receptors;

- human health;
- crops; and
- materials.

Emissions of SO_2, however, have also been linked directly to adverse impacts on forests. Sulphate aerosols, in addition to having adverse impacts on human health, also reduce visual range in that they scatter light. Furthermore, SO_2 is one of the primary causes of wet deposited ‘acidity’ or ‘acid rain’, along with NO_2. Acid rain can have direct impacts on vegetation, e.g. acid mists may damage the leaves of plantation forest trees, but the most serious impacts occur through the general acidification of ecosystems. Some of the associated effects include loss of biodiversity, and fish kills.

Since impacts on all of these receptors were not considered in the study, the benefit estimates only partially represent the total foregone damages resulting from reduced emissions of SO_2. That is, they represent a minimum estimate of the benefits of meeting the NAQS objective. Moreover, it is unlikely that all impacts on every affected endpoint within each of the three principle receptor groups is accounted for in the benefit analysis. For example, the damage cost estimates of the direct effects of SO_2 on crop yields but only considers six crops. There is every possibility that other crops may be adversely affected from exposure to SO_2.

Generally accepted impacts of SO_2, and related species, are listed in Table 6.14: a ‘+’ sign denotes the inclusion of the impact in the benefit analysis; a ‘-’ denotes the omission of the impact.

The omission of impacts on natural ecosystems, forests and visual range was not an oversight on the part of those who undertook the regulatory appraisal; rather it was a reflection of the difficulties associated with state-of-the-art valuation concerning these receptors. In fact, damages to forests and ecosystems and loss of visual range are typically not included in most air policy appraisals at present. Some of the reasons for their omission are highlighted below.

Table 6.14: Accepted Human and Environmental Impacts of SO_2 and Related Species

Species	**Impact**
SO_2	+ Human health - mortality
	+ Human health - morbidity
	+ Crop yield
	+ Building materials
	- Forests
Sulphate aerosols	+ Human health - mortality
	+ Human health - morbidity
	- Visual range
Acidic deposition	- Acidification of ecosystems and associated effects (change in species diversity, loss of fish, etc.)

Ecosystem Damage

Studies that seek to value the natural world can be divided into three groups:

- those that seek to place values on individual species, such as the blue whale, bald eagle, etc.;

- those that seek to value whole ecosystems; and
- those that seek to value ecology at the national level.

Markandya (1995) concluded that the first set of studies were not useful in the context of the valuation of air pollution damage. Results could not reasonably be extrapolated to other species, and in any case it is never clear whether the results of these studies are specific to the individual species concerned, or reflect a valuation of the broader ecology on which their survival actually depends. Transfer of results to new areas would also be hazardous.

Drawing on a review by MacMillan (1996), Pearce (1997) reported the results of a number of studies in the second and third groups. These are summarised in Table 6.15 overleaf.

However, the results of these studies have rarely been used in policy related analysis, as there are several problems with them:

- it is not clear to what extent they characterise WTP for protection of natural ecosystems alone, and to what extent they characterise general concerns about the effects of air pollution on health and other non-ecological receptors;

Table 6.15: Ecosystem Benefit Estimates

Study	Year	SO_2 Reduction Taken	WTP per Household per Year
Navrud, Norway, fish only	1988	30% 70%	59 ECU 73 ECU
Johansson and Kristrom, Sweden	1988	100%	514 ECU
ECOTEC, UK	1994	80%	34 ECU
MacMillan, Scotland	1996	100%	337 - 473 ECU

- some of the scenarios selected are unrealistic (e.g. elimination of all SO_2 emissions);

- the extent to which the results are specific to sulphur is unknown - it is possible that they simply relate to the protection of ecosystems against any form of damage from air pollution, and hence implicitly contain WTP for control of nitrogen and VOCs;

- the baseline emissions scenarios assumed were in some cases outdated, even at the time that the studies were being performed. Failure to account for all legislation that has been agreed (whether fully active or not) when formulating surveys, suggests that more political action may be needed to curb critical load exceedence than is really the case;

- studies that consider very large reductions in emissions appear to implicitly assume there is a linear relationship between the area of critical loads exceedence and WTP. The work of Navrud suggests that this is not so. For the first 30% reduction in sulphur emissions an annual household WTP of 59 ECU was found, whereas for the next 40% of emissions reduction the results indicated a figure of only 14 ECU;

- Pearce (1997) comments that the two studies with very high results (both greater than 300 ECU per household per year) both *relate to the "elimination" of pollution damage, and it is possible that respondents were genuinely willing to pay for what they saw as an "end" to pollution damage over very large areas*;

- it is not possible within the scope of this study to present detailed arguments for the conclusions that follow: the reader is referred to Mishan (1988, p.38), Sugden and Williams (1978, p.134) or Arnold (1995, p.84); and

- the studies with the lowest estimates (Navrud and ECOTEC) were both concerned with specific types of ecosystem (salmon and trout in Norway, and [separately] fisheries and upland ecosystems in the UK). The extent to which they can be reliably extrapolated to other countries and other types of ecosystem is unknown.

Loss of Visual Range

There is apparently a major disparity in attitude towards the effect of air pollution on visibility (or more precisely, visual range) between Europe and the USA. Well over one hundred pages of the NAPAP review (1990) were devoted to this subject, which has almost completely escaped attention in Europe. Consequently, all valuation data relating to effects on visibility are from the USA. Extrapolation to Europe is possible, but subject to large degree of uncertainty.

Forest Damage

The assessment of forest damage is more difficult than the assessment of crop damage, largely because of the different life cycles involved. The fact that trees live for a very long time has made the study of pollution effects upon them more difficult. Another major factor is the fact that, whilst agricultural soils are effectively managed through annual cultivation cycles, forest soils are more or less undisturbed, allowing acidification to proceed over time.

Analysis of damage to forests, particularly at the scale required in policy analysis, is prone to significant problems, which result from several factors, including:

- forests grow for many years before harvest (if indeed the trees are harvested), effects should therefore be integrated over the full life cycle;

- forest systems are not managed with anything like the intensity typical of agricultural systems;

- forests serve many purposes, of which wood production is only one; and

- much of the forest remaining in Europe grows in areas subject to, for example, low temperatures, high winds, low soil nutrient status, etc. and is therefore subject to a variety of stresses.

Valuation of effects on forest productivity (leaving aside concern over the appearance of woodlands) is complicated. Any shortfall in forest production can be met in the short-term by increased harvest rates. Problems can thus be offset for many years. Also, increased awareness of the issues involved could lead to changes in management systems which alleviate the problem. Quantified changes in forest productivity are typically valued using information on the price of timber. This represents a major simplification: it would be far more appropriate to use or develop a long term model of forest dynamics, integrating data on management regimes. However, this is beyond the scope of most analyses.

Materials damage

The stock at risk, on which the damage cost estimates are based, does not include buildings of cultural heritage. Available evidence on the valuation of pollution damage to historic buildings, most notably that of Soguel (1996) and Mourato (1997) and Mourato & Danchev (1997), suggests that damages may be significant. Furthermore, individuals' WTP to maintain such buildings is likely to be greater than for utilitarian buildings. Material damage costs are therefore likely to be underestimated.

Crops

The following potential pollution impacts on crops are unquantifiable because of a lack of data:

- stimulation of insect pests;
- changes in the performance of pathogens;
- changes in the interaction between plants and climate (e.g. drought, frost); and
- the effects of interaction between some pollutants.

It is possible that these effects are to some extent implicitly included in the exposure-response functions adopted. However, the artificial nature of experimental conditions makes it unlikely that interaction with these stresses is fully accounted for.

Livestock

There are few reports of the direct effects of air pollution on livestock. Those that exist relate to extreme events ('show' cattle held in London at the time of the Great London Smog being a notable example).

Changes in the yield of pasture grass might be expected to feed through to a change in the rate at which livestock grow, or in the production of other products, such as milk. In another scenario, farmers might be expected to respond to changes in animal production, so that meat and milk yields remain unchanged through the provision of additional resources in the form of concentrates or extra pasture land.

Livestock production is extremely important, with production of cattle, sheep, goats and milk making up a significant proportion of total agricultural output in the European Union. Accordingly, effects on livestock productivity may well be considerable.

Health

The epidemiology of air pollution effects at concentrations typical of those in Europe is all fairly recent. It is possible that some effects have not yet been detected. Coverage of chronic impacts appears particularly patchy. However, any assumptions regarding additional effects would obviously be little more than speculation. Note that not all reported effects are included in every analysis because of a lack of evidence in their favour. For example, the valuation of altruistic effects of air pollution health impacts (i.e. damage relating to concern that someone else is unwell, or has died) is prone to much speculation, and is therefore sometimes excluded from studies.

Climate Change Effects

Several primary and secondary air pollutants affect the radiative balance of the planet and hence affect the climate. However, these effects are extremely uncertain, a satisfactory consideration of them is therefore well beyond the scope of most studies.

Benefits from Secondary Emission Reductions

With respect to each source sector requiring additional abatement, the least cost measure applied by the linear program in all cases, involved switching from either coal or fuel oil to natural gas (recall Tables 6.4 and 6.5). The combustion of coal or fuel oil to produce one GJ of energy typically results in the emission of about 0.88 kg or 1.30 kg of SO_2 respectively; emissions of SO_2 from the combustion of natural gas are negligible. The benefits, in terms of SO_2 reductions, of switching to natural gas are clear.

Standard emission factors for other classical air pollutants (e.g. CO_2, CH_4, NO_x, CO, particulates) also vary with the type of fuel combusted. For example, the combustion of coal or fuel oil typically results in the emission of about 35g/GJ or 23g/GJ of particulates respectively; again, emissions of particulates from the combustion of natural gas are negligible. Switching from coal or fuel oil to natural gas as a source of energy therefore also reduces particulate emissions. This really represents an additional direct benefit of meeting the NAQS, assuming that it is met through fuel switching, which was not included in the original study. Emission of greenhouse gases, NO_x and CO are also likely to be reduced by switching from coal to natural gas.

Links have also been established between emissions of each of these classical air pollutants and impacts on human health, the built environment and natural ecosystems. Moreover, numerous studies have attempted to quantify these impacts in monetary terms. Table 6.16 provides a summary of damages in ECU per tonne for NO_X and particulates from a number of recent studies. All figures are in 1996 prices.

Table 6.16: Estimates of Damages from Recent EU and US Studies

Study	Study Area	Pollutant	Damage (ECU per tonne)		
Krewitt *et al* (1997)	UK/Germany	NO_X	17864		47003
Rowe *et al* (1995)	New York	NO_X	-992	-98	797
Thayer *et al* (1994)	California	NO_X		14241	
CSERGE (1993)	UK	NO_X		1005	
Krewitt *et al* (1997)	UK/Germany	Particulates	22046		60439
Rowe *et al* (1995)	New York	Particulates		20534	
Thayer *et al* (1994)	California	Particulates		46825	
CSERGE (1993)	UK	Particulates		12240	

Notes:

1 Differences in the Rowe *et al* estimates emerge from different sites in New York.

2 Differences in the ExternE estimates are from one study each in the UK and Germany. The NO_X estimates include damages from associated ozone.

Secondary Benefits

As mentioned elsewhere, secondary (benefits) effects refer to increases in income which are induced by multiplier processes following the creation of value added by the policy in question. In terms of meeting the NAQS objective, such effects comprise increased income in related sectors of the economy through backward and forward production linkages associated with the required policy of fuel switching, e.g. suppliers of goods and services related to gas-fired boilers. They also comprise increased income through additional spending induced by policy value added, e.g. any second order effects arising from spending made by additional workers employed to install any required capital equipment.

It should be noted that multiplier effects apply only in so far as full employment of local resources or other supply constraints do not prevail. Where capital projects are undertaken in depressed urban areas or regions, secondary effects may assume considerable importance.

It is not possible to make any generalisations concerning the potential significance of secondary effects arising from meeting the NAQS objective, without employing one of the multiplier approaches listed below – which would represent a significant undertaking – mainly due to the amount of data that would be required.

Uncertainty of the Damage Costs

It is beyond the scope of this case study to even begin to assess the uncertainties that affect the individual damage cost estimates for each receptor; they are discussed at some depth in CEC (1995).

6.3.4 Comprehensive Cost-Benefit Accounting Framework

Recall that the aim of an idealised CBA is to determine the 'full' net social value of the policy in question. To this end, it is necessary to identify all parties (consumers and producers) affected by the policy, whether directly or indirectly, and then value the effect of the policy on their welfare as it would be valued in monetary terms by them. Subject to this premise, a complete accounting framework for assessing the social costs and benefits of meeting the NAQS objective is presented in Table 6.17. This framework incorporates all foreseeable direct, indirect and secondary effects, and distinguishes between those effects included in, and omitted from, the original appraisal [31].

Drawing on the main results of previous sections, an indication of the potential significance and direction of each effect is also provided. A '+' indicates that an effect is likely to yield benefits, whereas potential costs are denoted by '-'. Effects that are perceived to be relatively insignificant are denoted by ''. Furthermore, '+ +' or '- -' indicates that, in the whole scheme of things, an effect is perceived to be relatively significant.

An indication of the state-of-the-art regarding the valuation of each effect is also provided. This essentially reflects the availability of suitable valuation techniques, data, general understanding of the effect, uncertainties, etc. For example, 'good +' indicates that theoretical sound methods exist for the valuation of the effect, data availability is relatively good, and uncertainties surrounding the estimated impacts are minor.

Table 6.17: Complete Cost-Benefit Framework for the Assessment of the NAQS Objective

Effects of Policy	**Type of Effect (Direct, Indirect, Secondary)**	**Possible Magnitude/ Direction of Impact**	**Valuation State-of-the-art**
Included in Original Appraisal			
1. Estimated Compliance Cost:			
a. Non-recurring costs	D	- -	good +
b. Recurring costs	D	-	good +

31 Approaches to valuing each type of effect are discussed elsewhere in this study.

2. Estimated Benefit:			
a. Human health damages avoided	D	+ + +	good -
b. Crop damages avoided	D	+	good -
c. Material damages avoided	D	+	good -
Excluded from Original Appraisal			
3. Effects on Related Markets:			
a. Coal producers	D/I	- - -	fair +
b. Refined petroleum products	D/I		fair +
c. Natural gas supplies	D/I	+	fair +
d. Others	D/I	+ / -	fair
4. Employment Effects:			
a. Sectors directly affected	D	+ / -	good
b. Principle related markets affected:			
Coal producers	D/I	- - -	fair +
Refined petroleum products	D/I		fair +
Natural gas supplies	D/I	+	fair +
c. Others	D/I	+ / -	fair
5. Multiplier Effects:			
a. Employment	S	+	fair
b. Income	S	+	fair
6. Impacts on Other Receptors:			
a. Additional endpoints:			
Human health	D		poor +
Materials	D	+	poor +
Crops	D	+	poor +
b. Other Receptors:			
Ecosystem damage	D	+ +	poor +
Loss of visual range	D		poor+
Damage to forests	D		poor +
Damage to livestock	D	+	poor
Climate change	D		poor
7. Benefits from Secondary Emission			
Savings:	D	+ / -	fair
a. Volatile organic carbons	D	+ + +	good -
b. Particles	D	+	fair
c. Carbon monoxide	D	+ +	good -
d. Nitrogen oxides	D	+ / -	fair +
e. Carbon dioxide	D	+ / -	fair +
f. Methane			

In an ideal world, the ‘full’ net social value of meeting the NAQS objective for SO_2, would be computed as:

- -	1	a
-	1	b
+ + +	2	a
+	2	b
+	2	c
- - -	3	a
	3	b
+	3	c
+ / -	3	d
+ / -	4	a
- -	4	b
+ / -	4	c
+	5	a
+	5	b
+	6	a
+	6	b
0	7	a
+ +	7	b
ÿ0	7	c
+ +	7	d
+ / -	7	e
+ / -	7	f
+	Net Benefit	

If the potential significance and direction of each effect is as indicated in Table 6.17, then the net social value of meeting the NAQS objective might be expected to remain positive, although to a lesser degree than indicated in the original appraisal. At the same time, it should be evident that the magnitude, and possibly the direction, of the net benefit estimate depends on which effects are included in the analysis. In other words, the scope and boundaries of the cost-benefit analysis will have considerable influence on the final results. It should be noted, that 'traditional' cost-benefit analyses of air pollution policies typically include (1a, b) and (2a, b, c), some studies incorporate (3) and (4), fewer (5), and virtually none (6). Whether (7) is included depends on the policy being examined.

As indicated in the last column in Table 6.16, even if an 'all encompassing' policy analysis was desired, valuation state-of-the-art would make this impossible; e.g. valuation methods are inadequate, data availability is limited, resources (in terms of time and money) are insufficient, etc.

Ultimately, however, scope and boundary definition is the responsibility of the decision maker, subject to needs, resources, etc.

6.4 Cost-Effectiveness and Opportunity Costs

The CBA accounting framework given in Table 6.17 can be used to estimate the 'full' net social value of meeting the NAQS objective. It is this value that should ideally be compared with alternative policies with similar or overlapping objectives. As stated previously, the NAQS objective for SO_2 was set with due consideration to health effects. A government, however, may pursue many other policy options to achieve similar health benefits, e.g. road safety, cancer research, health education, anti-smoking by-laws etc. Of course, each of these alternative options will consume resources and yield measurable benefits.

This is essentially what will be explored in the next stage. Now it is possible to compare the full range of potential effects from an air pollution policy, with an alternative policy of achieving similar health objectives. It may be useful to break this analysis down into individual effects, e.g. employment creation and pose the question 'is some environmental legislation a cost-effective way of creating jobs?'.

7. MUNICIPAL SOLID WASTE CASE STUDY

7.1 Policy Overview

This case study examines the cost-benefit analysis undertaken for the Commission on Municipal Solid Waste (MSW) management systems[31]. The overall aim of the study was to prepare "...a quantitative assessment of the environmental and economic costs and benefits related to different ways of recycling, reuse, incineration (with and without energy recovery) and landfilling of MSW...". The study was intended as the basis for determining the most economically desirable combination of MSW treatment methods in each Member State, with the intention of providing "...a substantial input to the elaboration of a comprehensive strategy in the field of MSW management...".

In its Fifth Environmental Action Programme[32], the Commission states that "...the management of waste will be the key task for the 1990s...". This statement follows the publication of a Community strategy for waste management up to the year 2000[33] that set out the waste management hierarchy. Primary emphasis is laid on waste prevention, followed by the promotion of recycling and reuse, and then by optimisation of final disposal methods for waste which is not reused (i.e. incineration with energy recovery, incineration without energy recovery, and finally the landfilling of waste).

To achieve this, the Commission laid down three main objectives together with EC targets up to the year 2000. These are shown in Table 7.1.

Table 7.1: The Management of Municipal Waste as Stated in 'Towards Sustainability'

Objectives	EC Targets up to 2000
Prevention of waste Maximum recycling and reuse of material Safe disposal of any waste which cannot be recycled or reused in order of: - combustion as fuel	Waste management plans in Member States Stabilisation of quantities of waste generated at 300 kg/capita Recycling/reuse of paper, glass and plastics of at least 50% Community-wide infrastructure for safe collection, separation and disposal No export outside EC for final disposal

31 Coopers & Lybrand (1996): **Cost-Benefit Analysis of the Different Municipal Solid Waste Management Systems: Objectives and Instruments for the Year 2000**, report prepared for DGXI of the European Commission.

32 CEC (1993): **Towards Sustainability: A European Community Programme of Policy and Action in Relation to the Environment and Sustainable Development**, Luxembourg, Eur Op.

33 SEC(89) 934 final, September 1989 (since updated by COM (96) 399 final).

- incineration - landfill	Recycling/reuse of consumer products Market for recycled materials Considerable reduction in dioxin emissions (90% reduction)

Following on from the publication of the updated Community strategy for waste management, the Council of the European Union presented a resolution that welcomed the strategy[34]. In particular, the Council called for the "...need for a high level of environmental protection taking account of the potential benefits and costs of action or lack of action and having due regard to the functioning of the internal market..." and shares the belief that "...in accordance with the polluter pays principle and the principle of shared responsibility, all economic actors, including producers, importers, distributors and consumers, bear their specific share of responsibility as regards the prevention, recovery and disposal of waste...".

The original study sets out two policy objectives considered to be of particular relevance:

(1) to internalise the externalities (the environmental costs) associated with particular MSW treatment methods; and

(2) to attain a specific pattern of MSW treatment (e.g. ensuring that no more than *x*% of MSW is sent to landfill or that at least *y*% of MSW is recycled).

However, concentration is given primarily to the development of a policy to promote the internalisation of the external, environmental costs of MSW management.

7.2 Scope of the Analysis

7.2.1 Overview of Approach

Analytical Framework

The study states that "...the most economically desirable combination of MSW treatment methods is that which either minimises the net total economic costs of MSW management, taking into account all private, financial costs and benefits and all externalities, or maximises the net social economic benefits...". To determine the combination of options meeting either criterion, the study attempts to determine both the direct financial and economic costs and benefits and the wider environmental costs and benefits.

Three categories of waste treatment options are considered:

- landfill (with and without energy recovery);
- incineration (with and without energy recovery); and
- recycling.

[34] Council Resolution of 24 February 1997 on a Community strategy for waste management.

Landfill

The two elements giving rise to costs and benefits from the landfilling of waste considered by the study are:

- collection and transport; and
- actual operation of the landfill site.

The total net economic costs of the collection and transport of waste for landfilling are the sum of private and environmental costs, where the latter are associated with vehicle emissions and the associated risk of accidents involving waste vehicles.

For the operation of the site, consideration is given to:

- private operating costs;
- revenue generated by energy recovery; and
- environmental costs (including displacement of low efficiency electricity generation).

Incineration

Incineration is treated in a similar way to landfill, in that a distinction between collection and transport of waste and the actual operation of the incinerator is made. Adjustments are also made to take account of the disposal of the waste remaining after incineration (around 30% of each tonne incinerated).

Recycling

The same basic approach is also adopted for recycling, with four new considerations:

- energy savings via the displacement of virgin materials;

- the user cost of virgin material (assumed to be zero given that "...the materials likely to be found in MSW - tin, steel, aluminium, paper, plastics, glass - are in plentiful supply...");

- varying costs of collection dependent upon the recycling system operated; and

- the adjustment of costs to reflect some waste arisings as only a proportion of MSW can be recycled.

Economic Costs and Benefits

The model determines the current financial costs of each MSW treatment option in each Member State and projects those costs into the future. Adjustments are then made to reflect economic costs and benefits (e.g. through the elimination of transfer payments).

Environmental Costs and Benefits

The evaluation of external costs and benefits relies upon the use of life cycle analysis (LCA) combined with economic valuation. LCA identifies the emissions or impacts for each MSW treatment option, with these then converted into money values via the use of shadow prices based on willingness to pay estimates.

7.2.2 Direct and Indirect Costs

In deriving the financial and economic costs of each treatment option, seven different cost elements were linked to five treatment categories. The cost elements are reproduced in Table 7.2.

Table 7.2: Cost Elements of Treatment Categories as Presented by Coopers & Lybrand

Cost Element	Treatment Category				
	Collection	Transfer	Material Recycling	Incineration	Landfill
Land			✓	✓	✓
Site development			✓	✓	✓
Plant and machinery	✓	✓	✓	✓	✓
Labour	✓	✓	✓	✓	✓
Other operating costs	✓	✓	✓	✓	✓
Post closure				✓	✓
Income from sale of recovered materials and energy			✓	✓	✓

In cases where the derived data were considered to be incomplete, inadequate, or unreliable, cost adjustment factors were used. These factors were used to extrapolate data from countries where it was available, to countries where no reliable data could be obtained. The cost adjustment factors are reproduced in Table 7.3.

All costs and revenues were valued so that they reflect the opportunity costs of the resources used, i.e. all transfer payments were removed and prices derived so as to reflect long-term marginal costs.

Costs for the various treatment options were based on country specific data with no adjustment factors. For example, the collection of mixed refuse, co-mingled and separated is based on Belgian data, both composting and processing costs (automatic and semi-automatic) are based on Danish data.

Table 7.3: Cost Adjustment Factors as Presented by Coopers and Lybrand		
Cost Element	**Adjustment Factor**	**MSW Treatment Option where Applied**
Land	Land prices	Reprocessing, incineration and landfill
Site development	Civil engineering costs	Reprocessing, incineration and landfill
Plant and machinery	None needed	
Staff	Labour costs	All treatment methods
Other operating	Diesel fuel costs Energy prices	Collection Incineration (re)processing composting
Post closure	Civil engineering costs	Incineration and landfill
Levies, etc.	None needed	
Income	Income from electricity	Incineration/landfilling
Operating profit	None needed	

Once the 'basic' costs were derived, the next stage for Coopers & Lybrand was to project these into the future. For this purpose, three scenarios were used:

- the 'as is' scenario (leading to no change in unit costs), i.e. current trends projected into the future;
- the 'green' scenario (an increase in the volume of recyclable material, leading to a reduction in net unit costs); and
- the 'technology' scenario (an addition to the 'green' scenario that reduces unit costs even further).

Table 7.4 presents information on the impact of these different scenarios on a selected number of MSW treatment options (of which there are 31 in total).

From the wide variety of treatment options, seven 'management systems' were defined to be used as integrated approaches:

- Management System 1: all MSW is landfilled;
- Management System 2: as much MSW as possible is incinerated (residue is landfilled);
- Management System 3: based on a bring collection for recyclables;
- Management Systems 4 and 5: based on the commingled collection of recyclables (composting is excluded in the former and included in the latter); and

- Management Systems 6 and 7: based on source separation collection of recyclables (composting is excluded in the former and included in the latter).

Table 7.4: Examples of Changes in Unit Costs					
Treatment		**Base Year Cost (ECU/tonne)**	**Percentage Change in Costs**		
No.	**Description**		**'As Is'**	**'Green'**	**'Technology'**
2	Collection of mixed refuse - urban	51	+4	-22	-26
11	Collection of commingled recyclables: kerbside, blue box	180	+6	-16	-28
15	Processing of paper and board	-57	0	+132	+175
25	Incineration with power recovery	67	0	+1	-11
28	Landfill without energy recovery: urban	32	0	+45	+33

Given these Management Systems, the calculated financial and economic costs were as indicated in Table 7.5.

Table 7.5: Net Economic Costs as Presented by Coopers & Lybrand (ECU/tonne)					
Management System		**Base 1993**	**'As Is' 2001**	**'Green' 2001**	**'Technology' 2001**
1	Landfill	95.3	96.6	93.8	91.4
2	Incineration	156.1	157.6	153.7	148.1
3	Bring recyclables	80.8	82.4	70.2	65.1
4	Commingled	101.3	102.5	87.8	80.6
5	Commingled composting	100.4	101.6	86.0	78.6
6	Kerbside source separated	109.8	111.4	94.8	88.8
7	Kerbside source separated composting	108.9	110.4	92.9	86.8

7.2.3 Direct Environmental Benefits

The starting point for the derivation of economic values for environmental impacts was the LCA. LCA principles were used to identify the environmental burdens associated with the life cycle of the various MSW treatment options. The results from the LCA were then used to investigate the combination of waste treatment options in each Member

State which produces the lowest net social cost. At the outset, boundaries for the LCA were set, and these are repeated in Table 7.6.

Table 7.6: Definitions of Boundaries for LCA

Treatment Option	**System Boundary**	**Unit**
Landfill	Kerbside collection to landfill + landfill process - displaced pollution	Various to add to the functional unit of 1 tonne of waste
Incineration	Kerbside collection to incinerator + incineration process + residues to landfill - displaced pollution	As above
Recycling	Kerbside collection to recycling point (or bring system) + recycling process + residues to landfill - avoided virgin material production	As above
Composting	Kerbside collection to the compost site + composting process	As above
Inputs		
Energy	Starting from the extraction of fuel resources for transport fuels and for electricity	GJ thermal energy per tonne of waste
Outputs		
Energy	Energy leaving the incinerator or landfill	kWh per tonne of waste
Displaced Energy	Extraction of fuel resources are included for the transport fuels and for the energy used	kWh per tonne of waste
Recovered materials	Material collection bank and exit of material recovery facility	Tonnes of material per tonne of waste
Displaced materials	Total impact for the virgin material from raw material extraction to produced material	KG of pollutant per tonne of virgin material and avoided tonnes of virgin material
Compost	Exit of biological treatment plant	Tonnes per tonne of waste
Air emissions	Exhaust of transport vehicles and stack of incinerators and stack of recycling plants and landfill lining/cap and energy generation	KG of pollutant per tonne of waste

Pollutants were divided into two groups within the study. The first were greenhouse gases, with economic valuation based on data taken from Fankhauser (1994). The second set of pollutants included acidifying emissions and valuation of the impacts of these on the environment, broken down into human health, buildings, crops, forests and fresh waters, was based on damage cost estimates taken from the ExternE report (CEC, 1995). It is worth noting that there are more recent estimates of greenhouse gas impacts available.

Further impacts considered include:

- dioxins (excluded due to lack of dose-response data);
- accidents or casualties;
- congestion (not included in the analysis); and
- noise (not included in the analysis).

In relation to these further impacts, only fatalities and serious injuries associated with transport were valued. The different environmental components that were valued are summarised in Table 7.7.

Table 7.7: Summary of External Cost Valuations

Greenhouse Gases	Acidifying Pollutants	Accidents or Casualties
ECU 4/tCO_2 ECU 7/tCO ECU 1,469/tN_2O ECU 86/tCH_4	ECU 4/tCO_2 ECU 7tCO ECU 86/tCH_4 ECU 1,469/tN_2O	ECU 2.6m (fatality) ECU 108,000 (serious injury)

Using the same approach as for the derivation of 'management system' costings, the total environmental costs were estimated, with the results as presented in Table 7.8. Adding together the 'economic costs' (presented in Table 7.5) to the 'environmental costs' provides the estimates of the total net economic costs. These total net cost figures are presented in Table 7.9. As an additional option, it is stated that the "...environmental benefits of source reduction are equivalent to the foregone environmental costs of the most expensive disposal option...". Hence, in a ranking of options, source reduction is clearly number one, as shown in Table 7.10.

Table 7.8: Environmental Costs as Presented by Coopers & Lybrand (ECU/tonne)

Management System		Base 1993	'Technology' 2001
1	Landfill	2 to 20	3 to 16
2	Incineration	11 to 23	11 to 24
3	Bring recyclables	-17 to -282	-
4	Commingled	-79 to -278	-127 to -192
5	Commingled composting	-	-
6	Kerbside source separated	-41 to -230	-101 to -156
7	Kerbside source separated	-	-

	composting		

Table 7.9: Total Net Economic Costs as Presented by Coopers & Lybrand (ECU/tonne)			
Management System		**Base 1993**	**'Technology' 2001**
1	Landfill	97 to 115	94 to 107
2	Incineration	167 to 179	149 to 172
3	Bring recyclables	64 to -201	-
4	Commingled	22 to -177	-46 to -111
5	Commingled composting	-	-
6	Kerbside source separated	69 to -120	-12 to -31
7	Kerbside source separated composting	-	-

Table 7.10: Ranking of Waste Management Options		
Rank	**Total Net Economic Costs**	**Environmental Costs**
1	Source reduction	Source reduction
2	Recycling	Recycling (excluding composting)
3	Landfill	Landfill
4	Incineration	Incineration
5	Municipal composting	Municipal composting

From the full analysis, Coopers & Lybrand draw a number of conclusions:

- recycling and source reduction are, on average, the two most desirable options;
- policy should be geared to promoting further source reduction and recycling;
- most of the environmental costs and benefits associated with MSW management arise from the use of energy and transport;
- a uniform policy across the EU would not be appropriate; and
- no single policy instrument is likely to be adequate, instead a package of measures is required.

However, there are three important caveats:

1) recycling is not consistently preferable to other waste management options;

2) landfill emerges as more attractive than incineration although the present study does not provide a comprehensive assessment of the environmental costs of landfill; and

3) composting, especially at the municipal level, generates relatively high costs.

It should also be noted that the choice of economic values associated with the various pollutants is of crucial importance in the decision to which waste management method is ‘best’. For example, alterations to the damage valuations may result in vastly different conclusions - this may be particularly significant when examining the impacts from landfill. This shall be discussed in more depth in the following sections.

7.2.4 Non-Valued Direct Effects

As highlighted above, there are three impacts that are discussed in the report but not taken forward for full economic valuation:

- dioxins (excluded due to a lack of dose-response data);
- congestion; and
- noise.

The latter two impacts may be considered to be significant in terms of local environmental impact. The valuation of congestion impacts (presumably resulting from waste collection/ transportation vehicles as well as those using their cars for recycling trips) can be valued in terms of the opportunity costs of time spent travelling rather than engaged in work or leisure activities.

This approach is used in the UK by the Department of Transport (in its COBA Manual) to determine the ‘costs’ of road congestion when determining if road improvements or a new road scheme is justified. However, in order to apply this approach, the level of congestion associated with each waste management option would have to be determined, with this then increasing or decreasing the net economic cost per tonne of waste. For example, consider a system that emphasises a large bring scheme (a scheme such as a bottle bank where users ‘bring’ the recyclables to the collection point) located near housing estates, shopping facilities and so on. The aim here is to minimise the use of the car to transport recyclables, thus minimising the ‘total’ level of congestion associated with this management option and thus reducing the net economic cost per tonne of waste (as compared to a scheme based on several smaller recycling centres).

It should be borne in mind that time savings are only one element of congestion costs. There are also changes in vehicle operating costs and emission factors, which in turn lead to changes in local air pollution impacts. In addition, with a ‘bring’ recycling system there are also the further ‘costs’ of the time spent sorting and taking recyclables to collection points and the pollution associated with these activities.

In the case of noise, the impacts can be sub-divided into:

- 'bring' recycling noise;
- noise associated with collection of materials;
- transportation noise; and
- noise from the operation of waste management facilities (be they landfill, incineration, sorting plants, and so on).

The valuation of such impacts could be undertaken utilising willingness to pay approach. Potential transfer values exist and are used in some road planning contexts in the UK, although the values relate to studies undertaken in a number of different countries using both contingent valuation willingness to pay and hedonic pricing methods (see for example Tinch, 1995). However, in practice this would be both time consuming and costly, especially given the need to determine the specific level of noise arising from different sources.

In addition to those impacts identified but not valued within the analysis, a range of other potential environmental, market, employment and wider economic effects may arise from adoption of the various management strategies, including those listed in Table 7.11 and are discussed further below.

In addition to potential impacts not being considered, there are also other waste management options that were not examined by the analysis as highlighted in Table 7.12.

Table 7.11: Impacts not Considered in the CBA

Environmental	**Economic and Social**
Water pollution (e.g. leachate) Other air emissions Visual amenity Recreation Non-use (conservation) values for specific areas House prices	Employment Income distribution Price changes International trade and competitiveness Productivity Industry

Table 7.12: Potential Options not Considered in the CBA

Policy	**Description**
Source reduction (i.e. consume less)	The first step in the waste management hierarchy is to 'reduce', as less waste means reduced waste disposal costs (both economic and environmental). A full analysis would have provided a more robust study and set of conclusions. It is unclear if LCA takes account of the use of fewer natural resources.
Re-use	Re-use as an option is not explicitly explored, yet it remains one of the least expensive management options.
Design for the Environment	Products that are designed to minimise packaging and their environmental impacts may be a viable approach to reducing levels of waste.
Education	An education process detailing how to reduce household waste may be a very effective means of achieving stated aims.

Tighter environmental standards	The tightening of standards on incineration and landfill may move management options towards options with lower environmental costs; equally it may also result in the internalisation of some of their external effects.

Water Pollution

The key concern here is from leachate as a result of the landfilling of waste. Clearly, the impact of leachate is not solely an 'environmental' problem, but can be classed as also resulting in odour problems and visual impacts to those living near a landfill site.

Following the life-cycle approach that Coopers & Lybrand used, alternative waste management systems may also have water polluting impacts, such as:

- deposition of pollutants from incinerators;
- discharges as a result of recycling activities; and
- leachate from large scale composting sites.

A key problem in surface waters is the potentially high biochemical oxygen demand (BOD) of leachate, lowering the oxygen content in watercourses and in turn impacting upon fish and plant populations. It may also be the case that leachate stimulates the growth of algae further restricting oxygen levels in the water.

As stated, the option with the greatest potential for environmental and social damage is leachate from landfills. It may be possible to go one step further and make a distinction between 'old' and 'new' landfills. Older landfills tend not to be self-contained but are simply holes in the ground, increasing the probability for pollutants to leach into the surrounding environment. On the other hand, modern landfills are usually designed through the use of clay and use liners to contain any leaching pollutants.

A quick review of the economics literature, however, has found no valuation studies directly concerning leachate pollution at a policy level (although it might be possible to draw on other water quality related work at a site level, such as values contained within the FWR Manual (FWR, 1996) and the Low Flow Alleviation Guidelines (RPA, 1997)).

Other Air Emissions

Although a number of air pollutants are considered by Coopers & Lybrand, a number of emissions such as dust and heavy metals generated by incineration are excluded. It is the presence of heavy metals that are of particular concern given health implications and the potential for them to enter the food chain.

It is unlikely that the remaining waste management options will add to the impacts contained within the Coopers & Lybrand report. However, it is worth bearing in mind that the processing of recyclables will have waste discharges, many of which will be to air and may be potentially as toxic as those from incinerators. On the whole, however, it is thought that these impacts may be negligible in the overall valuation.

Visual Amenity

The presence of waste management sites (be they landfills, incinerators, or recycling centres) has some form of visual impact. The design of such sites plays an important role in determining the effect on the surrounding area. For example, if a landfill is designed not to be visually intrusive via the use of landscaping, tree planting and so on, local residents or visitors to an area may not be significantly impacted by it. On the other hand, it is difficult to design an incinerator to be 'pleasing to the eye'.

When considering the life cycle of management systems, it may be the case that landfills offer the greatest potential for development after they have been closed. For example, some landfill sites have been utilised as areas of recreation (especially as golf courses). In the long run, therefore, the visual impact of the site may be lessened.

Recreation

As highlighted above, it is possible for landfill sites to be used for recreation purposes after they are closed. However, this does not change their impact whilst in operation. Any waste management centre may pose negative impacts for recreation purposes, individuals may not wish to undertake any form of recreation in view of a landfill site or an incinerator. In the case of an incinerator, individuals may have perceived health risks that influence their decision not to partake in recreation in the area.

House Prices

The impact of a waste management site may be felt via the lowering of house prices in the nearby area. The key problem is establishing the relationship between house prices and the waste management site - there will be a range of additional factors determining property values. As a result, isolation of the waste related impacts may take a large amount of time and statistical analysis.

From a review of the economics literature, the majority of studies examining house prices near waste management sites are US based. Although these values may be used for benefit transfer purposes, extreme caution is merited in so doing as noted earlier. It cannot be argued that the US housing market is similar to the EU, and one would expect the markets to vary significantly across Member States. Cultural, social and economic factors will weigh heavy in any analysis on site specific valuations and are therefore not recommended for transference.

Other

In addition to the above impacts, it is worth considering a point regarding the use of virgin materials. In the case where materials are recycled, this postpones the extraction of raw materials and the associated environmental impacts (disruption, minewater discharges, visual impacts and so on). On the other hand, the recycling of materials themselves results in waste discharges, many of which can be toxic (paper de-inking for example). It is not clear if Coopers & Lybrand tackle these problems and, if they do, the point should be made more explicit throughout the report.

Significance of Non-Valued Effects

From the impacts listed above, Table 7.13 uses a 'magnitude of impact' approach to highlight where the non-valued impacts may be significant. Where the impact is judged to be significant, then its omission from the Coopers & Lybrand study will potentially change its findings. Five management systems have been chosen for the purposes of this table and are numbered as they are in the study.

Table 7.13: Management Systems and Significance of Environmental Impacts

Management System		**Impact**						
		Water	**Air**	**Visual**	**Rec.**	**Non-Use**	**Housing**	**Ind.**
1	Landfill	--	+	-	-	?	-	?
2	Incineration	-	--	-	-	?	-	?
3	Bring recyclables	?	?	NA	NA	NA	NA	NA
4	Commingled	?	?	-	-	?	?	?
6	Kerbside source separated	NA	?	-	-	?	?	?

It can be seen that there is great uncertainty over the potential significance of many of the impacts (in particular non-use) and perhaps the greatest omission from the Coopers & Lybrand study are the impacts of leachates from landfill.

The valuation of the above impacts could be undertaken utilising the following techniques:

- dose-response relationships;
- willingness to pay surveys;
- benefit transfer;
- travel costs;
- hedonic pricing; and/or
- replacement costs.

The failure to consider many of these effects probably relates to their site-specific nature, which cannot be dealt with in a high level study such as this. However, the inability to place an explicit value on such effects does not mean that they should not be highlighted (at least) in the analysis.

7.3 Excluded Indirect and Secondary Costs and Benefits

It is in evaluating the change in net economic costs that the indirect and secondary impacts of a policy may be neglected. Potential wider impacts may include changes in:

- employment;
- income distribution;
- market prices;
- inflationary pressures;
- international trade and competitiveness; and
- productivity.

Employment

The employment creation opportunities associated with some waste management options can be significant. For example, consider a policy that has commingled collection and recycling. The commingled nature of the waste requires that a materials reclamation facility (MRF) is operated to sort the recyclables and, in turn, this means that there is a capacity for a large number of unskilled (and to a lesser degree semi-skilled) jobs available. If this policy is repeated through a number of Member States, a moderate number of new jobs might be created. On the other hand, maintaining 'traditional' disposal options such as landfill and incineration will probably not lead to significant changes in the levels of employment associated with this sector of economic activity.

In effect, the policy with the greatest attraction in terms of employment would be that which maximises employment per tonne of waste (or minimises unemployment). For example, a door to door collection system put into place in Créteil, France led to an increase of 23 jobs (from 22 to 45) without any increase in the costs of waste disposal. Without incineration, the new collection system cost 1.25FF per person per day, as compared to the traditional collection system which cost 1.18FF per person per day. However, when incineration of the final waste was included, the costs were equal at 1.39FF per person per day.

It may also be useful to consider the contribution that employing people in the waste management sector makes to overall GDP. For example, Table 7.14 contrasts the recycling industry with a 'traditional' industry, construction.

Table 7.14: Contribution to Employment and GDP in Two Industries in the UK*

	Gross Value Added at Factor Cost (£m)	Total Employment	Gross Value Added per Employee
Recycling Industry			
1994	177.8	3,500	£51,000

1995	247.6	7,100	£35,000
Construction Industry			
1994	19,349.6	873,400	£22,000
1995	22,948.8	967,700	£24,000
* Source: ONS (1998): **Production and Construction Inquiries - Summary Volume 1995**, London, The Stationary Office.			

As can be seen from the above table, even though the recycling industry contributes far less to GDP (as shown by the 'Gross Value Added' column), it's contribution per employee is much higher than that of the construction industry. This suggests that if employment is increased within the recycling industry (and perhaps throughout the waste management industry as a whole), GDP will increase more than it would have done if employment is encouraged in other sectors requiring a similar skill base.

Income Distribution

There is a tendency for workers in waste management to be categorised in the D/C2 social class index. Incomes within these groupings tend to be lower than other sectors of society. Increased employment within the waste sector will therefore distribute income towards the less well-off in society. This is particularly true in the case of any kind of centralised sorting scheme (such as the MRF discussed above) where unskilled/semi-skilled labour will be used. However, the realisation of such benefits assumes that a 'fair' wage is paid, taking into account the country were they work in, the standard of living and the cost of living.

Potential approaches which could be used within a CBA such as that carried out for MSW to examine this issue include consideration of:

- average wage of workers in the waste management industry;
- change in average wage rate between the different social class groupings;
- average hourly wage per tonne of waste;
- average hourly wage for each social class grouping; and
- marginal changes in wage rates per tonne of waste.

Price Changes

As noted earlier, if prices in related markets change or if the social and private costs associated with these markets differ and quantities demanded change, then the implications of such changes should be examined within a CBA. Within the context of waste management, a number of policy options may lead to changes in price levels throughout the economy.

For example, if producers are given full responsibility for the disposal of their used products (such as packaging, electrical products, vehicles, etc.), the result may be an increase in the price of final products to pay for the additional costs placed on producers.

The opposite may also be true. If producers manufacture goods that minimise the resultant waste generated from their disposal, there may be a cost saving. Such a cost saving may be passed on to the consumer.

Similarly, recycling and reuse of waste materials may lead to changes in the demand for raw materials and hence to shifts in the demand curves faced by the raw materials producers. This in turn could lead to changes in the prices faced by other industries reliant upon the same raw materials or involved in their chain of production.

The degree to which a change in price will be passed on to the consumer will in theory be dependent upon the price elasticity of demand for that particular product, which reflects the sensitivity of demand to changes in price. For example, if a product is 'price inelastic', a change in price will have only a small impact upon the quantity demanded. Where demand is highly sensitive to price (price elastic) then a small change in price will have a large impact on the quantity demanded. Producers are more likely to pass on a price increase in cases where demand is more price inelastic.

From a practical perspective, the incidence of compliance costs will depend on the type of regulation being imposed. If the regulation is not related to the economic activities of a sector, or imposes costs on some companies but not their competitors, then it will be very difficult to pass on the cost burden. If the regulation does neither of these, then the burden will tend to be shared between producers and consumers. The actual proportions will depend on whether the regulation increases variable operating costs, capital investment, and whether we look at the short-term or the long-term - and of course, the slope of the supply and demand curves, as noted above.

However, a range of other factors will also determine the degree to which price changes are passed on to consumers, including factors such as advertising, full-cost pricing, mark-up pricing, oligopolistic structure of markets, social factors, and so on. Understanding the degree to which price is likely to change within this context is more difficult. Some producers may wish to undercut prices to increase demand for their products, or prices may increase (or decrease) in a uniform manner as firms act in unison.

Estimation of the impact of waste management policies upon prices, and in turn, other sectors in the economy, is therefore subject to a range of uncertainties. However, the growing importance of the recycling sector in the UK economy for example, as illustrated in Table 7.15, means that its potential influence on other markets cannot be completely ignored.

Input-output tables could be used to gain formal insights into the structural interdependency between sectors (including the recycling sector), and as mentioned elsewhere, used to quantify the direct and indirect effects of, for example, increased levels of community recycling.

Inflationary Pressures

Closely connected to changes in employment, the redistribution of income and price changes is the threat of inflation throughout the economies of the Member States. There is a tendency for the less well-off in society to spend any increase in income they receive (in other words their propensity to consume is high). This means that there is a possibility for aggregate demand and consumption to increase throughout the economy (hence generating secondary (multiplier) benefits), leading to rising inflation. Of course, such demand-led inflation can be offset with tighter fiscal and monetary policy.

Table 7.15: Contribution to the UK Economy from Recycling (1995)

Factor	**Recycling**	**Recycling of Metal Waste and Scrap**	**Recycling of Non-Metal Waste and Scrap**
Number of enterprises	564	338	226
Total sales and work done (£m)	1,237.6	1,002.9	234.7
Gross output (£m)	1,251.9	1,015.9	236.0
Net output (£m)	319.0	219.2	99.8
Net output (£ per head)	45,181	52,586	34,506
Gross value added at factor cost (£m)	247.6	169.7	77.8
Gross value added at factor cost (£ per head)	35,059	40,712	26,909
Employment (total, thousands)	7.1	4.2	2.9
Wages and salaries (operatives, £m)	57.1	36.3	20.9
Wages and salaries (administrative, £m)	32.7	21.5	11.2
Net capital expenditure (£m)	61.1	43.2	17.9

Source: ONS (1998): **Production and Construction Inquiries - Summary Volume**, London, The Stationary Office.

International Trade, Competitiveness and Productivity

The impacts on 'industry' in general may be wide ranging depending on the type of policy being assessed. For example, an option that follows a similar approach to the DSD system in Germany or the Packaging Covenant in the Netherlands may place additional costs on industry (internalising the external cost generated by their products).

The direct and indirect effects of such policies will impact different industries in different ways. The cost burden will therefore be shifted away from those at the 'end of pipe' i.e. from waste handlers to those producing the products that ultimately result in waste (mostly packaging, but also some products). Impacts will be case specific, but may be significant enough to potentially affect profitability and, in turn, competitiveness and employment.

The aggregate impact on companies may be sufficient to induce macro-economic effects on Member State's trade, competitiveness and productivity. Measures for such impacts may include:

- a change in balance of payments per tonne of waste;
- a change in measures of economic productivity (e.g. GDP, GNP) per tonne of waste (probably not helpful at the plant level); and
- a change in 'green' measures of productivity per tonne of waste (such as those measures in countries 'Green Accounts').

The above are micro programmes relative to the whole economy. The competitiveness impacts will arise from increased costs to key producers. These can be estimated, but should be small. However, it must be remembered that the original study has different options for different Member States and this may result in intra-Community effects on competitiveness and productivity.

Comparison of Impacts

Comparability of the different options does not need to be difficult, even within a CBA framework. Even though CBA is being used, it is possible to present secondary or wider impacts alongside the CBA results to present an overall picture to decision makers. Such a table is given below with '+' and '-' reflecting the potential magnitude of impact.

Table 7.16: Total Net Economic Costs (ECU) and Potential Wider Impacts

Management System		**Base 1993 per tonne**	**Employment**	**Price Changes**	**Income Distribution**
1	Landfill	97 to 115	-	+	-
2	Incineration	167 to 179	-	+	-
3	Bring recyclables	64 to -201	+	-	+
4	Commingled	22 to -177	++	++	++
6	Kerbside source separated	69 to -120	+	-	+

In the example given in Table 7.16, it can be concluded that management system 4 provides not only the least net economic cost, but also the greatest potential for employment opportunities and income redistribution; it is also the most likely to give rise to negative impacts such as increases in price levels (brought about by increases in aggregate demand).

7.4 Comprehensive CBA Framework

If we bring together all of the impacts and associated discussions above, it is possible to prepare an accounting framework for assessing waste management policies across the EU. Such an accounting framework sets out the impacts that were included in the original study together with the additional direct, indirect and secondary impacts that should be considered. Table 7.17 sets out the full framework for one management option (a move to commingled collection).

Table 7.17: Complete Cost-Benefit Accounting Framework for One MSW Management Option*

Effects of Policy	Type of Effect (Direct, Indirect, Secondary)	Possible Magnitude/ Direction of Impact	Valuation State-of-the-art
Included in Original Appraisal			
Costs:			
a. Costs of operation	D	-	Good
Environmental and Social Impacts:			
a. Greenhouse gas emissions	D	+	Good
b. Acidifying pollutant emissions	D	+	Good
c. Accidents and casualties	D	?	Good
Excluded from the Original Appraisal			
Multiplier Effects:			
a. Employment	S	+++	Fair
b. Income	S	++	Fair
Benefits from Abated Emissions:			
a. Dioxins	D	++	Poor
b. Congestion	D	?	Good
c. Noise	D	-	Good
d. Water pollution	D/I	++	Good
e. Further air emissions	D	-	Good
f. Visual amenity	D	?	Fair
g. Natural Resource Consumption	I	++	Poor/Fair
h. Recreation	D/I	+	Fair
Impacts on Related Markets			
a. House prices	D/I	?	Fair
b. Industry	D/I/S	++	Good
c. Price changes	I	?	Fair
d. International trade and competitiveness	I/S	?	Poor
e. Inflationary pressures	I	?	Poor
f. Productivity	I/S	+	Fair
Employment and Wider Effects			
a. Social effects	D/I	++	Poor
b. Employment	D/I	++	Fair
c. Income distribution	D/I	++	Fair

* Policy in this case is the commingled collection for recycling as compared to the current situation.

As a result of the failure to include all of the environmental and economic effects set out in the table, the net social benefit estimates calculated for each of the options by Coopers & Lybrand cannot be said to provide a reliable indication of the full social costs and benefits.
Moreover, it should be noted that there is considerable variation in the damage cost values used within the Coopers & Lybrand analysis compared to those adopted (or developed) as part of other research. Such differences in value may be significant and their importance should be tested.

In reviewing the literature, two studies were identified as having approaches that are similar to that undertaken by Coopers & Lybrand:

- CSERGE *et al* (1993): Externalities from Landfill and Incineration (undertaken for the UK Department of the Environment); and

- Brisson (1997): Assessing the Waste Hierarchy - a Social Cost-Benefit Analysis of Municipal Solid Waste Management in the European Union.

There is some similarity between the work undertaken by CSERGE and the economic analysis for the Coopers & Lybrand study, which is understandable given that CSERGE was involved in the latter. However, there are also some important differences as can be seen when the values given in Table 7.7 are compared to those given in 7.18.

Table 7.18: Economic Values used in 'Externalities from Landfill and Incineration' (CSERGE *et al*, 1993)

Type of Externality	ECU/tonne
Global pollutants	
CO_2 as C	6.2 - 46.8
CH_4	48.2 - 209.1
Conventional pollutants	
SO_2	64.2
NO_x	49.4
TSP	2147.4
Leachate	
Existing landfills	0 - 3.4
New landfills	0
Casualties	
Mortality	1.1m - 3.0m
Serious injury	32,465
Minor injury	656.9

* Converted to ECUs using £1=ECU 1.51.

The most consistent value is that for mortality, while the values assumed for CO_2 and CH_4 differ greatly - being much lower in the Coopers & Lybrand study which relied on more recent valuation data. It is difficult to say therefore whether the estimates for these environmental impacts for both landfill and incineration are more reliable or result in an underestimate. As landfill is given a preference over incineration and composting in the ranking of disposal routes, sensitivity analysis indicating the importance of such differences would be of value. These differences again highlight the importance of what economic values are used in an appraisal and how the results may differ.

In the case of Brisson, the approach adopted is almost identical to that used by Coopers & Lybrand, i.e. the integration of LCA with ExternE and use of the Fankhauser & Pearce economic damage values. The analysis was conducted on three scenarios: a base case (relying on a bring system of recycling); a co-collection at kerbside; and a separate collection at kerbside.

The results for these scenarios for each waste management option are presented in Table 7.19.

Table 7.19: Net Economic Costs as Presented by Brisson (average cost in ECU/tonne)

	Base Case		Co-Collection at Kerbside		Separate Collection	
Option Rank	**Option**	**Cost**	**Option**	**Cost**	**Option**	**Cost**
1	Recycling	-170	Recycling	-131	Recycling	14
2	Landfill	92	Landfill	91	Landfill	96
3	Incineration*	115	Composting	102	Incineration*	119
4	Incineration**	150	Incineration*	114	Composting	133
5	Composting	170	Incineration**	148	Incineration**	155

* With energy recovery displacing old coal electricity generation.
** With energy recovery displacing average EU electricity generation.

As in the Coopers & Lybrand study, recycling and landfill are ranked in the first two places (source separation is not discussed). The values are therefore consistent with those generated in the case study CBA. It is worth noting that although the approaches bear striking similarities, the Coopers & Lybrand study is not referenced in the Brisson report.

Further studies that discuss waste management issues (rather than a whole disposal method or policy) are presented in Table 7.20 at the end of this section. As can be seen from this table, values for particular effects can vary considerably, suggesting the importance of sensitivity testing to any transfer value based CBA.

7.5 The Opportunity Costs Question

The full CBA accounting framework given in Table 7.17 sets out the total net social value of a shift from the current situation to that of commingled sort recycling. This table shows the full impact of such a policy and emphasises the fact that the Coopers & Lybrand study excluded a wide range of impacts from the analysis.

The exclusion of potential impacts makes it difficult to determine what are the opportunity costs of adopting one management system as opposed to another. For example, the opportunity costs of adopting a landfill-based system as opposed to commingled collection and recycling cannot be determined with any accuracy, although the potential importance and relative significance of non-valued effects could be guesstimated. The difficulty in this case, however, is that the true opportunity costs associated with the adopting of one system over another needs to take into account not only specific costs and benefits but also the implications of different strategies at the wider economic level. This suggests that the types of analyses carried out for policies which may have significant site specific and macro level effects may need to be more comprehensive in scope if the true opportunity costs are to be understood by decision makers. As the situation is different in each Member State (and indeed regions within Member States), determining the opportunity costs of adopting one system over another would require more detailed country specific analyses. Of course, the ability to undertake such an ideal appraisal will be restricted in practice by data availability, and resource and time constraints.

Table 7.20: Summary of Waste Related Valuation Studies

Value	Type	
$1414 median WTA $1415 median WTA $1404 median WTA $1054 median WTA	Dichotomous choice: 90% confidence: $852 to $1976 (mean characteristics) 90% confidence: $5623 to $2267 (more income, increased risk, reduced offer) 90% confidence: $1167 to $1641 (older people) above all include protect bids (14% of total respondents) excluding protest bids Compared with open ended WTP bids, only 22% gave positive WTP result, 78% gave zero, 3% were non-protest zeros (responded with hazardous landfills are safe)	Groothuis PA, *Using Co* *Compensation I* *of a LULU: Th*
$1711 mean increase in housing values before a hazardous waste site is closed $837 after	Hedonic study that explicitly calculates the effect of a subjective risk variable on the disamenity value compared to a no risk situation	McClelland et a *Property Valu* *Site*, in Grooth (1997): *Using* *Compensation I* *of a LULU: Th*
Phone survey sign-up: 71.4% (at $6) 74.8% (open ended) 71.2% (dichot. at $6) Mail survey WTP: $2 (open ended) $1 (payment card) $2.43 (dichot. choice) $2.69 (stated pref.)	Survey to assess value of a scheme for landfill gas recovery project and planting of 50,000 trees. Used dichotomous choice and open ended questions for a $6 sign up to the scheme. Mail survey also included WTP	Schulze W et *Contingent Valu* Grant Number R National Center (http://es.epa
Table 7.20: Summary of Waste Related Valuation Studies (continued)		
Denmark (per metric ton): $34 to landfills $29 to incineration Holland (per metric ton): $17.8 landfill tax $0 tax to incinerators UK (per metric ton): £2 ($3) inactive waste £7 ($10.7) other waste South Korea (Seoul) $0.08 to $1 for 5l bags $1,090 to $1,450 for 100l bags	Municipal waste user charges Denmark: quantity of waste reduced, reuse of buildings waste increased, slight increase in illegal waste disposal Holland: main purpose to raise revenue, but secondary purpose of discouraging waste generation. Also promoting incineration with average waste treatment costs of $50 per metric ton. South Korea: household waste can only be disposed of in standardised bags sold in official places. Prices vary slightly from area to area. Amount of waste sent to landfills fell by 40% after implementation of the system (although much of this may have been due to uncontrolled incineration or private disposal). Bags are also not biodegradable and fees too low to cover disposal costs.	OECD (1994), a (http://206.29.48 37a8525
$300 to $495 per annum per mile	Property value studies near hazardous sites Guntermann: open sanitary solid waste landfill reduces the value of industrial land by 45%. Property around closed solid waste landfills are not adversely affected. Landfills sell at 51% discount (small sample) Smith & Desvouges (1986): consumer surplus for each mile between residence and hazardous landfill Kohlhase (1990): distance variable significant in only 1 of 3 regressions, prices	EPA (1996): *C* *Industrially Zo* 12, December 1 (1995): *Sanitar* *Values*, The Jou 5, pp 531-542, a

$461 in premier neighbourhoods $5,000 (4%) rise after landfills closed	increased up to 6.2 miles from a waste site, and premium disappeared around a cleaned site Michaels & Smith (1990): annual benefit from removing waste site, negative in below average areas McClelland, Schulze & Hurd (1990) Average increase was 3% to 4% of mean property value.	
12% at landfill boundary 6% at about one mile.	Empirical model to estimate the price effects of one Minnesota landfill on the value of 708 nearby homes during the 1980s. Beyond 2 to 2.5 miles, adverse effects are negligible.	Nelson AC, et *House Values*,
$4 per household per month	Contingent valuation (most conservative mean household estimate) of the economic feasibility of drop-off recycling in a rural/suburban area of Tennessee.	Tiller KH, et al *for Dropoff* Resource E
Table 7.20: Summary of Waste Related Valuation Studies (continued)		
$1,300 per mile linear $1,700 semi-log (1.2% to 1.6% of average house price for all sites)	Property values near sanitary landfills: studies indicate they are far less likely to impact property values than hazardous sites: Thayer, Albers & Rahmatian (1992): value increases with distance from hazardous and non hazardous waste sites. Function not smooth, curve levels at 1.25 miles from site. Nonhazardous landfill loss 35% of hazardous loss in linear model, 60% in semi-log model. Reichert, Small & Mohanty (1992): for 3 of 5 landfills not significant, one case was but wrong sign, final case variable small and negative. Bleich, Findlay & Philips (1991): no impact for 1,628 house sales between 1978 and 1988. Cartee (1989): four studies which found neither positive or negative effects.	EPA (1996): *C* *Industrially Zo* 12, December (1995): *Sanita* *Values*, The Jou 5, pp 531-542),
$227 per household per annum or an extra $141 for households whose drinking water supplies were at risk of contamination	Contingent valuation used to estimate external costs of siting a landfill in the Carter community of Knox County, Tennessee. WTP to avoid having a landfill in the community was dependent upon household income, size, years in the community, perception of health risks.	Roberts RK *Municipal Lan* *Analysis: A Ca* Economics, Vo
Travel cost estimate: $95,396 CV survey: $47,400 to $72,500 Cost of hazardous waste collection: $1.6 million	A zonal travel cost model to develop a demand curve and consumer surplus estimate for household hazardous waste collection and disposal in King County, Washington. Travel cost method considers trips to the hazardous waste disposal facility. Method fails to account for 'existence value' benefits that users derive from knowing the service is available.	EPA (1995): *Values Travel* 10, December 1 *Travel Cost Me* *Waste Dispo* Management,
Average expenditure per ton was $75	Study of 18 recycling programs, 12 of which achieved less than 4% diversion rates, recycling percentages for the others ranged from 7.89% (Largo, Florida) to 15.61% (Santa Monica, Calif.). Statistical analysis showed a positive correlation between % population completing high school and diversion %. Spending more money on public education or administrative services did not increase diversion rates.	EPA (1996): Newsletter a documents: Wo *Succeed*, Resou D (nd): *Evalu* *Drop-Off Recyc* North Ameri
£5.78 per household per	Waste generation models used to predict total monthly WTP for recycling.	Jakus PM, et a

month		*Rural Househo* Economics,

8. SHORT-CHAIN LENGTH CHLORINATED PARAFFINS CASE STUDY

8.1 Policy Overview

Under the European Community Existing Substances Regulation (EEC 793/93), the UK was designated as the rapporteur for assessing the risks associated with the use of short-chain length chlorinated paraffins (SCCPs - chain length C_{10-13} inclusive) and to consider controls for reducing these risks where they are considered to be too high. To this end, the UK Department of the Environment (DoE) commissioned an assessment of the risks associated with these substances in three applications: textile applications and leather finishing; sealants, rubber and paints; and the metalworking industry.

With respect to the metalworking industry, the assessment concluded that the use of SCCPs poses a risk to aquatic organisms and that, as a result, there is a need for risk reduction measures to be implemented. On this basis, a further study was commissioned to develop a series of possible risk reduction measures and the following were highlighted for further consideration:

- classification and labelling;
- voluntary agreement;
- limit values for emissions and effluent monitoring; and
- marketing and use restrictions.

This case study examines the partial CBA undertaken for the UK's Department of the Environment (DoE) which assessed the advantages and drawbacks (risks and benefits) associated with these risk reduction proposals, with the primary focus being on the adoption of 'marketing and use restrictions'. The results of this study have now been passed on to the Commission for further examination from an EU wide harmonisation and trade perspective[35].

8.2 Scope of the Analysis

8.2.1 Aims and Objectives

The main aim of the study was to assess the advantages and drawbacks of marketing and use restrictions on the use of SCCPs in the metalworking industry to enable a judgement as to whether the benefits to the environment of adopting the restrictions outweigh the

35 RPA (1997): **Risk-Benefit Analysis on the Use of Short-Chain Length Chlorinated Paraffins in Cutting Fluids in the Metalworking Industry**, report prepared for the UK Department of the Environment.

consequences to society as a whole of imposing the controls.

Within the context of EC Regulation 793/93, once a series of risk reduction options have been defined, the associated costs and benefits require identification. Where marketing and use restrictions are proposed, a quantitative analysis should be undertaken to the fullest degree possible.

Stage 1 of the study, therefore, involved a qualitative assessment of the risks, costs and benefits associated with the proposed risk reduction options. It concluded that the classification and labelling option and the voluntary agreement were likely to be ineffective. Stage 2 of the study then focused on an assessment of the risks and benefits associated with marketing and use restrictions and the limit value option. This second stage quantified as many of the identified risks, costs and benefits as possible, with the assessment culminating in a part qualitative, part quantitative assessment.

8.2.2 The Approach

The approach to the study was led by the guidelines presented in Steps 5 and 6 of the *Draft Technical Guidance Document on Development of Risk Reduction Strategies* (CEC, 1995a). Data collection was based heavily on consultation with users, producers and formulators, trade associations, environmental regulatory agencies, relevant government departments and Competent Authorities throughout the European Union.

This data included details of:

- the profile of the UK and European industry, in particular the proportion of large, medium and small metalworking fluid users;
- the consumption and disposal practices of different sized users to assess the associated impacts on environmental exposure;
- the economic implications of the options for the stakeholders involved in the production, sale and use of SCCP-based metalworking fluids, in particular the costs of reformulation and the adoption of new fluids by users;
- whether moving to alternative agents would decrease/increase health risks (focusing on dermatitis and asthma) and whether these changes could be quantified in monetary terms;
- the change in environmental risk from implementation of the options and whether these changes could be quantified in monetary terms;
- the feasibility of implementing the options in the UK and Europe in terms of regulatory and policy instruments;

and also, specific to the limit value option,

- the actual concentrations of metalworking fluids in sewer and surface waters in order to assess the impacts on users of meeting a limit value for emissions; and

- the costs of treating emissions in order to satisfy an end-of-pipe control limit.

8.2.3 The Study Findings

Metalworking fluids are used to cool and lubricate the interface between a metal and its metalworking tool to maximise tool life, reduce energy consumption, and remove metal 'chips' (or 'swarf') from the working area to improve surface finish. Metalworking fluids which contain chlorinated paraffins fall into one of two categories: neat oils or emulsions. The former are supplied ready for use and consist of a base oil (usually a mineral oil) and various additives. Emulsions are similar in make-up to neat oils, but are mixed with water prior to use. SCCPs provide advantages over other fluids in terms of their light colour, low odour, low viscosity and ability to produce high quality finishes during extreme pressure processes.

The main drawback to the use of SCCPs concerns their disposal and the risks associated with their discharge to the aquatic environment. The risk assessment found that the predicted environmental concentrations as calculated using the standard assumptions set out by the EC risk assessment procedures exceeded the predicted no effect levels for certain aquatic species.

The analysis recognised that the environmental success of any risk reduction option would depend on its ability to minimise the risks posed to the aquatic environment from the usage of metalworking fluids. Success would be affected, therefore, by the nature of any alternative fluids and thus the risks associated with their use. Consultation indicated that for most metal-cutting processes alternatives were available and based upon current data that appeared to be of lower environmental risk than SCCPs. It was recognised, however, that risks to the environment and/or human health vary significantly depending on the specific formulation of the fluid. For example, one sulphur/phosphorus mixture for which data exist present 'severe' health risks to skin and eyes.

8.2.4 Marketing and Use Restrictions

The findings indicated that the following impacts would arise with the transition from SCCPs to the alternative fluids:

- changes to the fluid formulation;
- lack of efficacious substitutes for some specialised, severe operations;
- increased formulation and thus fluid prices; and
- reduced tool life.

For the UK as a whole, this switch to alternative fluids was estimated to cost from ECU 6 to ECU 12 million (£4 to £8 million), where this includes both increased fluid costs and costs associated with reduced tool life. The range in these estimates reflects uncertainty as to the fluids which would be adopted, the actual impacts on tool life, the degree to which there may be increased machine down-time or reduced quality finished pieces and any reduced levels of output.

8.2.5 Limit Values

The costs incurred by a metalworking facility under a limit value option were found to depend upon how SCCPs were disposed, the size of the facility, the nature of the fluids used and its location. Thus, although this option would directly control releases of SCCPs into the aquatic environment, it was not possible to determine with certainty the impact it would have on industry. The analysis noted that companies would adopt the least cost response to the imposition of limits, and thus the costs to industry should be no higher than those under marketing and use restrictions and could be lower. In contrast to the marketing and use option, however, there were likely to be higher administrative, monitoring and enforcement costs to the water utilities and environmental regulators under this option.

With regard to environmental risks, control would be targeted at those areas where concentrations exceeded EQOs and thus, if effectively implemented, the risks associated with SCCPs would be reduced to an acceptable level (assuming proper monitoring and enforcement). Where users switch to alternative fluids rather than face increased disposal costs, the comments made above concerning the impacts of marketing and use restrictions for both the environment and human health apply. They will depend on the specific formulation, site specific factors, etc.

8.2.6 Derogations

In addition, the study concluded that marketing and use restrictions provided the preferred form of risk reduction, but that should these be adopted there may be a case for derogations. The most important of these concerned those processes which could not switch in the short-term owing to the use of SCCPs being part of an approved process (under a medical directive for example). The potential need for derogations aimed at ensuring that alternative formulations did not pose greater levels of environmental or human health risks was also flagged.

The study also found that the UK is likely to be impacted more than other EU countries by the introduction of the ban due to a greater reliance on SCCPs by industry. Other EU Member States are less reliant on the use of SCCPs, with the current trend in several already being away from the use of SCCPs and chlorine-based additives in general.

8.3 Comprehensive CBA of the Policy

8.3.1 Impacts on Industry

The study identified the following potential impacts on industry:

- increases in formulation and resulting cutting fluid costs;

- potential need for new capital investment to replace equipment which would only function using SCCPs;
- reduced tool life for some metal-cutting processes;
- increased machinery down-time;
- potential increase in production time, leading to reduced productivity; and
- loss in quality for some products.

Generalised cost data are provided by the report, together with more detailed data concerning the impacts on individual companies. However, it was not possible to place costs on many of the above impacts, to a large extent owing to uncertainties surrounding whether or not the impact would actually be experienced and the magnitude of the effect. Conflicting evidence was provided by different companies as to the nature of the effects, with many who had already switched from SCCPs to alternatives indicating that impacts were experienced in the short-term only.

For some industry sectors, however, the lack of, or limited number of, alternative fluids appropriate for a particular metal-cutting operation was expected to result in impacts being experienced over the medium-term (e.g. five years or so) until better alternatives came onto the market.

8.3.2 Environmental Effects

One of the biggest questions that the study leaves unanswered, however, is what are the actual benefits associated with risk reduction. The source of this problem lies with the EU process of chemical risk management which differs greatly with, say, the approach taken in the US and Canada. In these countries, the process is aimed at providing a full probabilistic assessment of the environmental risks, where this includes details of the consequences associated with different exposure levels. In contrast, the EU approach relies on the calculation of hazard potential in terms of the PEC:PNEC ratio (predicted environmental concentration to the predicted no effect concentration).

In the case of SCCPs, the PNEC value was calculated by a risk assessment following the EU approach. In order to translate, the PEC:PNEC information into a format which would allow monetary valuation of the resulting environmental impacts, further data would be required on concentrations within the aquatic environment and of the exposure-response relationship of different aquatic species to different concentrations. Without such information, it is not possible to place a monetary valuation on the reduction in environmental risks as has been achieved in the previous two case studies (at least to some degree).

8.3.3 Health Effects

Similarly, although the risk assessment did not highlight the health risks associated with the use of SCCPs as being unacceptable, the analysis raises questions over the potential for marketing and use restrictions to lead to the adoption of sulphur and/or phosphorus based fluids which could increase risks to workers. Owing to a lack of safety data and risk information on potential alternatives, these potential impacts could not be quantified and brought explicitly into the analysis.

8.3.4 Employment

As emphasised above, SCCPs are used to aid metal-cutting processes and restricting the use of fluids containing SCCPs will lead to increases in fluid costs and potentially reductions in production efficiency. These effects could, in turn, lead to job losses in the engineering industry.

Engineering firms (in terms of metalworking engineering) tend to work on small margins and any change in cost and production related factors could impact upon such firms greatly. The impact will obviously be greatest on small to medium firms who may rely on a smaller product range and operate on tighter margins than the larger companies. Thus, there may be impacts at the firm level; but across the sector as a whole one would expect any losses in work experienced by one company to be taken up by another company and thus overall levels of employment to remain unaffected. This is because the changes required by the proposed regulation relate to a minor input of production rather than to a more significant input of production.

It is unlikely that the manufacturers of SCCPs who are large chemical companies will be impacted greatly by a complete phase-out, as these firms are likely to shift into the production of the alternative products. Similarly, the intermediary formulators are unlikely to be significantly affected, given that there will still be a demand for cutting fluids and hence their services.

8.3.5 Price Changes

Given the predicted increases in fluid costs and impacts on production efficiency, another consideration relates to the impact which environmental regulation may have on end-product prices, and market prices more generally. In this case, we are concerned with a situation where the regulation will impose costs on some companies but not on others (as many do not currently use SCCPs). Hence, it would be difficult for most companies to pass on any increases in costs, with the exception being those specialist processes where there are only a few firms providing the same metal-cutting service.

Where the prices of metal-cutting services do increase, there is the potential for customers to pass on price increases to secondary industries and consumers. Again, though, this will depend on the degree of competition within the sector, the nature of the cost increase, etc.

It is not expected, therefore, that price increases would result from these regulatory proposals, given the low magnitude of the cost impacts and the fact that these are spread over a large number of companies. Further attempts at quantification would not, therefore, be merited.

8.3.6 Trade and Competitiveness

It should be noted, however, that the proposed regulations would affect the metal-cutting industry in some EU countries to a greater degree than in others. This is due to the fact that restrictions on the use of SCCPs already exist in some EU countries, with industry already having to absorb the increased costs associated with the switch to alternatives. One could argue that the lack of harmonisation in this regard is already creating imbalances at an intra-EU level with regard to trade and competitiveness. The introduction of legislation across the EU could therefore erase any of these imbalances, yet lead to other distortions in the short-term.

8.3.7 Comprehensive Cost-Benefit Accounting Framework

As discussed above, appraisal of an environmental regulation should examine the full social implications of the proposed regulatory changes. In cases such as this one, where the appraisal is concerned with regulation of a minor input of production to a single industry sub-sector, the scope of effects covered by the CBA will be smaller than would be required for regulations which impact upon a significant cost input or which would affect a number of industry sectors (as in the air quality and waste case studies discussed in the previous chapters). Thus, given the nature of the policy, the application of other analytical approaches such as input-output tables would be inappropriate. Table 8.1 provides a summary of the types of effect which should be considered (although some may not be taken forward quantitatively).

Table 8.1: Comprehensive Cost-Benefit Accounting Framework

Effects of Policy	**Type of Effect (Direct, Indirect, Secondary)**	**Possible Magnitude/ Direction of Impact**	**Valuation State-of-the-art**
Included in Original Appraisal			
Estimated Compliance Costs			
a. Non-recurring costs	D	-	Good
b. Recurring costs	D	-	Good
Excluded from Original Appraisal			
Estimated Benefits (damage reduction)			
a. Aquatic environment	D	++	Poor
b. Human health	D	?	Poor
Effects on Related Markets			
a. Secondary products	D	≅	Fair
b. End-consumers	I	≅	Fair
Employment Effects			
a. Metalworkers	D	≅	Good
b. Other Sectors	I	≅	Fair
Risks from Alternatives			
a. Environmental Receptors	D	?	Poor
b. Human health (worker safety)	D	?	Poor

8.4 Cost-Effectiveness and Opportunity Costs

This case study raises interesting questions concerning both the cost-effectiveness and the opportunity costs associated with the potential risk reduction options. Because of the inability to quantify the damage costs avoided as a result of either of the proposed risk reduction measures (owing to the risk assessment providing only an indication of hazard potential), the analysis itself is reduced to only a partial CBA which essentially determines the most cost-effective measure out of those being examined. Monetary estimates of costs are not explicitly and directly compared against monetary estimates of the benefits to the aquatic environment.

In this case, the costs to industry are small, particularly given that they will be spread over a large number of companies. As a result, any opportunity costs associated with the adoption of the proposed risk reduction measures should also be small. Moreover, if one assumes that the actual risks (as opposed to those predicted through a series of hypothetical default assumptions) posed by SCCPs are significant, the proposed measures are also likely to be cost-effective in comparison to other potential policy options for reducing chemical related damages on the aquatic environment.

However, such arguments may not hold true for other hazardous chemicals assessed under the same procedures. In some cases, for example chemicals which are more significant inputs to production, the costs arising from proposed risk reduction measures may be far higher and have more significant direct and indirect impacts on industry and related markets. The inability to go from the risk assessment to a more quantitative analysis of the benefits of risk reduction in such cases will result in there being much greater uncertainty as to the justification of a proposed policy, the associated opportunity costs and hence its cost-effectiveness.

9. OPPORTUNITY COSTS AND POLICY EVALUATION

9.1 Introduction

In Section 2, we define opportunity costs as the foregone benefits of an action, and they arise in any situation where resources are scarce and, thus, some kind of mechanism is required to allocate these resources between different uses. As defined earlier, opportunity costs may arise at two levels:

- as a result of the use of individual resources, e.g. capital, labour and energy, etc. which acts as inputs to a policy or project; and

- at a higher level as a result of the implementation of one policy (or set of policies) over another policy (or set of policies).

In section 9.2, we consider the importance of properly accounting for the opportunity costs associated with use of particular resources when appraising a specific policy proposal. Two distinct cases are examined: the valuation of resources in financial policy appraisal and the valuation of resources in economic policy appraisal. It is important to note that the discussion provided in Section 9.2 relates solely to the valuation of an individual policy's costs and benefits. Although much of the preceding text concerns how impacts should be valued within CBA, the aim here is to make clear the importance of ensuring that each policy proposal is correctly valued before moving on to compare the relative merits of different policies.

Appraisal of the opportunity costs which arise in the selection of one policy over another is then discussed in Section 9.3. This section sets out the decision criteria which should be used at a budgetary level for determining which policies out of the range of potential policies should be adopted from an economic opportunity costs perspective.

As has also been made evident from the discussion in earlier sections and the case studies, the valuation of non-marketed environmental goods is complex and, in many cases, subject to significant levels of uncertainty. In some cases, the problems associated with the valuation of environmental and health effects can be avoided through the use of cost-effectiveness analysis. Section 9.4 looks at how CEA has been used to provide an indication of the potential opportunity costs associated with adopting different policy measures. The advantages of adopting such an approach, as well as the need for caution, are highlighted.

Section 9.5 then concludes by examining the implications of the findings presented here in the context of government expenditure within the EU.

9.2 Opportunity Costs and Valuation

9.2.1 Introduction

The first definition given above relates to the need to consider the opportunity costs of adopting a given policy proposal. Following this principle requires that, within any policy appraisal, all inputs should be valued in terms of the 'true' economic costs of the individual resources which would be consumed, and all outputs from the policy should reflect consumers' willingness to pay for them (at the margin). In order to explore the importance of this concept to policy appraisal more thoroughly, we need to start by examining what is typically included within what we have termed here as 'financial policy appraisals' and then compare this to the requirements of economic policy appraisal.

In financial appraisals, all costs and benefits accruing from a policy are valued as they would appear in the financial balance sheet of a company, i.e. they are measured in terms of market prices. The financial benefits of the policy are, therefore, measured by the market value of the policy's output. In these appraisals, it is generally assumed that markets are competitive and thus that prices provide a minimum measure of what individuals are willing-to-pay for a unit of the policy's output, at each level of output demanded.[36]

At the same time, the (financial) costs of the policy are measured by the actual price paid for the policy's inputs. In a (competitive) market economy, the price at which producers are just willing to supply an input reflects opportunity costs, or the value in their next best alternative use, of the resources used to produce the input.

Where the assumptions that markets are competitive hold, appraisals based on the use of market prices will provide an adequate reflection of opportunity costs for those goods and services which are traded in the market place (which by definition excludes environmental and health related effects). In such circumstances, therefore, the CBA analyst can be confident that, for example, using the market wage rate to value the labour costs of a policy will properly account for the opportunity costs of labour inputs.

In many cases, however, the assumption that markets are competitive will not hold. In

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[36] Note that if the policy's output is sold in a competitive market, with no rationing or price control for the good concerned, and the policy is sufficiently small so as not to change the price of goods, its market price will equal its competitive demand price. In addition, in the absence of taxes and subsidies, the market price of an input will equal its competitive supply price at each level of production. This is the price at which producers are just willing to supply the input, and this price reflects the opportunity costs, or the value in their next best alternative use, of the resources used to produce the input. Where competitive market conditions do not hold, prices may not reflect true opportunity costs.

these cases, market prices are unlikely to reflect true economic values (i.e. opportunity costs), due to what are termed 'market imperfections'. Given that the aim of policy appraisal is to ensure that opportunity costs of adopting a policy are taken into account, some technique is required to correct for market imperfections. This leads us to the concept of shadow pricing within economic appraisal methods. Shadow prices are used to correctly value marketed goods and to place a value on non-marketed (typically environmental and health) goods. This use of shadow pricing is the key factor that distinguishes financial policy appraisal from economic policy appraisal, as is discussed further below.

9.2.2 The Need for Shadow Prices in Economic Policy Appraisal

What does it mean when we say that many of the assumptions required for market prices to reflect opportunity costs fail to hold in reality? Consider the need to value the impacts of a proposed policy on levels of employment. Financial appraisal assumes that, in the labour market, no single supplier can affect the price to be received. In reality, however, individuals may bargain collectively with an employer by using union representation. The resulting wage rate, therefore, may not necessarily reflect the real opportunity costs of taking a job to the individuals affected. Similar issues arise where the mobility of labour to move between jobs normally assumed in appraisals does not exist, perhaps because of costs associated with finding out information about other jobs. Again, using the market wage rate to value labour costs will not reflect the true opportunity costs of the labour input arising from the policy.

In general, market failures (which will distort market prices) may be grouped under four major headings[37]:

- interventions in (including those related to government policies) and failures of goods markets, including the markets for internationally traded goods;
- interventions in (again including those related to government policies) and failures of factor markets, including the markets for labour, capital and foreign exchange;
- the existence of externalities, public goods and consumer and producer surplus; and
- imperfect knowledge.

From this categorisation, it is not difficult to draw the conclusion that market prices will often not reflect opportunity costs. This is even more apparent when considering the nature of environmental policies, which are specifically aimed at correcting the third type of market failure.

With regard to the environment, the failure is more complete in that there is no (or only few) markets for environmental quality or environmental assets. There is no market for

[37] Perkins (1994): **Practical Cost-Benefit Analysis**. Melbourne: MacMillan Education Australia Ltd, p 97.

clean air, for example, for the simple reason that no ownership, or property rights to clean air exist. There are no property rights because it is impossible to exclude others from the use of clean air. Since markets are a function of exchange in ownership of a physical or financial resource, it is easy to see why environmental quality and many other environmental assets have no markets. However, simply because there may be no market for clean air and thus no price for, say, a certain level of air quality, does not imply that air quality is not valued in terms of improved welfare by individuals, for example in avoided asthma attacks.

It is, thus, essential for policy analysis to incorporate all such welfare effects if one is to avoid a potentially misleading outcome to the exercise; failure to take such effects into account in an appraisal will ultimately result in a sub-optimal allocation of resources. The true opportunity costs of adopting a given policy will not adequately be addressed if policy selection is based solely on the valuation of marketed effects, if other sources of value (e.g. aspects of the environment) are neglected solely on the basis that they do not have the attributes needed for them to be traded in a market.

Given these conclusions, in appraising environmental policies, in order to properly account for the opportunity costs of selecting one policy over another, the analysts needs to:

1) attempt to adjust the distorted market prices of the effects in question to better reflect their true economic value; and

derive prices for those effects which are external to normal markets to allow them to be properly accounted for within the appraisal.

Achieving the above requires that the second approach is adopted and that market prices are adjusted by calculating the 'shadow prices' for the relevant inputs and outputs. Failure to make such adjustments may ultimately lead to a decline in National welfare over time, as financial profits, and not 'social or community' welfare, are maximised.

A shadow price reflects the increase in welfare resulting from one unit of an output or input being valued. In practice, shadow pricing usually involves making adjustments to market prices to correct for distortions and to take account of consumer and producer surplus. Equally, shadow prices refer to the values attached to non-marketed effects. The resulting adjusted price should then reflect the true opportunity cost of using an input (its marginal social cost), or individual's willingness to pay for an output (its marginal social benefit).

The need to adjust market prices so that they reflect their economic shadow prices is distinct from the point that public sector funds have their own welfare cost. The latter arises because in order to raise funds, governments have to tax someone, and that taxation causes distortions which lead to welfare costs. The 'marginal cost of public funds' as discussed in the literature is a well established concept. In the EC, for example, a value of 1.4 has been taken, implying that government budget allocations should have a cost of 40% above the nominal values (Auto-Oil programme Report, 1998)[38].

As long as impacts are properly measured in terms of their shadow prices, then decisions concerning whether or not a policy is worthwhile will take into account the full opportunity costs of adopting or not adopting the policy. The methods set out in Sections 3 to 5 of this report recognise the concept of shadow pricing. A full discussion of the process of determining appropriate shadow prices is beyond the scope of this study; most good texts on CBA will discuss this process to some extent.

9.2.3 Uncertainty in Pricing Non-Marketed Environmental and Health Effects

As the preceding discussion has demonstrated, the basis of economic resource valuation is the expression of individual preferences. With regard to the estimation of shadow prices for non-marketed environmental effects, a key concern as highlighted in Section 4 is the degree to which the various valuation techniques are able to reliably capture individuals' preferences.

Section 4 notes some of the key criticisms which surround the use of the various valuation techniques with regard to the valuation of environmental and health effects and, for the sake of brevity these are not repeated here. It is important, however, that the potential implications of such criticisms in terms of the degree to which decision makers and others accept such values and the ability of an appraisal to reflect true opportunity costs are recognised.

Monetary valuations of environmental and health effects may fail to reflect opportunity costs for a number of reasons:

- the base data required by the various revealed preferences techniques are unreliable;

- the valuations derived through the use of survey data are unreliable as a result of, for example, biases in the survey instrument, respondents having difficulty in understanding the valuation question, or the failure to include income constraints within such analyses; and

- the inappropriate transfer of data developed for one environmental risk issue to another issue which varies in key characteristics (such as the nature of the risk generating activity, the characteristics of the risks and the affected population).

Within the context of this study, questions concerning the valuation of changes in mortality risks have arisen, with particular reference to the application of VSL estimates developed for use in road safety to air quality issues. As noted in Section 4, the most recent research in this field (e.g. the work reported in NERA & CASPAR, 1998 and DoH,

38 Auto-Oil II Cost-Effectiveness Study: First Consolidated Report. Presentation to ENVECO Meeting 23-24 April, 1998, European Commission, DGII.

1999), recommends that the transfer of a roads VSL to air quality (and thus other environmental issues) include adjustments for differences in the nature of the risks, the health status of those at risk and the age of those at risk.

The case which is being made by those who are arguing for such adjustments is one of opportunity costs (e.g. Smith, 1998). Given that the aim of valuation of mortality risks is to determine how much of scarce resources should be devoted to the implementation of one policy versus another, it is critical that the valuations reflect the willingness to pay of the affected population rather than the values of a different population and/or for a different risk. It is these same concerns which have led others to propose that valuation of mortality risks be based on the value of life years lost in those cases where the risk of concern is one of premature death as opposed accidental death.

Even if such adjustments are made (assuming the empirical data exist to do so reliably), it must be recognised that the valuation of mortality risks is, and is likely to remain, an inexact science. The same holds for the valuation of morbidity and for many ecological functions and services. The uncertainty which, therefore, surrounds the ability of such estimates to reflect 'true' opportunity costs is often used as the basis for arguing against the use of monetary valuation. The obvious danger of adopting this line of argument within the context of environmental policy appraisal is that the failure to incorporate such welfare effects will ultimately result in a sub-optimal allocation of resources.

Instead, the aim should be to derive the most reliable values possible, incorporate these values in the same cost-benefit equation as appropriately valued marketed goods and services, and provide decision makers with an indication of the level of uncertainty surrounding environmental valuations and the importance of that uncertainty to the choice of option. As demonstrated by the case studies, non-monetised benefits may effectively be excluded from further consideration in the appraisal given the reliance of CBA on monetary valuation unless explicit steps are taken to ensure that this does not happen.

9.3 Policy Selection Criteria[39]

9.3.1 Introduction

The above discussion highlights the importance of ensuring that the impacts associated with individual policies are correctly valued. The overarching decision context must also be recognised in that government policy is itself aimed at addressing market failure. In essence, the raison d'etre for environmental policy is to drive the internalisation of environmental externalities by reflecting true opportunity costs. This then leads to the question of how governments should address the issue of opportunity costs when choosing between a set of policies which are all competing for scarce resources. Whether the expenditure is incurred by public agencies or by the private sector, requirements for spending on one environmental policy measure will reduce the amount of money

[39] In this section the terms 'policy' and 'project' may be used interchangeably.

available for spending on other policy measures. In order to ensure that the returns earned by such expenditure are maximised across all of the competing policy measures (whether environmental or going wider and including other policy areas), examination of opportunity costs at the budgetary level should also take place.

The first question which has to be addressed at this level concerns the degree to which the appraisals of competing policies provide correct measures of opportunity costs and can thus be compared on equal terms. To build on the previous section, consider a policy which has some marketed and non-market (environmental) effects. It may be appraised at three different levels as described by each quadrant in Table 9.1. Only if the policy appraisal may be described as falling in quadrant C will the true net economic costs of the policy be measured.[40]

Table 9.1: Degrees of Policy Appraisal

	Valued at Unadjusted Market Prices	**Valued Using Shadow Prices**
Includes marketed effects	**A**	**B**
Includes marketed and non-marketed effects		**C**

Now assume that a decision-maker needs to compare the relative merits of three different policies, each with varying levels of similar effects. A policy appraisal of type A should not strictly be compared with an appraisal of type B or C, or B with C. In each case the extent to which the true economic value of the policy has been identified will vary. In such cases, if the decision-maker is not fully aware of the type of analysis conducted, what is included and what is not, he or she may select a policy which yields lower net economic benefits and thus has high opportunity costs.

For example, policy X and Y may both have positive net benefits, but be competing for government funding. Policy X which has been analysed using a type C appraisal has on paper higher net benefits than policy Y which was analysed using a type B appraisal, even though the true net benefits associated with policy Y are greater. The failure to fully capture the non-market effects in the type B appraisal, however, may lead decision makers to selecting Policy X. If this occurs, opportunity costs associated with the allocation of government expenditure will arise during the policy selection process.

This potential outcome highlights the importance of policy makers understanding the following when selecting from a portfolio of potential policies:

40 Note that each quadrant in the matrix given in Table 9.1 could be sub-divided further between direct, indirect and secondary effects, thereby increasing the possible levels of policy appraisal, and making the task faced by the decision-maker even more complex.

- what effects have been included in the appraisal of the individual, competing policies;
- whether or not these effects are directly comparable within policies and across policies; and
- how accurate are the cost and benefit estimates.

Assuming that such information is available and clearly understood, the next question concerns what decision criteria should be used at the budgetary level to ensure that opportunity costs are minimised. The decision criteria which are most widely used to determine whether or not an investment decision should go ahead are: the net present value (*NPV*) criterion; the benefit-cost (*B/C*) ratio; and the internal rate of return (*IRR*). The criteria are equally relevant to financial appraisals undertaken by business as part of private sector decision making and economic appraisals undertaken as part of government decision making.

NPV is measured as the present value of benefits (*PVB*) less the present value of costs (*PVC*), where benefit and costs streams are discounted at the appropriate discount rate *r* (for example, the opportunity cost rate of return on the resources employed in the policy). The *B/C* ratio is simply the ratio of the sum of the policy's discounted benefits to the sum of its discounted investment and operating costs (i.e. *PVB/PVC*), again discounting at *r*. The internal rate of return is that rate of discount applied to the benefit and cost streams which equates *PVC* and *PVB*, or which sets *NPV* equal to zero. The *IRR* may alternatively be seen as the rate of return generated on the outstanding capital in each year of the life of a policy.

In the following sections the use of these three leading criteria in two different situations is examined: one in which the acceptability of a single, independent undertaking is to be determined; and one in which multiple policy proposals are to be ranked in order of desirability subject to a input constraint. This second case directly addresses the question of how to minimise opportunity costs when allocating funds across different areas of government expenditure.

9.3.2 Accept/Reject Decisions for Independent Policies

Before a policy is included within the capital budgeting process, and subsequently compared with other policies, it must first pass some form of accept/reject decision criteria for independent policies.

Independent policies are those that are not in any way substitutes for each other. A decision maker is free to choose among such policies, selecting any (or all or none) that will contribute positively to social welfare. In the case of an independent policy, the decision rule in relation to its estimated *NPV* is: any policy for which *NPV* > 0 may be accepted.

If the *NPV* of the policy is negative, then the policy should be rejected. The funds that would have been used for this investment should be left in the bank, returned to (or not collected from) tax-payers, or used to implement a policy whose *NPV* is positive.

In terms of the B/C ratio, a policy may be accepted if its B/C ratio is greater than 1, that is, if its discounted benefits exceed its discounted costs. In many situations, a policy is also socially worthwhile if its *IRR* is greater than some target rate of return, typically *r*. However, it is to be noted that the *IRR* loses uniqueness and more than one rate emerges when there is more than one sign change in the net benefit stream[41].

For completeness, it is worth noting that a characteristic of these three criteria is that each involves what may be an unrealistic assumption concerning the reinvestment of policy proceeds during the life of the policy. Specifically, the *NPV* and *B/C* decision rules assume that proceeds are reinvested at *r*, while the *IRR* assumes reinvestment at the internal rate of return. Clearly, once the possibility of consuming annual benefits is recognised, as for example occurs in the case of non-marketed benefits, then any reinvestment assumption may be inappropriate. Moreover, even in the absence of benefit consumption, there may be no reason to suppose that reinvestment opportunities are available at the *IRR* of the policy in question even if the reinvestment at *r* remains a reasonable assumption.

For *NPV* and *B/C*, a policy is worthwhile if:

$$\sum_{t=0}^{n} \frac{B_t}{(1+r)^t} > \sum_{t=0}^{n} \frac{C_t}{(1+r)^t}$$

Multiplying through by $(1 + r)^n$ gives:

$$\sum_{t=0}^{n} B_t (1+r)^{(n-t)} > \sum_{t=0}^{n} C_t (1+r)^{(n-t)}$$

41 This is because the *IRR* is the solution to a polynomial equation which will have a root for every degree in the equation.

Hence, if the stream of benefits compounded forward to period *n* at the rate *r* is greater than the stream of costs similarly compounded, the policy is acceptable. Equation 2 is formally equivalent to equation 1 and makes explicit the assumption of reinvestment at rate *r*.[42]

9.3.3 Input Constraints: Capital Rationing

Where the decision concerns whether or not to adopt a set of mutually exclusive policy options (e.g. only one can be adopted as the policies are in some way substitutes for each other), the aim is to maximise the net benefits through the choice of option. In this case, welfare is maximised if the decision maker chooses the policy with the highest *NPV*. In such cases, neither the IRR or the B/C ratio can be used to choose between policies as they could result in inconsistent rankings.

The problem faced by a public agency (or a private firm), however, is not simply whether to undertake an individual policy, subject to the opportunity cost of using the required inputs; rather, it is to ascertain the 'best' use of the inputs across the agency as a whole.

42 In order to circumvent the reinvestment problem, it is necessary to have information regarding the proportion of annual benefits reinvested. With this information, the most convenient procedure is to compound benefits forward to the terminal year of the policy using the minimum return required on the investment in respect of the proportion reinvested each year, and the minimum return required as compensation for foregone consumption in respect of the proportion consumed (Mishan (1994), Chapters 37 and 38). Costs are also compounded forward so that benefits and costs may be compared at terminal rather than present value. This procedure, however, has not yet been widely adopted; the information requirement regarding the reinvestment proportion and separate discount rates being somewhat severe.

When faced with unlimited resources (inputs), a public agency can adopt every policy with a positive *NPV*[43]. In doing so, by definition, it will secure higher net benefits than using those inputs in any other way. (Of course, this assumes, albeit unrealistically, that the public agency has only one objective, i.e. to maximise net social benefits.) Public agencies however, are rarely blessed with unlimited inputs. The context is usually always one of 'input constraints', with the most common constraint being that of limited capital funds. As governments tend to ration the amount of money that may be spent on investment during any given period, a public agency needs to be able to rank policies in terms of their desirability, and work down the list until the available investment funds are exhausted.

9.3.4 Single Period Input Constraint

If capital funds are limited and if the agency's objective is to maximise the total present value over the group of proposed policies, this implies that it should seek to maximise the net benefit (receipts minus recurring costs) per unit of constrained input (in this case, investment costs). This can best be achieved by using a measure known as the net benefit investment ratio (NBIR).

The *NBIR* is derived from the ratio of the present value of a policy's benefits minus its recurring costs, to the present value of its investment cost. It is the correct appraisal criterion to use when there is a single period budget constraint because it indicates which of the alternative viable policies will earn the greatest net returns per unit of investment.[44] A policy's *NPV*, on the other hand, can only show the difference between its discounted benefits and discounted costs, over the policy's life.

The policy with the highest *NPV* is not necessarily the one with the highest net returns per unit of investment[45].

43 In such circumstances, there are no opportunity costs issues, as all policies may be implemented. Hence, all policies meriting investment are undertaken and no available welfare gains are forgone.

44 It should be noted that it is not correct to use the *NBIR* to rank policies within the **group** of selected policies; the *NBIR* can only be used to determine which **group** of policies should be selected subject to the budget constraint.

45 The internal rate of return of policies **cannot** be used to rank a group of independent policies whose *IRR* is greater than the target rate. If all of the policies in the group have an *IRR* greater than the target rate, all that can be said in this situation is that all policies should be undertaken. Similarly, the *B/C* ratio **cannot** be used to rank projects if there is a single period budget constraint. As mentioned, the investing agency

will want to rank and select policies so as to maximise net receipts per unit of investment costs, rather than per unit of total costs, as it is investment funds that are short supply.

To illustrate this, consider the portfolio of hypothetical policies listed in Table 9.2. These may be thought of as four different proposals, open to a public agency for, say, 'saving lives'. According to the table, if the *NPV* criterion were adopted, the proposals are ranked D, C, B and A. By contrast, in terms of *NBIR*, the rankings become D, B, A and C. Subject to a capital constraint of 1.5 billion ECU, if the *NPV* criterion were adopted, then 1 billion ECU should be spent on policy D (*NPV* = 0.476 billion ECU) and 0.5 billion ECU on policy C (*NPV* = 0.5/2.0 x 0.396 = 0.099 billion ECU), to yield a total net gain of 0.575 billion ECU[46].

Table 9.2: Ranking Hypothetical Policy Proposals

Proposal	Capital Costs in Year 0 (10^9 ECU)	Net Benefits Per Year (10^9 ECU)	Life of Policy (Years)	NPV in Year 0 (10^9 ECU)	Net Benefit Investment Ratio	NPV Per Unit of Capital
Policy A	1	0.15	15	0.284	1.284	0.284
Policy B	1	0.11	40	0.312	1.312	0.312
Policy C	2	0.6	5	0.396	1.198	0.198
Policy D	1	0.22	10	0.476	1.476	0.476

Use of the *NBIR*, on the other hand, involves adoption of proposals D and B for an overall net gain of 0.632 billion ECU (i.e. 0.5/1.0 x 0.312 + 0.476), the highest attainable. The recommended decision rule in the presence of a budget constraint, therefore, is to estimate the *NBIR* of all policies under consideration, and select those with the highest *NBIR* up to the point where the budget is exhausted[47].

If policies are ranked using their *NBIR*, the decision maker can determine the opportunity cost (or social cost) of not selecting the policies with the highest net benefits per unit of investment. This finding is illustrated further in the discussion below.

9.3.5 Shadow Price of Capital

46 For simplicity, it is assumed that each policy proposal is of a kind that any proportion of it may be undertaken, i.e. each policy is divisible and the returns per unit of expenditure are invariant with the size of the expenditure.

47 It should be noted that the exact same outcome can be achieved by grouping policies (so that the budget constraint is not exceeded) and finding the group with the highest cumulative *NPV*. However, for a large investment programme, it is much less cumbersome to rank the alternative policies by their *NBIR*.

If the budget constraint were to be relaxed slightly, Table 9.2 indicates that for every additional unit of capital (1 billion ECU) invested in policy A, the agency's economic surplus increases by 0.284 billion ECU. Similarly, for every additional unit of capital (1 billion ECU) invested in policy B, the agency's economic surplus increases by 0.312 billion ECU; and for policy C, an additional unit of capital generates an additional 0.198 billion ECU surplus. Therefore, any additional capital would be most efficiently used if the agency invested in policy B. This implies that one unit of capital, with a nominal value of 1 ECU, has at the margin a value of 1.312 ECU to this agency. In other words, the marginal opportunity cost (or 'shadow price') of capital is 1.312 ECU (Sugden & Williams, 1990, p 77)[48].

This 'shadow price', in turn, could be charged for the use of capital by the agency, thereby explicitly taking into account the opportunity cost of capital. To do this, the original capital costs shown in Table 9.2 are simply multiplied by 1.312, and the *NPVs* recalculated. The results are shown in Table 9.3, with the rationale for such shadow pricing also illustrated in Figure 9.1.

Table 9.3: Ranking Policy Proposals Based on the Shadow Price of Capital

Proposal	**Shadow Capital Costs in Year 0 (10^9 ECU)**	**Life of Policy (Years)**	**Notional NPV in Year 0 (10^9 ECU)**
Policy A	1.312	15	-0.028
Policy B	1.312	40	-
Policy C	2.623	5	-0.228
Policy D	1.312	10	0.165

According to Table 9.3, policy proposal D should still be undertaken; it is the only proposal with a positive notional *NPV*. Proposals A and C should not be undertaken, while proposal B, as expected, is at the margin between accept/reject. Policy B has a notional NPV of zero since what is being implied is that "*…it defines the marginal policy in the presence of the capital constraint…*" (Pearce & Nash, 1991, p 152)[49].

The best strategy implied by Table 9.3, is for the agency to undertake policy D, and then invest the remainder of the budget on policy B. Therefore, the ranking by the shadow

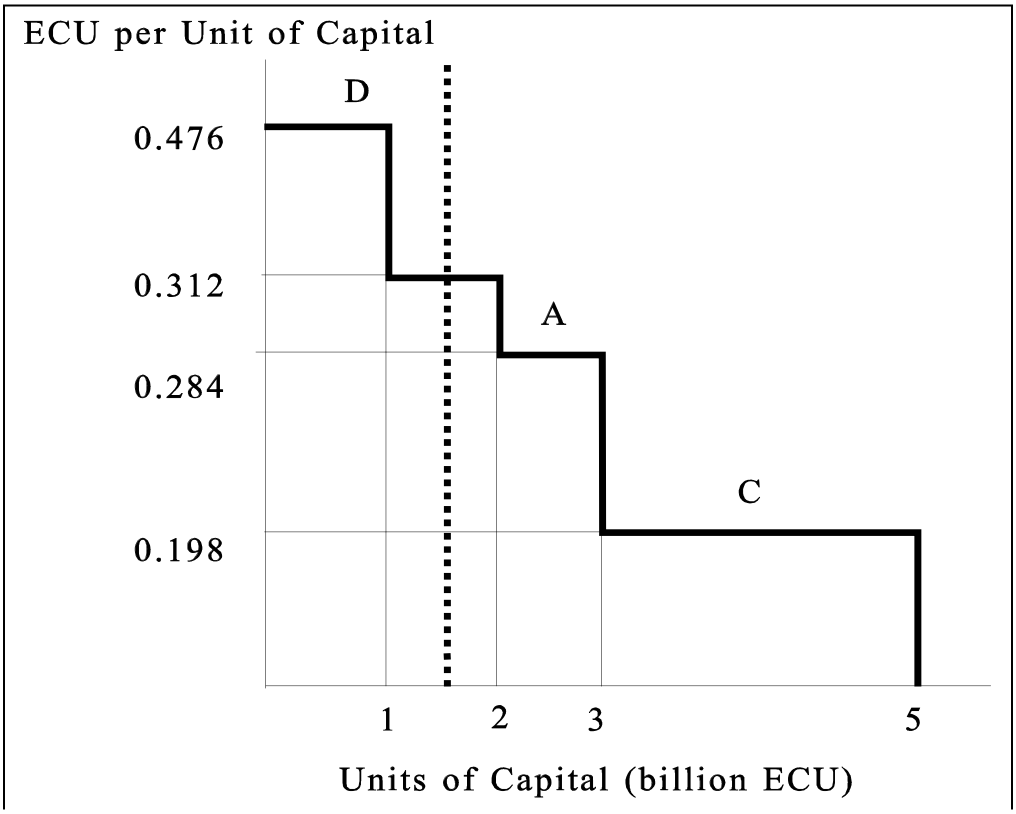
48 Sugden & Williams (1990): **The Principles of Practical Cost-Benefit Analysis**. Oxford: Oxford University Press.

49 Pearce & Nash (1991): **The Social Appraisal of Projects: a Text in Cost-Benefit Analysis**. London: MacMillan Education Limited.

pricing approach is entirely consistent with the ranking recommended above, in terms of the NBIR (net benefit investment ratio).

The 'shadow price of capital', therefore, is the price (or opportunity cost) of the last desirable policy allowed by the budget constraint. Thus, where investment funds are limited, calculation of this 'shadow price' will explicitly account for the opportunity cost of using limited capital in one policy relative to the 'marginal' policy. In the example provided above, the opportunity cost of diverting one unit of capital away from Policy B

Figure 9.1: The Marginal Value of Capital



is given in the fourth column. For example, increasing expenditure on policy A by one unit will involve opportunity costs of 0.028 billion ECU.

Overall, the opportunity costs of the selected investment programme (i.e. policy D and B) are zero; basically, the decision-maker cannot reallocate capital in any way that would increase net welfare. As indicated above, opportunity cost only arise when funds are diverted away from the set of policies that maximised welfare, subject to the budget constraint.

It is interesting to note that this concept is applied in practice as part of public sector decision making. For example, in CBAs carried out by the World Bank, capital funds (in both the public and private sector) have shadow prices in excess of one so as to reflect the scarcity of capital to the 'borrowers'. So, where the shadow price of capital is calculated at 1.2, a project involving investment in a pollution control system with a market value of 1 million ECU would be costed at 1.2 million ECU in order to reflect the true opportunity cost of the funds used to purchase the equipment. A similar approach could easily be adopted within the EC to account for opportunity costs of capital investment in alternative policy areas.

9.4 Cost-Effectiveness and Opportunity Costs

9.4.1 The Need for an Alternative Measure

The use of the capital rationing criterion or the shadow price of capital in order to determine the opportunity costs of adopting different policies assumes that all the costs and benefits have been included in the individual policy appraisals and that the levels of accuracy associated with those estimates are comparable.

Clearly, these assumptions are not going to hold in many comparisons of alternative policy measures. As a result, it is useful to consider what alternative approaches exist for ensuring that resources are used in an efficient manner. Related to the concept of the opportunity costs at a cross-policy level is that of the cost-effectiveness of policies in delivering specific end goals. This latter concept involves consideration of the comparative costs of alternative policy measures which would achieve an equivalent level of protection for the environment or human health.

Such analyses can be extremely useful in answering the following questions:

- can we achieve similar health or environmental benefits at less cost through a different set of policies? or

- can we achieve greater benefits for the same cost through a different set of policies?

9.4.2 How Can We Compare Effectiveness?

There are two concepts of cost-effectiveness that are relevant to integrating the types of issues raised above into a comparison of the performance of alternative policy measures. The first of these is the 'cost per life-year saved' or the 'cost per life saved', while the other relates to the per unit cost of achieving a target level of environmental protection (e.g. cost per hectare of wetland). The use of these concepts leads to a rephrasing of questions concerning opportunity costs to the following:

- are we spending too much on this policy to save a statistical life year? Can we spend less on an alternative policy to save the same number of lives? and/or

- are we spending too much on this policy to protect the environment? Can we spend less on an alternative policy to gain the same level of environmental protection?

In general, the cost-effectiveness of policies is likely to decrease as they tend towards achievement of 'zero risk'. In other words, as the level of risk posed by an activity or substance decreases, the costs of further reduction are likely to rise. This point has been illustrated by research undertaken in the US examining the cost-effectiveness of worker and general public health and safety legislation [e.g. Tengs *et al* (1995), Tengs & Graham

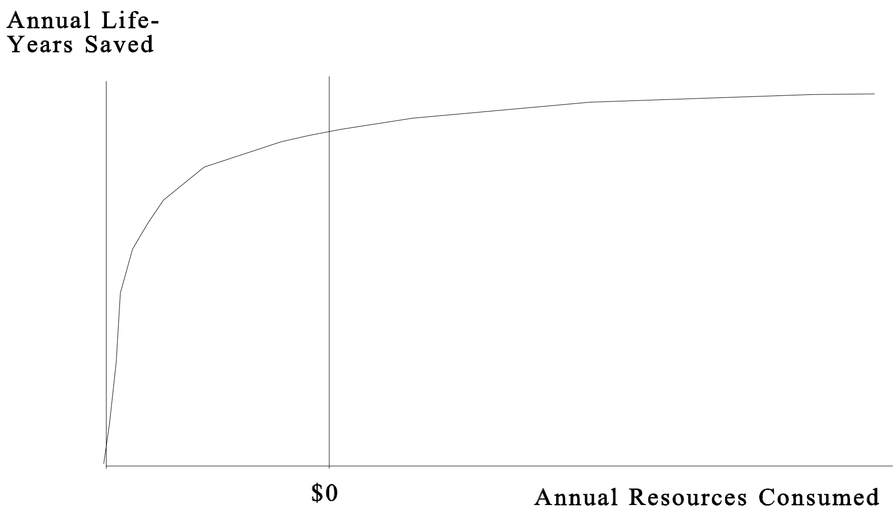
(1996), Viscusi (1996) and Hahn (1996)].

For example, Tengs *et al* (1995) compared '500 life-saving interventions' in terms of 'the cost per life-year saved' and the median values found for different types of interventions were:

- medical intervention costs: $19,000/life-year;
- injury reduction: $48,000/life-year; and
- toxin control: $2,800,000/life-year.

The considerable difference in the median value for toxin control as compared to either medical interventions or injury reduction is of particular note. Indeed, toxin control costs per life-year saved ranged from zero for controls such as a ban on the use of amitraz pesticide on apples and SO_2 controls by installation of capacity to desulphurise residual fuel oil, to $20bn for benzene emission control at rubber tyre manufacturing plants, to $99bn for a chloroform emissions standard at private wells affected by 48 pulp mills.

Looking at the relationship between total annual life-years saved as a function of total annual resources produces a function as illustrated in Figure 9.2 (adapted from Tengs & Graham, 1996). This Figure suggests that many life years can be saved at zero or negative cost, but once past the '$0' threshold, costs increase substantially for very few life-years gained.



The research undertaken by Viscusi (1996) reaches somewhat different conclusions, however. In this case, the analysis calculated the 'cost per life saved' associated with each of the policy measures, with the a selection of the results given in Table 9.4.

Table 9.4: The Costs of Various Risk-Reducing Regulations per Life Saved in the US

Regulation	**Initial Annual Risk**	**Annual Lives Saved**	**Cost per Life Saved (millions, $1990)**
Regulations passing a cost-benefit test			
Passive restraints/belts	9.1 in 10^5	1850	0.39
Seat cushion flammability	1.6 in 10^7	37	0.77
Hazard communication	4.0 in 10^5	200	2.32
Benzene/fugitive emissions	2.1 in 10^5	0.31	3.61
Regulations failing a cost-benefit test			
Benzene	8.8 in 10^4	3.8	22.03
Ethylene oxide	4.4 in 10^5	2.8	32.97
Asbestos	6.7 in 10^5	74.7	115.03
Formaldehyde	6.8 in 10^7	0.01	92741.89

As can be seen from this table, those regulations which passed a cost-benefit test have implied costs per life saved of $3.6 million or less. Of the regulations that fail a cost-benefit test, the cost per life saved is very high in comparison. This analysis further shows that it is not always correct to assume that only the relatively low risk cases will be prohibitively expensive. For example, the annual level of risk posed by seat cushion flammability is lower than that associated with formaldehyde (1.6 in 10^7 and 6.8 in 10^7 respectively), yet the higher number of lives saved per year and the lower costs of implementation for the seat cushion regulation increases its relative cost-effectiveness.

9.4.3 Comparative Evaluation Using Cost-Effectiveness

The findings of research on the cost-effectiveness of previous health and safety policies has led to some researchers suggesting that rules of thumb are set as part of environmental policy making. For example, it has been suggested that decision makers (Tengs & Graham, 1996):

> *"invest in all interventions costing less than some threshold (for example, $5 million per life saved) and in none of the interventions costing more...".*

Although such a rule of thumb is brutally simple, it reflects the view that efforts should be made to ensure that environmental and health and safety improvements are achieved in the most cost-effective manner possible. What a simple rule such as this fails to recognise, as do some of the analyses presented above, is that many environmental and health and safety regulations will provide more than just one type of benefit. At a simple level, a number of health benefits (related to reduced injuries or other morbidity effects) in addition to reduced mortality may result from a measure. Or, as is the case with the National Air Quality Strategy (NAQS) case study, a wide range of both health and environmental benefits may arise. The use of a single attribute measure of effectiveness, therefore, provides a misleading indication of the cost versus benefit implications of the policy.

However, measures such as the 'cost per life saved' or 'cost per life year saved' do have a role in enabling comparisons of environmental policies which have the achievement of health benefits as their driving aim (such as spending on a health service, or a road safety campaign and so on). Clearly the most striking difference between CBA and CEA, is that the latter does not require that 'benefits' be expressed in monetary units. Given some of the concerns identified in this study as regards the accuracy of 'benefit valuation' and thus the difficulty of developing proper shadow prices for environmental and health impacts, the ability to side-step these concerns could be looked at as being a 'plus' for CEA-type analyses.

However, CEA alone cannot provide an indication as to the appropriate size of, for example, a roads safety programme; nor will it resolve the problem of policy selection when the policies yield a mixture of differing benefits, as may result from different air quality programmes. In such cases, it is necessary to express the varying benefits in a common unit to facilitate direct comparison, with money being the logical choice. This brings us back to CBA.

9.5 Opportunity Costs in EU Policy Making

9.5.1 Opportunity Costs in the Appraisal of Individual Policies

What are the implications of the discussion presented above for policy making within the

EU? In order to examine this question, we need to take a stepped approach, starting with the need for appraisals to be comprehensive in their coverage of costs and benefits and for these to be measured in terms of their proper shadow prices.

The three case studies presented in Sections 6, 7 and 8 of this report illustrate the difficulties which arise in the preparation of CBAs of environmental regulations. From these three cases, we can draw four key conclusions:

- in none of the case studies were all potential direct, indirect and secondary costs and benefits quantified for incorporation into a cost-benefit accounting framework;

- for some impacts, this failure was due to a lack of adequate scientific data, with this posing problems for all three cases. The lack of appropriate valuation data was also an issue, as was uncertainty as to the manner in which industry would respond. Time and resource constraints were obviously factors affecting the appraisals in this regard;

- the robustness and hence final conclusions of each appraisal is affected by the failure and/or inability to include all potential effects, although this varies between the case studies; and

- the scope required by the CBA, in terms of the relevance of considering employment, related market and price effects also varies across the case studies.

The air quality case study concerns a policy which has as its main objective the reduction of pollution related health effects. These effects are valued within the CBA and are considered to be the most important direct effects. A key question with regard to this case study, however, concerns the valuation of mortality effects. Two values were used in the appraisal: a value of statistical life (VSL) figure of MECU 3.1; and a value of a life year lost (VLYL) figure of ECU 98,000. Obviously given the differences in magnitude of these two figures, one would expect them to yield widely varying results in terms of the value of the benefits associated with reduced mortalities. The choice of the appropriate value and unit of measure in this case is highly important to ensuring that the appraisal results reflect true opportunity costs (see also the discussion in Section 4.6).

It was also possible to examine the cost-effectiveness of the proposed reductions in SO_2 in terms of the implied value per life saved. This was calculated at MECU 0.144 per acute mortality avoided for the basic policy considered in the analysis. It is important to remember that this figure would be reduced if the benefits associated with reductions in chronic mortality and morbidity effects and reduced impacts on crop yields, buildings and materials and ecology were including in the calculation. Even with the inclusion of these other benefits, however, this figure compares favourably to the VLYL in the study of MECU 0.098.

The most significant omissions from the air quality analysis were the benefits from secondary emission savings and reductions in ecological damages, and the net effect on related markets, including employment in those markets. The latter two may be particularly important and the failure to include them indicates that the analysis is not providing an indication of the true net opportunity costs arising from the proposed policy. In this case, inclusion of such effects could result in a switch between different policy options.

The original waste management study would also appear to have omitted a number of important direct, indirect and secondary impacts. Valuation here focused on the costs of adopting the different strategies and transport related effects. It fails to examine a range of other environmental and resource effects, employment related effects, wider market effects and thus prices, and potentially effects on trade and productivity. The inability to value some effects was recognised and noted in the study report, however, a large number of the potentially significant effects were not identified. Of most concern, may be the failure to consider employment and wider effects on related markets, particularly with relation to recycling and reuse options. The inclusion of some of the above effects may have an impact on the total net social benefits associated with the different waste management strategies and could alter their relative ranking. Again, this highlights the potential for opportunity costs to arise in the selection of the end management strategy.

The range of impacts which were valued in the SCCP case study is smaller still, with only those related to increased input and production costs estimated. Failure to value the environmental benefits (and of lesser concern in this case, the health benefits) associated with the two proposed regulatory measures considered stemmed from a lack of data on the actual reduction in risk which would result from their adoption. The absence of such data made valuation impossible. Thus, the analysis in this case is more partial than in the other two case studies, although in this case the risk assessment procedures preceding the appraisal determine (politically) that the risks are unacceptable. Hence, although the appraisal should address the economic costs of adopting risk reduction, it is almost reduced to a cost-effectiveness analysis. The phrase ‘almost’ is relevant as it should be recognised that the different measures lead to variations in environmental and health benefits, in addition to different cost streams.

9.5.2 Opportunity Costs in Public Expenditure

The above findings suggest that more can be done within environmental policy appraisals to ensure that they better reflect the true economic costs of adopting a proposed measure. The implications of these findings for the allocation of government expenditure must also be considered, however.

For example, it is estimated that approximately BECU 31 is spent annually within the EU on pollution abatement and control, while BECU 144 is spent on public services (assumed here to relate to roads), and BECU 240 is spent on health care. If we assume that policy in each of these areas is focused solely on saving lives, and that the average expenditure per policy is at the values quoted in the second column of Table 9.5, then these current levels of expenditure would imply the following:

- the BECU 31 on pollution control would be equivalent to 10 000 lives saved per annum at the margin;

- the BECU 144 on public services would be equivalent 144 000 lives saved per annum at the margin (using the road safety value); and

- the BECU 240 on health care would be equivalent to 4.6 million lives saved at the margin.

However, not all measures to save lives within a given policy area will be as costly as this in the first two cases, nor will they be as inexpensive in the case of health care measures. This is, therefore, a false analysis.

Table 9.5: Interventions and Lives Saved Under a Budget Constraint

Type of Policy/ Intervention	**Value Assumed**	**Basis for Valuation**	**Number of Lives Saved Given MECU 100 Budget**	**Budget Required to Save 100 Lives (MECU)**
Air Quality	MECU 3.1	Review of WTP across different risk contexts	32.25	310
Roads Safety	MECU 1.0	WTP but also other direct and indirect costs	100	100
Breast Cancer Screening	ECU 52,700	Cost per life saved based on UK health care costs	1,900	5.3
Policy Area	**Values Assumed**	**Basis for Valuation**	**Number of Life Years Saved for MECU 1 Budget**	**Budget Required to Save 100 Life Years (MECU)**
Air Quality	ECU 98,000	Based on VSL of MECU 3.1	10.2	9.8
Heart Transplant	ECU 7,750	Cost per life year saved based on health care costs	130	0.8
Doctor’s Advice on Smoking	ECU 258	Cost per life year saved based on health care costs	3 875	0.03

Of more interest is consideration of what the existing WTP figures imply in terms of the cost-effectiveness of measures across policy areas. Table 9.5 provides a comparison of the number of lives saved and the number of life years saved which could be achieved across different areas of government expenditure assuming a fixed budget using relevant WTP figures and health care cost figures. It also indicates the level of budget which might be required under each of these policy area in order to 'save' 100 lives.

In using this table, it is important to note that the estimated number of lives saved or life years saved differ in nature. Those estimates related to air quality and roads safety should effectively be considered minimum numbers as there will be many policies which are able to provide reductions in the number of lives lost or life years lost at costs below the WTP values quoted here. The estimates based on health care costs instead relate directly to the actual costs of the health care interventions themselves and thus reflect the average cost per life saved or life year extended.

The reason for developing this table is that it clearly illustrates how opportunity costs at a cross policy level may become an issue. First, we must assume that a government's sole aim is to target expenditure so as to save lives and that a range of different measures across different policy areas exist for achieving this. An appraisal carried out for a proposed air quality programme, or for that matter a road safety programme, using the values given in the table may be able to justify that a policy requiring government expenditure go ahead even though there are alternative competing policy areas (in this case health care) which would provide greater benefits in terms of the number of lives saved.

Taking a real example, the cost-effectiveness of the proposed reductions in SO_2 in terms of the implied value per life saved were calculated in the air quality case study presented in Section 6. The resulting figure was a cost per acute mortality avoided of MECU 0.144 for the basic policy considered in the analysis. If this figure is compared on a one for one basis with the health care cost estimates presented in Table 9.5, it can be seen that there are likely to be a range of other measures which would provide for a greater number of lives saved at a lower cost. As noted in Section 9.4, however, making such simplistic cross-policy comparisons can be dangerous. For example, the figure of MECU 0.144 relates only to acute mortality and fails to include benefits associated with reductions in chronic mortality, morbidity effects, damages to crops, building and materials and ecological effects.

Care is therefore required when undertaking cross-policy comparisons to ensure that the appraisals have been prepared on a consistent basis, as has been stressed earlier in this section. It must be remembered that:

- decision makers are not only concerned with a single goal, that of saving lives, and to the detriment of all other goals such as environmental protection, social security, employment, defence, etc.;
- policies in different arenas are aimed at dealing with different types of risks, different types of risk generating activities and different affected populations; and

- individuals' willingness to pay may vary significantly over different policies and policy options.

Nevertheless, the Table 9.5 illustrates the need for greater use of the type of decision rule set out in Section 9.3 above to ensure that across all relevant areas of government policy, the opportunity costs of meeting government objectives are minimised.

10. SUMMARY AND CONCLUSIONS

10.1 Overview

The preceding sections have reviewed the key principles and concepts, and many of the practical issues, which arise in the application of economic appraisal techniques to environmental policy appraisal. Given the wide range of material covered, it is important that the main points arising from the study are pulled together and summarised at this stage.

This section has been drafted with this in mind:

- it starts by setting out the differences between what is covered by the different forms of economic appraisal;
- the key issues affecting the valuation of the costs and benefits of environmental policies are then considered, including both theoretical and practical issues; and
- the importance of considering opportunity costs at both the individual policy and cross-policy levels is then re-stated.

It is also recognised, however, that the findings of this work provides the basis for setting out simple guidelines for use by policy makers within the EC. These guidelines set out the types of questions which policy makers should ask when commissioning, interpreting and using the results of economic appraisals.

Finally, we provide some recommendations on further research to both build upon the work undertaken within this project and to improve and further appraisal practice within the environmental field.

10.2 The Economic Appraisal Methods

The ideal policy appraisal is one that gives a true indication of all the implications of adopting a particular policy. To this end, it is necessary to identify all of the parties that would be affected by the policy (where this includes consumers, producers and other actors within the wider economy), how they would be affected (whether this is directly or indirectly), and to then determine the magnitude and significance of policy's effects on their welfare.

Cost-effectiveness analysis (CEA) and cost-benefit analysis (CBA) are the two approaches used most frequently in the appraisal of environmental policies. The advantage of these two techniques is that they provide a direct indication of the resource costs involved in implementing a policy; CBA has a further advantage in that it provides an indication of the extent to which the benefits arising from a policy measure would outweigh the costs. It, therefore, yields data on the net resource implications of a proposed policy.

In general, the context of 'conventional'[50] CBA or CEA is that of 'partial equilibrium' analysis. That is, the analysis focuses on the direct effects (gains and losses) of a policy in a single, or a few related, markets. So, for example, pollution abatement measures, their costs and the consequences of their adoption are examined in great technical detail, with aggregation to a national level requiring the adoption of a range of generalising assumptions. The assessment of a policy's impacts in this manner is referred to as a 'bottom-up' approach. This type of analysis is appropriate when the effects arising from a policy are not significant enough to affect other market sectors or the economy more generally. In such cases, a conventional CBA or CEA can provide a reasonable indication of an environmental policy's effect on the target industry and the environment.

Current practice excludes the assessment of some potentially important direct effects. For example, most analyses exclude the valuation of losses in competitiveness, changes in employment patterns and changes in secondary emissions. Analysis of the former is usually undertaken outside the analysis, if at all. While direct employment effects are often excluded on the basis of theoretical assumptions concerning the existence of a flexible and mobile labour market which results in any 'surplus' labour moving elsewhere or being taken up by excess production capacity in another sector. As such assumptions can rarely be said to hold in reality, extending CBA through the use of other analytical approaches may be important.

There are, however, important exceptions where the assumptions underlying the adoption of a conventional CBA will not hold. Some policies will have significant (non-marginal) impacts on an industry sector, which is itself highly integrated with other industry sectors within the economy. In such cases, **direct effects** on the target sector could lead to price changes and thus **indirect effects** on other sectors' activities and possibly to **secondary effects** on the economy more generally[51].

Where the prices of non-target goods and services are changed as a result of the project or programme, then 'conventional' CBA as the basis for the appraisal becomes inappropriate. In such cases, where the implications of indirect and secondary effects is likely to be important, the appraisal can turn to the use of what are referred to here as

[50] We use the term 'conventional' to emphasise that most CEAs and CBAs are typically exercises in 'partial equilibrium' analysis.

[51] For the purposes of this study **direct effects** have been defined as 'those effects and impacts that can be primarily attributed (i.e. of the first order) to a policy/project'; **indirect effects** as 'changes in output in related sectors of the economy through backward and forward production linkages (i.e. second- or third-round responses)'; and **secondary effects** arising when 'policy expenditures and surpluses generate demands for commodities that, in turn, lead to a secondary increase in output'.

'top-down' approaches, e.g. input-output analysis and computable general equilibrium modelling. These approaches recognise that the implementation of regulations affects the manner in which individual companies act as both 'buyers' and 'sellers', and thus their subsequent interactions with other companies within the same and other sectors. Because 'top-down' approaches explicitly model the interactions between markets and thus account for the effects that a change in one market has on another, they give a more accurate assessment of the overall impact of a non-marginal policy, than would be obtained through 'conventional' CBA. As these methods focus on the sectoral level (rather than the individual company level), however, they essentially sacrifice technical detail to provide greater spatial coverage.

Input-output models rely on the use of fixed coefficients to represent the input per unit output linkages between different sectors of the economy. By examining changes in the output or costs faced by one sector, it is possible to determine the impacts on levels of production faced by other sectors. Such fixed coefficients can also be developed to link changes in production to changes in employment and to environmental quality. However, the use of fixed coefficients to represent environmental quality effects has been criticised owing to difficulties in accounting for residuals and threshold effects in such models. There are also several other theoretical shortcomings associated with input-output models.

The other main form of top-down analysis is that of general equilibrium modelling. In contrast to input-output analysis, these models rely on a series of systems equations to describe supply and demand linkages throughout the economy. Because they consider both supply and demand, they are capable of dealing with longer time-horizons and of providing a greater level of information on economic structure, product mix and future growth. Their use in environmental policy appraisal has been criticised, however, because they make assumptions about what should happen (rather than what does) and tend to assume that there is no unemployment at equilibrium in the economy.

In addition to the theoretical shortcomings of these two 'top-down' approaches, their inherent complexity means that the amount of time and effort required to collect the basic data, and build a suitable model, is often prohibitive.

Table 10.1 overleaf sets out the different forms of analytical technique covered here and the main policy effects which are generally included within them. What is clear from the above discussion and from this table, is that there is overlap between what both the bottom-up (or partial equilibrium) and top-down (or general equilibrium) models can provide, yet that there are also significant differences.

10.3 The Valuation of Costs and Benefits

In practice, CBA is the most commonly used appraisal methodology, with most analyses focusing on the direct impacts of the policy itself.[52] As a result, Sections 2 to 4 focused

[52] Henceforth, unless otherwise stated, when we refer to CBA, we automatically mean 'conventional' CBA.

on the application of CBA and the methods which are used within it to value direct, indirect and secondary impacts. This included discussion of approaches which should be adopted for estimating the value of impacts on producers (industry) and consumers, administrative costs, employment, the environment and human health.

Table 10.1: Analytical Approaches and Inclusion of Direct, Indirect and Secondary Effects				
Effect	**Analytical Approach**			
	CBA	**CEA**	**I-O**	**GE**
Direct and indirect impacts on industry	✓	✓	✓	✓
Increases in costs of end products to consumers	possibly		✓	✓
Change in costs of administration	✓	✓	✓	✓
Changes within and between sectors	possibly[1]		✓	✓
Economy-wide price impacts			✓	✓
Distributional effects	possibly[2]		✓	✓
Employment	possibly[3]		✓	✓
Change in relative competitiveness	possibly[4]	possibly[4]	✓	✓
Environmental impacts (use and non-use values)	✓	as target	possibly[5]	
Improvement/deterioration in human health	✓	as target	possibly[5]	

1 It is possible within a partial equilibrium framework to consider the impacts of a few linked markets.
2 This can be accomplished with the use of distributional weights.
3 Although this would require the use of 'Keynesian' type multipliers.
4 This is possible, however, it would tend to be undertaken exogenously to the CBA and CEA.
5 As indicated in the discussion, environmental quality matrices can be built into the analysis.

The key points which arise from the discussion provided in these sections are as follows:

- Impacts on industry may include both changes in capital and operating costs. The impact of changes in costs upon the competitiveness of an individual company, industry sector or national economy will depend on the degree to which the costs can be passed on to customers and are imposed equally across all competing companies. Regardless of who bears the costs, a policy will only impact on the competitive position of a company if it is not imposed equally across all companies that compete in the same market. Given that policy is harmonised within the EU, the concern then relates to whether an EU policy impacts on an industry's ability to compete at a global level.

- In addition to consideration of changes in direct costs, consideration should also be given to any indirect costs (or benefits) which may arise, for example, from the adoption of substitutes. These may include the need for process changes, impacts on the quality of end-products, or impacts on health and the environment where the substitutes themselves pose risks.

- Although many environmental policies may have only negligible impacts on employment, there will be cases where the magnitude of the costs facing a particular sector or sectors are significant enough to create a moderate net change in employment. It is not possible to generalise about the potential magnitude of such changes as they will be policy and media specific. Where net effects are judged to be significant, they may be included in the analysis, providing it is possible to estimate the change in the number of jobs. Such appraisals should consider not only the net change in income (i.e. the net salary minus any social security benefits) associated with the loss (or creation) of a job, but also the wider loss in social welfare arising from increases in unemployment.

- A range of techniques exist for the valuation of environmental costs and benefits, with the applicability of these varying depending on the nature of the goods and services to be valued. The results of the revealed preference and surrogate market techniques are more likely to be accepted than those derived through the use of hypothetical market (or survey-based) techniques, even though the latter are more flexible and hence applicable to a wider set of impacts. For example, only survey-based methods are capable of assessing non-use values. Although a wide array of valuations exist, the ability to identify reliable transfer values is constrained in many areas, and in particular with regard to ecological functions.

- The valuation of reductions in fatalities poses particular problems at this point in time owing to a lack of studies specific to the types of issues associated with many environmental problems. Most estimates of the value of a statistical life (VSL) which are currently in use have been developed in the context of road safety (although there is a range of worker and public safety studies). In contrast to road safety (which is valuing a reduction in the risk of accidental death), the impacts of concern for environmental policies often relate to premature death, latent effects and chronic effects. In addition, there are differences in the populations affected in terms of both health and age. As a result of such differences, the transferability of baseline VSL values developed in the road safety context to environmental policies is questionable. Researchers are, therefore, suggesting that adjustment factors be applied to road safety VSLs to take into account the differences in the context and nature of environmental quality risks. Others are promoting the development of measures based on the value of a life year lost which, although based on a VSL at present, would relate more obviously to the question of premature death. In any event, further research in this area is essential as the reliability of either approach continues to be questioned (e.g. Department of Health, 1999).

- Similarly, further work is required on the valuation of morbidity effects, with only some of the end-points of concern having been addressed and many of the values being derived in a manner which makes their transfer to environmental policy making potentially unreliable.

10.4 The Opportunity Costs Issue

10.4.1 Opportunity Costs at the Individual Policy Level

The welfare economics principles, which provide the foundation for CBA, require that the analysis should include all impacts on each economic actor (consumers, producers and the wider society) affected by a policy, and that the effects of the policy on their welfare should be valued in monetary terms as it would be valued by them, in order to reflect real resource losses and gains. Policy appraisal conducted in accordance with this principle will ensure that each policy is assessed as regards its net contribution to society's welfare; the full net opportunity costs arising from the adoption of a given measure will be taken into account. In such cases, the total value of a policy's output (as measured by society's willingness-to-pay) and the cost of all resources used by the policy (as measured by their opportunity cost) will be taken into account.

Two questions are, therefore, paramount as to whether a policy appraisal conforms to the above principle:

Are all the relevant effects included in the appraisal (the issue of completeness)?

Has each effect been valued correctly (the issue of accuracy)?

These questions are considered below. If we are reasonably confident that the answer to these questions is "yes", then any independent policy with a positive net present value (NPV) is worthy of implementation; the benefits generated by the policy are sufficient to offset the opportunity cost of all resources used by the policy[53]. However, just because a particular policy yields positive net benefits does not, on its own, indicate that the policy should be adopted. It is a necessary, but not sufficient condition, as decision makers will also want to take into account a range of other factors.

If we are not confident that the answer to the above two questions is "yes", then the appraisal may fail to reflect the true opportunity costs associated with the proposed policy, and a misallocation of resources may occur. In other words, expenditure on the proposed measure may not represent the best use of resources, with either the benefits not justifying the costs or their being an alternative which yields higher returns.

In practice, undertaking a CBA which captures the full opportunity costs of adopting a given policy is not straightforward. These analyses are usually broken down into 'what should be included' and 'what can be included'. In an ideal world, all costs and benefits will be quantified in monetary terms and there will be no value uncertainty associated with the monetary estimates. Table 10.2 sets out a list of what 'shoulds' and 'cans' within an environmental policy context.

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[53] This is not to say that the resource used by the policy cannot be put to better use, for example, by using them in another policy which generates higher benefits; in terms of the assessment of whether or not an independent policy is worthwhile in its own right, this is not an issue as long as NPV is positive.

Table 10.2: The Difference in the Reality of Undertaking CBA	
What Should be Included in an Appraisal	**What Usually is Included in an Appraisal**
Consideration of alternative policy options Direct and indirect costs and benefits to industry Environmental costs and benefits Human health costs and benefits Impacts on all related markets (as appropriate) Impacts upon employment Economy-wide price changes (as appropriate) Multiplier effects (as appropriate) Impacts upon the distribution of income	Direct and (sometimes) indirect costs and benefits to industry Environmental impacts (subject to uncertainty) Human health impacts (subject to uncertainty) Impacts on some related markets

As discussed above, practice often falls short of theory with regard to environmental and health benefits; however, many appraisals also fail to go beyond the direct costs and benefits, even when a policy has significant indirect and secondary effects. This statement is supported by our conclusions from the case study work undertaken as part of this study:

- in none of the case studies were all potential direct, indirect and secondary costs and benefits quantified for incorporation into a cost-benefit accounting framework;
- for some impacts, this failure was due to a lack of adequate scientific data, with this posing problems for all three cases. The lack of appropriate valuation data was also an issue, as was uncertainty as to the manner in which industry would respond. Time and resource constraints were also obviously factors here;
- the robustness and hence final conclusions of each appraisal is affected by the failure and/or inability to include all potential effects, although this varies between the case studies; and
- the scope required by the CBA, in terms of the relevance of considering employment, related market and price effects also varies across the case studies.

Across the case studies, the most significant issues to arise which we identified were the failure to include secondary effects (e.g. emission savings), the failure to place monetary values on ecological damages, the failure to include net employment effects in the targeted and (where appropriate) related markets, and the failure to consider net effects on related markets (where judged to be significant). Inclusion of such effects in the appraisals may have led to differences in the rankings assigned to the various policy options considered within each appraisal.

10.4.2 Questions over Appropriateness and Accuracy

The failure to consider all policy effects, e.g. impacts on secondary markets and employment, are mainly issues of scope, in that either the policy maker when commissioning the study, or the analyst when undertaking it, must set some boundaries on the appraisal to be carried out. However, the issues are somewhat different with regard to the valuation of environmental and human health effects, which is also the subject of some controversy.

As discussed, a number of techniques are used to estimate the magnitude of such effects in monetary terms, but the use of hypothetical market techniques such as contingent valuation remains a source of debate. This is particularly true in the valuation of non-use values (relating to ecological and conservation functions of the environment) and morbidity and mortality. These issues can then be exacerbated through the transfer of benefit estimates from one decision context to another - the issue surrounding the use of a roads safety VSL in an environmental policy context. In particular, in a context where the policy aims to reduce the risk of health effects that are chronic, latent or will be felt into the future, and where the loss of life expectancy is considerably shorter than it is in cases of accidental death.

In general, questions about the appropriateness of monetary valuation within an environmental policy context are of two types:

- questions about the ethics of placing a value on the environment and health; and
- questions about the accuracy of the valuation techniques.

With regard to the ethics of monetary valuation, such criticisms ignore the reality that choices sometimes **must** be made that involve trade-offs between non-marketed and marketed goods. Furthermore, such trade-offs are implicitly made on a continuous basis by public agencies, whether or not they are actually expressed in monetary units. The counter-argument, therefore, is that it is better to make the trade-offs explicit and to allow direct incorporation of all impacts into the analysis than to hope that non-marketed goods are given sufficient weight by decision makers.

The second criticism, which questions the accuracy of the valuation techniques, is more justified. All the valuation techniques have shortcomings of one kind or another, with the severity of these shortcomings varying between techniques. To address such concerns, they are subject to continuous refinement. As the approaches continue to evolve, however, it is important that the shortcomings of each are made clear so that decision makers are aware of the caveats which should be associated with the results. There is little justification, however, for omitting non-marketed goods from the cost-benefit equation on the basis of accuracy alone. This may result in the decision maker implicitly assigning them a value of zero, which in turn will result in a mis-allocation from an opportunity costs perspective.

Although there are concerns over the accuracy of environmental and health valuations, it must be recognised that the accuracy of estimates of direct market related effects, such as the costs faced by industry in implementing a policy, can be questioned. Such estimates may also be out by an order of magnitude owing to technological changes, and industry making different assumptions in reality than those assumed in the appraisal.

The decision maker, therefore, needs information on the degree of certainty attached to the cost and benefit estimates, and must ultimately decide the weight to be given to each item in the cost-benefit equation. He must have an appreciation of the valuation methods, and of the sensitivity of the end results to changes in assumptions. For this to take place, the analyst must present the results of the analysis in a transparent format and work closely with the decision maker.

10.4.3 Other Constraints on Completeness

However, it must be recognised that to include every impact within an appraisal poses serious financial and elapsed time requirements on a study. Given that CBAs are commissioned by governments or bodies whose research funds (especially public) are limited, to expect a fully comprehensive CBA is unrealistic. Three restrictions may face both the analyst and decision maker in this regard:

- lack of available data;
- lack of human or financial resources; and
- lack of enough time to undertake the research required.

It is usually the case that appraisals are conducted on fairly short time-scales (such as three to six months) which do not allow sufficient elapsed time for the collection of the data required to estimate indirect and secondary impacts. Even a 'standard' CBA which is focused on direct and key indirect impacts faces problems with time scale and the collation of data. A full CBA in every case is therefore an unlikely possibility. The analyses should, therefore, be treated as containing elements of uncertainty and omissions in every case.
In all cases though, the aim should always be to include quantitative measures of the most significant effects and ensure that the most significant unquantified effects are, at a minimum, identified and recognised.

10.5 Opportunity Cost at the Cross-Policy Level

10.5.1 Opportunity Costs and Decision Criteria

In a world with unlimited resources, Governments would implement most policies that have a positive net social value (assuming that the primary objective was to increase social welfare). The real world limitations on available funds, however, prevents this from being possible in practice. In order to ensure that the opportunity costs of choosing one policy over another are minimised, the aim should be to select the package of policies which yields the highest possible net social value, subject to available resources, or alternatively to maximise benefits per unit of expenditure. Both approaches will indicate the opportunity cost of selecting one course of action over another, where opportunity cost is measured in terms of foregone net benefits (e.g. the value of the increased number of lives that could have been saved had an alternative policy been adopted).

Where a range of policies compete for funding, a further decision criterion can be called upon to determine that policy (or set of policies) which will provide the greatest net returns subject to the budget constraint. This is the net benefit investment ratio (NBIR) criterion, which can be used within a budgeting exercise to determine the implications in social welfare terms of selecting one policy over another. It provides a direct indication at the cross-policy level of the opportunity cost of not selecting those policies with the highest net benefits per unit of investment. Through this criterion, the decision maker can explicitly account for the opportunity cost of using limited capital in one policy relative to another.

10.5.2 Policy Cost-Effectiveness

However, environmental legislation is very rarely considered as one potential measure within a wider package of alternative measures, especially when the objectives of such legislation span other policy areas such as social security and employment. The often fragmented and competing structure of government departments and associated policy responsibilities makes this impossible. In many countries, each government department will be given a certain amount of funding to implement its own policies (usually on a yearly basis) or will be given policy targets to be met through regulation of the private sector. There may be a vested interest, therefore, in pushing through policies within their own remit. This effect can be exacerbated when the remits of individual departments overlap, resulting in a potential lack of cooperation and coordination brought about by departmental single-mindedness.

The air pollution case study provides one example of where there may be alternative policy measures which would result in the same end-point (reduced health effects at a national level) but which would also require a completely different set of actions by industry or government. However, the inability to quantify and assess all proposed policies on the same basis, prevents such a rationalisation of actions across government departments from taking place. As a result, there is a tendency to consider policies in isolation, in terms of whether ‘to implement or not to implement’, as opposed to considering them in relation to one another through, for example, the use of the NBIR criterion.

It is issues such as these, which have led to some analysts proposing that, instead of CBA, the cost-effectiveness of policies in achieving their main objective be examined. These analysts argue that there may be a range of policies across different government departments which can provide the same types of end-benefits; it is, therefore, important from an overall resource efficiency perspective that the cost-effectiveness of these is considered. Similarly, it may be the case that a number of low expenditure policies could be implemented, which could achieve more overall in terms of benefits, yet at the same costs as a single, broader scale policy. Thus, there may be significant opportunity costs associated with adopting the latter in preference to the former.

10.6 Guidelines for Policy Makers

It is important that CBA (or any of the other decision-aiding methods discussed here) is placed in context. At the simplest level, CBA is a set of procedures for defining and comparing benefits and costs. In this sense, it is a way of organising and analysing data as an aid to decision making. Moreover, a CBA can be done well or done poorly, and there are issues affecting how such appraisals are undertaken. To be done well, the analyst undertaking the CBA must have a sound knowledge of economics, and be sensitive to ethical and philosophical issues. Equally, the informed decision maker needs to be aware that data are always imperfect, that the very process of quantification imposes limitations on the conceptual framework, and that a number of subjective assumptions will always underlie the analysis.

With this in mind, we set out below ten key points which decision makers should bear in mind when commissioning or interpreting the results of CBA.

When developing a policy proposal, consider commissioning a CBA as early on in the process as possible. This will help ensure that the analyst has adequate time to carry out as thorough an analysis as possible. Also take advice on the degree to which new research may be required to develop appropriate valuations for appraisal of environmental and/or health effects.

Consider the likely scope and magnitude of the impacts and thus the likely need for the analysis to go beyond direct costs and benefits. Where indirect and secondary costs and benefits may be an issue, make it clear that they should be examined as part of the analysis.

Make it clear that all costs and benefits should be measured in opportunity cost and willingness-to-pay terms to ensure that the appraisal provides a good indication of the net social effects of adopting any particular policy option. Require the analyst to provide an indication of the extent to which appropriate 'shadow prices' were not used.

Where benefit estimates from previous studies are used to value environmental and health effects, require that the analyst spells out any caveats that should be applied to such values. Make sure that you have a sound understanding of the degree to which use of the transfer values is appropriate, of the reasoning behind any adjustments made to the transfer values and whether or not alternative measures exist which may give different results.

Require information on the reliability of the results, where this includes details of those values and assumptions which are considered to be the least robust. Request that sensitivity analysis on key assumptions be undertaken and use these to determine the robustness of the end results. Do changes in the assumptions made affect the ranking of alternative options? Would an option no longer be justified on net benefit grounds under any probable set of assumptions?

Ask questions about the comprehensiveness of the appraisal and thus its comparability to appraisals of other policy issues.

Compare the end results of the appraisal to some rough rules of thumb in terms of the policy's cost-effectiveness. How well does the policy under consideration perform when compared to other competing environmental policies in terms of the costs per unit of benefit delivered?

Consider how well the policy performs when compared to policies in other areas of government responsibility, which may deliver the same types of benefits. Is there a case for switching funding from one policy area to another?

Use the net benefit investment ratio criterion to allocate funding across a portfolio of policies competing for government funding to ensure that opportunity costs at the cross-policy level are minimised. In so doing, however, make sure you are aware of any differences in the comprehensiveness and reliability of the various appraisal results being used.

Remember that, not only are appraisals useful in their own right, but their use shapes the framework for decisions. Economic appraisals require a formal report and assessment of proposed policies. If conducted in a transparent manner, this has a positive result in that all of the techniques and data used are exposed to possible criticism, discussion, revision and improvement.

10.7 Recommendations for Further Research

This study has demonstrated the difficulty of undertaking a fully comprehensive CBA which will then allow decision makers to 'choose' between alternative measures for implementing a given policy and between competing policies. Even if fully comprehensive analyses are possible, the analysis forms only part of the decision making process - hence, there are further (usually political) factors to consider. With this in mind, we recommend the following areas for further research:

- research on the value of a statistical life and the value of a life year lost which is more tailored to the needs of environmental policy appraisals, taking into account the nature of context of the risks of concern and the population at risk;
- the development of comprehensive appraisal guidelines which address the need to, and enable the inclusion of, direct, indirect and secondary effects within environmental policy appraisal;
- the development of 'approximation' methods for indirect and secondary effects;
- a detailed case study to assess the degree to which the final result actually does change when a full analysis is performed;
- the development of guidelines for reporting requirements for economic appraisals;
- the development of robust guidelines for policy selection techniques; and

- further examination of the potential role of cost-effectiveness analysis of 'cost per life saved' at the policy level and how this could feed into EU decision-making.

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