norden

Nordic Guideline for Cost-benefit analysis in waste management

TemaNord 2007:574

Nordic Guideline for Cost-benefit analysis in waste management

TemaNord 2007:574

ISBN 978-92-893-1555-5

Copies: 311
Printed on environmentally friendly paper
This publication can be ordered on www.norden.org/order. Other Nordic publications are available at www.norden.org/publications

Printed in Denmark

Nordic Council of Ministers
Store Strandstræde 18
DK-1255 Copenhagen K
Phone (+45) 3396 0200
Fax (+45) 3396 0202

Nordic Council
Store Strandstræde 18
DK-1255 Copenhagen K
Phone (+45) 3396 0400
Fax (+45) 3311 1870

www.norden.org

Nordic co-operation

Nordic cooperation is one of the world's most extensive forms of regional collaboration, involving Denmark, Finland, Iceland, Norway, Sweden, and three autonomous areas: the Faroe Islands, Greenland, and Åland.

Nordic cooperation has firm traditions in politics, the economy, and culture. It plays an important rol in European and international collaboration, and aims at creating a strong Nordic community in a strong Europe.

Nordic cooperation seeks to safeguard Nordic and regional interests and principles in the global community. Common Nordic values help the region solidify its position as one of the world's most innovative and competitive.

Contents

Preface

In the preparation of this guideline we have drawn on several existing national and international guidelines. In particular the guidelines by the Danish National Environmental Research Institute, the Danish Environmental Protection Agency and the Danish Forest and Nature Protection Agency (Møller el al. 2000), the Norwegian Ministry of Finance (2005), the Danish Ministry of Finance (1999) and the HM Treasury (2003), as well as publications by the OECD (Pearce et al. 2006; Smith 2005). The Danish guideline, Møller el al. 2000, is currently under revision.

Since a vast number of thorough guidelines on cost-benefit analyses exist, most of them explaining the theory in detail, this guideline aims at issues that are particularly relevant for cost-benefit analyses in the area of waste and resources.

It has also been attempted to combine some key concepts from the life cycle assessment methodology with the cost-benefit methodology as there are a number of similarities between these two decision support tools. In this guideline we recommend to carry out a life cycle assessment or as a minimum an inventory of environmental effects, as the environmental assessment part of the cost-benefit analysis.

This guideline has been prepared by Mette Skovgaard, the Danish Topic Centre on Waste and Resources, Karin Ibenholt, ECON AS and Tomas Ekvall, IVL/Svenska Miljöinstituttet.

A draft version of the guideline has been peer reviewed by Markku Ollikainen, Helsinki University, and Mikael Skou Andersen, Danish National Environmental Research Institute, in August 2006.

The authors would like to thank the experts who participated in the seminar in September 2006 for their comments and contributions.

Copenhagen, January 2007

Summary

In recent years, debates on environmental policies have been marked by discussions whether cost-benefit analysis (CBA) is a suitable tool for gathering information about socio-economic impacts of a given action before political decision-making.

This publication is a proposal for a state-of-the-art guideline for cost-benefit analysis in the waste management sector. One of the aims is to provide insight in life cycle assessment in order to improve the system analysis and the assessment of environmental effects in the CBA.

It is important to keep in mind that CBA is a decision *support* tool, not a decision-*making* tool. The CBA is supposed to assist in the learning process and provide the best available information about the subject in question. Not all information can or will be captured in a CBA and decision-makers may also have other political issues to consider, which means that the CBA results do not represent "the final truth".

Defining objective and scope

Problem to be examined and baseline scenario

The first questions to address are why the CBA should be undertaken, what kind of information it should provide, to whom and for what purpose. Understanding the context in which the analysis is to be conducted is important for defining the problem to be examined.

An initial literature review of studies within the subject might provide inspiration and avoid duplication of work. Involvement of stakeholders in the process is also a way of collecting and making use of available information. In some cases information and data necessary for the analysis are only available from private companies or local authorities. Establishing a steering committee or reference group enables stakeholders to obtain insight in and participate in the development of the study: methodological choices, assumptions etc. This can contribute to enhancing the learning process of stakeholders as well as CBA practitioners, and to improving the CBA. Ultimately, it can make the results of the CBA more readily accepted.

Identifying the target audience, e.g. policy-makers, public administration, industry representatives and NGOs, can help define the problem to be analysed. It may also be desirable to describe the roles and responsibilities of public authorities involved and affected by the policy or project. This would indicate which authorities should be involved in, or at least consulted during the process.

The baseline scenario serves as the reference situation, to which the alternative scenarios will be compared. This scenario is often a continuation of business as usual, and serves to describe the anticipated development in the absence of the policy or project. The description of the baseline scenario should be as detailed as possible, and should account for conditions that are uncertain and assumptions that have been made in the baseline. Moreover, the baseline scenario should account for foreseen developments in demographics, economics (e.g. scarcity of particular resources) and technology over the time horizon of the policy or project.

Identifying alternative scenarios

All relevant alternatives to the baseline scenario should be considered in the analysis. To identify these alternative scenarios, it may be necessary to survey the issue for feasible and desirable options, and to study previous analyses within the field. A steering committee may provide input to the definition of relevant scenarios. In doing so, options that are not feasible for one or another reason (e.g. technical, financial, regulation, time constraints) can be excluded at an early stage. The number of scenarios should not be too high, as the analyst and target audience may lose overview of the entire analysis.

Functional unit

The costs and benefits of waste management are often calculated and presented per kg or tonne of managed waste. The denominator in the unit is called the functional unit. If it is 1 tonne of waste treated, the CBA results imply that the quantity of waste is unaffected by the management measures investigated. Having identical amounts of waste treated in different scenarios makes it possible to simplify comparative analyses by neglecting the production and use of the materials. This simplification is sometimes referred to as the “zero burden assumption”, suggesting that the waste enters the system boundary without burdens.

A CBA that presents the environmental burdens per tonne of waste generated allows for comparison of different options for dealing with this waste, but is inadequate for the identification and assessment of waste prevention strategies. In studies made to assess waste prevention, the functional unit needs to be adjusted. For example, the unit may be the treatment of the annual quantity of waste generated (or expected to be generated) in a geographical area. When scenarios include different waste quantities, the zero burden assumption is no longer valid.

System boundary

It is recommended to pay special attention to the implications of the decision on system boundary, i.e. which life cycle phases are included in the CBA. An illustrative way of presenting the system included in the study is to make a flow-chart showing how the resource or waste stream question is managed.

The most important economic effects are the collection and treatment of waste and the valuation of time spent by consumers on separation of waste (if this is accounted for in the CBA). The valuation of environmental effects may also be of importance.

The most important environmental effects stem from treatment of waste (landfill, incineration and recycling); manufacturing of the material which is replaced through recycling; production of energy which is replaced through incineration of waste with energy recovery; manufacturing of the material which becomes waste (mainly relevant for waste prevention studies); and production of electricity and heat used in the waste management system and the manufacturing of materials.

Geographical boundary

In the CBA the geographical boundary is conventionally often the nation-state. However, as more and more goods (including waste) are traded globally, this guideline supports the recommendation that the conventional focus on the nation-state in CBA should be extended, in particular with regard to environmental effects arising in other countries if:

- the impact relates to an issue where the effect in other countries has been acknowledged as being of joint interest and responsibility by means of an international treaty or international cooperation,
- there is an accepted ethical reason.

Time horizon

The choice of time horizon is important as it will affect the present values of costs and benefits. The only rule is that it should account for all relevant costs and benefits of the scenarios studied.

Environmental effects have very long time horizons. Effects of climate change are, for example, estimated for a horizon of 100 years. The health effect of several substances is still unknown, thus it is difficult to even set a time horizon for the impact of such effects. The environmental effects of landfills (emissions of methane gas, leaching of metals, etc.) can occur for hundreds of years. The emissions also vary over time and depend on the composition of waste.

A Danish guideline recommends that when certain environmental effects occur after the project or policy as such has been terminated, the time horizon should be chosen to account for such environmental effects. This may imply that the time horizon will be infinite. Another approach is to set the time horizon according to the uncertainty of future estimates. The rationale for this approach is that it is difficult to know what the situation will be in 40 or 50 years' time.

Inventory

The inventory is a list of all resources necessary to implement the policy or project. The resources are divided into resource use (inputs to waste management, production etc.) and the result of this resource use (products, environmental effects, health effects, etc.). Ideally, the inventory should cover each year of the project's time horizon as the effects will depend on changes in factors such as demographics and technology. Presenting the effects in a table as shown below would provide an easy overview.

Inventory of effects

	$Year_0$	$Year_1$	….	$Year_n$
Resource use				
Investment (money terms, no.)				
Operational costs				
- labour (man-hours, money terms)				
- natural resources (tonnes, m^3, GJ, money terms)				
- import of goods (no., money terms)				
Result of resource use				
- environmental effects (tonnes, m^3, no.)				
- health effects (no., in percentage)				

Source: Adapted from the Danish Ministry of Finance (1999) and Møller et al. (2000).

It should be decided whether the CBA aims at analysing which scenario is beneficial for society here and now, taking into account the investments already made, or whether the aim is to analyse the long-term benefits to society. The first approach will consider investments already made as sunk cost, and therefore exclude these from the analysis. The second approach will include all investments regardless of current investments, as it aims to analyse the optimal solution in the long term when society is not tied to any current investments. Hence, the first approach will favour systems and policies already in place and is usually only adopted if the CBA is carried out from the perspective of an individual utility. The second approach, which aims to analyse the long term optimal solution for society, is more relevant in CBAs at the level of the national economy.

In order to enhance the transparency and reliability of the CBA as a decision-support tool, it is recommended that all data sources, assumptions and methods used are specified in the analysis.

Economic effects

Effects from collection, separation and transport

Depending on the defined functional unit, the analysis would take its starting point in the generation of the waste stream under consideration. This waste is generated in households, private companies, public institutions, etc. The effects from the activities of collection, separation and transport will, among other things, depend on: “
the waste producer’ (i.e. single-family/multi-storey housing; manufacturing/commercial companies); the type of waste (mixed waste; separate collection/residual waste); and the waste collection system (kerbside; bring banks; recycling centres). Collection of waste from the “waste producer” should cover investments in equipment necessary for the activity, e.g. indoor bins, outdoor bins/containers, trucks, as well as separation and pre-treatment facilities. Likewise, effects from the operation of the activity should be included, and these may cover bags/sacks, manpower, fuel, electricity, administration costs and information campaigns.

As detailed data on waste collection may be difficult to collect, an approximation can be obtained from the public tender collection price from a selection of local authorities. In most cases, it would be fair to assume that waste collection is in competition and therefore the price reflects the marginal costs.

Effects from treatment

In this guideline the term “treatment” describes the activities after the use of the product, i.e. reuse, recycling, recovery and final disposal of waste.

The three waste treatment options (landfill; incineration with energy recovery; and material recycling) generate effects on energy production (heat and electricity from incineration; methane gas from landfill), production of recycled materials, and effects from avoided production of other energy and materials.

In practise, information about input of resources necessary for waste treatment is best obtained from collaborating with or contacting a state-of-the-art facility to gain insight in the investment required and the operational costs. In some respects, information is also available from the facility’s annual accounts or environmental reports. Inspiration may also be found in other CBAs in the field of waste and resources.

Time consumption and space in households

Consumers spend leisure time on separating waste and transporting it to a recycling centre, bring bank or similar. If the time spent on this activity reduces welfare of consumers, then it may be regarded as an additional cost. However, if they undertake these actions on a voluntary basis, the time can be excluded from a CBA.

Source separation requires not only time but also space for storing the multiple waste streams.

Environmental effects

This guideline recommends that the environmental effects be estimated using the methodology for life cycle assessment (LCA). The methodology helps expand the perspective beyond the waste management.

To calculate the environmental effects, it is necessary to collect environmental data on the technological processes and transports in the waste management system. When the functional unit is the annual quantity of waste managed, data on the annual emissions from different activities can typically be added directly to calculate the total, direct environmental effects of the waste management system.

However, an LCA should also include the environmental effects arising outside the waste management system to the extent that they are caused by this system. In other words, the LCA should include environmental effects caused by the production and transport of electricity, fuel, materials and equipment that are used in the waste management system. The LCA should also include the environmental effects that are avoided when energy and material from the waste management system reduce the need for producing energy and material elsewhere.

As a consequence of the broad perspective, an ideal LCA model includes environmental data from a very large number of activities. Simplifications are required to make the study feasible. The general rule is, of course, to focus on getting good data on the activities that are most important for the overall LCA - and CBA - results. For many other activities, it is sufficient to use data from literature or existing databases, which reduces the effort required for data collection.

The following methodological issues are discussed in this guideline:

- identification of the material replaced through recycling,
- choice between using average environmental performance or marginal changes in production volume,
- identification of the technology used for marginal electricity production,
- identification of the heat production technologies that are replaced through waste incineration,
- identification of the source for marginal production of fuel,
- choice of time horizon in the estimation of environmental effects from landfills.

Monetary valuation

The effects identified in the inventory should be assigned a monetary value in order to estimate the social costs and benefits of a project. The definition of the monetary value depends on whether the goods are traded in a market (hence a market price exists) or not (hence no market price exists). The latter includes environmental benefits.

Marketed goods and services

Some guidelines recommend for the analysis of social costs that the effects are expressed in physical units in an inventory and that these effects are directly valued at their social costs. Other guidelines recommend starting with a financial analysis, and that the market prices are converted to *shadow prices,* i.e. the price that reflects the opportunity cost. Which method to choose is partly a question of available data and the resources set aside for the CBA. If a financial analysis exists already, it can be used as the basis for the social cost analysis.

In the CBA it is important to distinguish between the real economic effects from pure transfers of money. Pure transfers have no net effect in a social cost analysis and should therefore not be included in the calculation. Hence, a waste fee should not be included in the CBA.

Taxes

For a good or resource that is traded in a well-functioning market, i.e. a market without distortions, the shadow price is equal to the market price. Markets can be distorted by external effects; taxes used to generate revenues; imperfect competition; and disequilibrium such as unemployment. If this is the case, the market price has to be corrected in order to estimate the shadow price.

The determination of the shadow price for taxed input factors (goods or resources) is treated differently in various guidelines. Taxes on input factors and other goods create a wedge between the willingness-to-pay and the opportunity cost for these products, and this wedge should be accounted for in a CBA.

Shadow prices and taxes

		Public monopoly	Competitive production
Labour	Market wage incl. taxes and employers' contribution * net-duty factor	Market wage incl. taxes and employers' contribution	Market wage incl. taxes and employers' contribution
Input factors	Market price excl. reimbursed taxes * net-duty factor	Price excl. taxes, but incl. taxes correcting for external effects	Price incl. the same taxes as in the private sector

Source: Møller et al. (2000) and Norwegian Ministry of Finance (1997)

A general recommendation is to use the same method as other CBAs in the country the analysis is performed for. If no national guideline exists the method recommended by the Norwegian Ministry of Finance is probably the "simplest" alternative.

Environmental taxes introduced to internalise an external effect (i.e. to correct for a market distortion) need no further correction. If these environmental taxes are believed to be a good estimate of the actual damage, they can be used as a proxy for the environmental cost, but then estimates for these external effects should not be included in the separate analysis of external effects, as this would be to double-count the environmental effects.

Financing projects and policy (marginal cost of public funds)

When a policy or public project is financed via taxes a wedge between private and social costs is created. The tax cost for using general taxes to finance the policy/project will probably differ between countries but not between projects/policies in the same country. For Norway, Denmark and Finland the tax cost is estimated to be 20%. For Sweden the tax cost is 30%.

User charges that do not reflect the user costs, i.e., where there is no direct link between the costs the user actually inflicts and the charge the user has to pay, will also create a wedge between social and private costs, that is a marginal cost of public funds. User charges, e.g. waste fees, which do reflect the user cost, do not need to be adjusted for the tax costs.

Householders' time

There is no consensus on how to monetise the time spent on source separation in households. One approach is based on the assumption that each hour spent on household chores means one hour less available for work. With this approach the value of the leisure time of citizens can be set equal to the full salary cost, including tax, or equal to the net salary, excluding tax. A different approach is to estimate how much it would be worth for consumers to avoid spending time on source separation. Here, time is treated as a non-marketed good and the value of the time is identified through willingness-to-pay methods. A third approach is to account for the marketed as well as the non-marketed aspects of time. This requires information on what share of the source separation time would be spent at work.

Valuation of environmental benefits and costs

Despite the fact that the valuation of environmental effects is often fraught with methodological and practical difficulties, it is usually better to include an uncertain estimation rather than excluding it from the CBA. A decision made without a measure of the value of environmental effects, will *implicitly* assign a value to these effects. Hence, most CBA guidelines recommend that environmental costs and benefits should be included explicitly, and as far as possible measured in monetary terms.

Several methods exist for estimating the monetary values of environmental effects, e.g. emissions and discharges to air, water, etc. If it is decided to transfer a monetary value that has been estimated for another geographical area, it should be considered carefully which parameters that can be transferred and how the transfer is made.

A database for exploring the latest developments within the growing field of valuation techniques has been developed and will be extended with Nordic valuation studies in the beginning of 2007.

Discounting

The choice of discount rate is an important exercise and may have a decisive influence on the outcome of the CBA. Nevertheless, discounting should be made, and this guideline recommends that the national choices of discount rates are applied in the CBA. Moreover, a sensitivity analysis should be made for the discount rate (e.g. +/-2%).

Discount rates in the Nordic countries

Denmark[1]	Finland	Norway[2]	Sweden
6%	5%	4% / 6%	4%

Notes: 1) The Danish Ministry of Finance recommends a discount rate of 6%. The Danish EPA recommends a discount rate of 3% combined with a return of capital calculated as an alternative rate of return of 6%. 2) In Norway a discount rate of 4% is recommended (calculated as a risk free discount rate of 2% and a risk premium of 2% covering systematic risk, whereas the risk premium should be 4% for projects with a substantial systematic risk profile; but for projects less dependent on the economic business cycle, a discount rate of 4% can be applied.

Source: Møller (2003), the Norwegian Ministry of Finance (2005).

When the same discount rate is used for all costs and benefits, it is necessary also to account for the fact that the prices for environmental benefits are likely to increase over time, e.g. due to scarcity of resources. Existing methods for environmental valuation typically do not account for this dynamic effect. This means that they need to be supplemented by a time-dependent technique, if the discount rate is applied to environmental costs and benefits.

Evaluation

When all the effects of a project or policy have been listed in an inventory and monetised, the net benefit for each period is calculated and discounted to period 0. This exercise of weighing the net benefits for each period is the calculation of the net present value (NPV). The general rule is to choose the project(s) with a positive net present value, and if ranking is necessary the project with the highest NPV should be chosen.

Sensitivity analysis; analysis of risk and uncertainty

The purpose of a sensitivity analysis is to study how sensitive the result is to changes in key assumptions. For this reason, it is recommended always to conduct a sensitivity analysis of the key assumptions made in a CBA.

In a CBA it is the expected cost that should be used, not the most likely value. Uncertainty, on the other hand, is when the probability cannot be calculated. As a result, it is difficult to deal with "pure" uncertainty. Nevertheless, it is recommended to include an analysis of the risk and uncertainty.

Presentation of results

In the presentation of the results, the objective and scope of the analysis should be presented in order to allow the decision-maker and other stakeholders to assess whether the objective has in fact been achieved.

This guideline recommends that the CBA result is presented as the net present value. In doing so, it is also recommended that net present value is split into: social costs excluding externalities, environmental costs, time spent by householders and space occupied by waste collection equipment (if included in the CBA).

The environmental effects arising outside the nation-state should as a minimum be estimated and presented in physical units. However, it is clearly preferable that these environmental effects are also monetised.

In cases where a monetary valuation of environmental effects is not desirable, the environmental effects should be quantified in physical units if possible. Otherwise, the effects should be listed (e.g. odour, noise, etc.)

The presentation should also include a description of the data that are uncertain and highlight the assumptions that are important for the outcome of the CBA. Likewise, if any relevant aspects or elements have been omitted from the analysis, this should be presented as well.

Consequences for specific groups in society should also be described. An identification of how different groups are affected by a policy or project will also show who will be in favour of it.

1. Introduction

1.1 Motivation

In recent years, debates on environmental policies have been marked by discussions whether cost-benefit analyses (CBA) are a suitable tool for gathering information on socio-economic impacts of a given action before political decision-making.

The Treaty of the European Union (the Maastricht Treaty) requires action to take into account several factors one of which is “the potential benefits and costs of action or lack of action”, (Pearce 2001, 173). In its communication on a thematic strategy on the prevention and recycling of waste from 2003, the European Commission stated that CBA is a suitable tool for evaluating waste policy targets, (CEC 2003a). Furthermore, the European Council of Ministers has declared that CBAs are to be carried out in connection with proposed amendments to rules put forward by the European Commission. Therefore, it is likely that CBAs will be used increasingly before decisions are made on new legislation and amendments to existing directives in the field of waste and resources.

Moreover, life cycle thinking and life cycle assessment (LCA) has gained acceptance as tools for waste management planning and policy-making. They are now being used in various contexts, ranging from local planning to policy making at national and international levels. An example of this is the thematic strategy on the prevention and recycling of waste presented by the European Commission in December 2005, (CEC 2005a). For LCA an international standard has been developed in order to ensure that an LCA has a certain quality and transparency regarding the assumptions made. In case the LCA compares different products the results are to be peer reviewed. No such international “standard” exists for the preparation of CBA. However, the establishment of a standard of some kind may enhance the reliability of the CBA as a decision-support tool.

A review of CBAs of paper and cardboard waste management concludes that there are considerable differences in the application of the method and the transparency of estimates, Villanueva et al. (2006). Especially the definition of system boundary seems to cause problems, and in general there is room for improvement of the environmental assessment.

1.2 Objectives

The objective of this publication is to provide a proposal for a state-of-the-art guideline for cost-benefit analysis in the waste management sector. In addition, the objective is to provide insights from LCA in order to improve the system analysis and the assessment of environmental effects in CBA.

Where possible, the guideline will provide recommendations on decisions to be made. However, it is important to stress that the purpose is not – and it is not possible – to give *concrete directions for how to carry out a given CBA*. Examples will be given of how a given problem has been solved in specific cases, and explanations will be given for the consequences of the different choices. Hence, the guideline serves a checklist of issues to be considered during a CBA study.

The target group of these guidelines is CBA practitioners, staff members in public (European, national and local) administrations, and staff members in public and private companies in the waste and resource management sector.

The guideline does not aim at presenting the theoretical aspects of the CBA, as the economic theory is thoroughly explained elsewhere.

1.3 Approach

The basis for the present guideline is *existing knowledge* (in the Nordic countries and Europe) about the application of CBA in the environmental field and specifically in the field of waste and resources.

First step has been a *literature search* with a view to drawing up an outline of existing guidelines for CBA, as well as analyses carried out primarily in the waste and resources field, but also in other fields that may inspire the guideline.

Second step has been the preparation of the actual contents of this guideline. In this process, recommendations in existing Nordic guidelines on CBA have been presented and sometimes contrasted. The two guidelines from the Danish and Norwegian Ministries of Finance are general in nature, and the guideline from the Danish Ministry of the Environment is specific for CBAs for environmental projects.

The guideline has been presented at a seminar in Copenhagen in September 2006, giving a number of Nordic experts the opportunity to present their assessment and discuss selected, critical spheres of issues.

1.4 Structure of the guideline

The guideline is designed in a way that it can be used as a manual for persons preparing a concrete CBA. Under the different activities the choices to be made will be described and recommendations given where possible. The guideline is structured according to the typical stages of a cost-benefit analysis:

In Chapter 2 a brief introduction to CBA is given and the fundamental principles are presented and the limitations of this decision support tool are discussed. In addition, a brief introduction to LCA is given.

In Chapter 3 the initial considerations for the preparation of the CBA are presented. In addition to the definition of objective and scope of the CBA, the initial considerations also include decisions on the activities to be included in the analysis (the system boundary) and the geographical boundary. In particular, the CBA might benefit from further attention to these two aspects.

In Chapter 4 issues relevant for the inventory of all the physical effects arising from the implementation of a policy or project are discussed. The inventory contains input to economic activities, such as investment and other input factors, and environmental effects.

In Chapter 5 various ways of monetising economic activities and environmental effects are shown. Examples of relevant valuations for the Nordic countries are also given.

One of the most debated issues of the cost-benefit analysis is discussed in Chapter 6: Discounting. When all costs and benefits are calculated on the basis of the inventory and the monetary values, they need to be discounted to present values.

In Chapter 7 the net present value is presented as the main evaluation criterion. Sensitivity analysis and methods for analysing risk and uncertainty are briefly introduced.

Finally, Chapter 8 contains an extensive literature list with links to internet sources. In Annexes I and II references to guidelines on CBA and Nordic CBA studies are presented. At the very end of the guideline, in Annex III, a glossary of relevant concepts and terms is found.

2. Introduction to CBA and its limitations

The intention of cost-benefit analysis is to compare the economic efficiency implications of alternative options that may be implemented to address a particular project or policy requirement. The benefits of an option are contrasted with its associated costs (including the opportunity costs) within a common analytical framework. To the extent that is possible, all costs and benefits should be expressed in a common unit or numeraire, and this is monetary value.

The main advantage is that a CBA gives a comprehensive overview of all important effects from a policy or project, and that these effects can be compared through the use of a common unit. By using a common unit for all effects, the benefits and costs of implementing a policy or project can be weighted against each other to help decision-makers choose the alternative that gives the highest net benefit to society. As a general rule projects with a positive net benefit should be implemented; while projects with a negative net benefit should be rejected. In practice however, not all beneficial projects or policies will be undertaken simultaneously, either because a budget restriction might limit the possibilities, or because the projects are mutually exclusive. Then the projects will have to ranked according to their net benefit[1].

It should be emphasised that CBA is a decision *support* tool, not a decision-*making* tool. The CBA is supposed to provide the best available information about the subject in question. However, not all information can or will be captured in a CBA and decision-makers may also have other political issues to consider, which is why the CBA does not represent "the final truth".

2.1 Types of CBA

Two types of CBAs exist: ex-ante and ex-post. An ex-ante study examines the efficiency of various options for implementing a policy, e.g. an EU Directive, whereas an ex-post study examines whether the actual, implemented policy has been efficient.

It has been debated whether ex-ante and ex-post CBAs reach the same result or whether ex-ante CBAs overestimate the costs. Vanner and Ekins (2005a, 12) conclude that: "Overall ex-ante cost estimates tend to overes-

[1] Ranking is explained further in Chapter 7.

timate the cost of implementing Directives. The case studies considered here have indicated that this difference is typically of a factor about two, and in many cases this tends to be driven by conservatism in setting assumptions about technological development. In addition, system complexity and (in hindsight) limited scope of the technologies considered add to the likelihood of overestimation, because it is often only in the process of implementing regulations that the least-cost solution is found". However, the workshop where these conclusions were presented, found that the case studies did not present overwhelming evidence for a large, systematic overestimation of ex-ante estimated cost compared to ex-post cost, (Vanner and Ekins 2005b, 5). Rather, it was suggested that a factor of one fourth or one third might be a more valid estimate.

In a literature review of 27 studies, and a comparison to another study of 25 examined regulations, (MacLeod et al. 2006, 5) found that although ex-ante costs were overestimated for around half the regulation studies, they were also frequently underestimated and occasionally accurate.

2.2 Theoretical foundation

The theoretical foundation of CBA is economic welfare theory and is based on the following assumptions, (Pearce et al. 2006, 42):

- An individual's preferences reflect the tastes and habits concerning consumption goods. Individuals' preferences may differ so that one individual prefers good A whereas another individual prefers good B.
- The preferences define the willingness-to-pay (WTP) for a benefit, and the willingness to accept (WTA) for a cost.
- Individuals' preferences can be aggregated to reflect the social preference, or utility.
- Individuals that benefit from a policy or project are, in principle, able to compensate the individuals that lose, and still have a net benefit[2].

An analysis of private cost uses financial values, whereas a CBA uses social values, and these can differ from the financial values. When estimating the social value the benefit should express the affected people's willingness-to-pay and the cost the opportunity cost (that is the value the used resources would have received in the best alternative use).

For further reading on the theoretical foundation, see Pearce et al. (2006), Boardman et al. (2006), or Hanley and Spash (1993).

[2] This is the Kaldor-Hicks compensation criterion.

2.2.1 Distributional effects

Although a policy or project will be beneficial for society, it may have an adverse effect on certain individuals or groups in society, such as local authorities, households, waste management companies, manufacturers etc.

Identifying how different groups will be affected enables policy-makers to compensate those groups that are affected negatively, at least in principle. Also, any conflicting interests of the parties affected by the policy or project are illustrated (Møller et al. 2000, 165). The private cost of a policy or project will influence a party's preference for that policy or project, and so by identifying these effects it becomes clear who will be in favour of or against the policy. Hence, Møller et al. (2000, 165ff) recommends including an analysis of the private cost for relevant groups. This analysis is largely based on the same inventory as the CBA.

The Norwegian Ministry of Finance recommends that the distributional effects for those groups in society that are particularly affected should be described and be part of the decision-making process. However, the weighting of different groups' benefits in the final decision-making is a political question and not part of the cost-benefit analysis (Norwegian Ministry of Finance 2005, 10).

Pearce et al. (2006, 224ff) notice that in general, conventional CBAs disregard distributional effects. Nevertheless, a first step to incorporate distributional effects would be to identify the effects and possibly to provide detailed information about them. Then the issue of equity could be considered in the CBA by assigning implicit and explicit distributional weights to calculate the net benefits received by individuals[3].

For further discussion of distributional effects, see Boardman et al. (2006), Møller et al. (2000), and Norwegian Ministry of Finance (2005).

2.3 Limitations of CBA

The CBA has advantages as well as disadvantages, and some of the latter have been widely debated. Several objections to CBA have been raised (e.g. Pearce 2001, 175ff):

Credibility: the costs and benefits of an option, and their monetary value, are often highly uncertain. Wynne (1992) distinguishes four types of uncertainty; risk, uncertainty, ignorance and indeterminacy. If the possible outcomes can be defined and their probabilities can be assigned in a meaningful way, one is talking of risks. If the possible outcomes are identifiable, but their probabilities cannot be determined, one is faced with

[3] Kriström has suggested a three-level hierarchy: 1) identifying and cataloguing how project-related cost and benefits are distributed; 2) calculating implicit distributional weights; 3) re-calculating the project's net benefit using explicit distributional weights. C.f Pearce et al. (2006, 297): Kriström, B. (2005), 'Framework for Assessing the Distribution of Financial Effects of Environmental Policies', in Y. Serret and N. Johnstone (eds.), *The Distributional Effects of Environmental Policy*, Cheltenham, Edward Elgar and OECD, Paris, forthcoming.

uncertainty. Ignorance refers to when we do not know what we do not know. Finally, indeterminacy is used to describe situations in which the complexity of the system is so large and so little is known about the relevant parameters and their relationships that modelling becomes a matter of hit and miss (Mickwitz 2003). Where ignorance and indeterminacy may be at play, and it will often be the case because of the complexity of social and environmental issues, decision-making will have to rely on other tools in addition to the CBA. A CBA can account for risk, and the sensitivity analysis can, in principle, deal with uncertainty. However, if the full uncertainty is properly accounted for, the CBA results might encompass a level of uncertainty that makes them difficult to interpret and use. On the other hand, if the uncertainty is not properly accounted for, the study lacks in credibility.

Moral objections: CBA is based on utilitarian moral philosophy, which means it is based on the assumption that all types of negative effects can be compensated by positive effects. It can be argued that certain negative effects, e.g., the loss of human life or the extinction of a species, cannot be compensated for by positive effects. Furthermore, individuals that benefit from a policy or project typically do not, in practice, compensate the individuals that lose. As a result, the CBA should be complemented by an identification of negative (and positive) effects that are difficult to compensate (or off-set) by other effects; and by an analysis of the distribution of positive and negative effects for various groups in society.

The efficiency focus: the objective of the CBA is to assess how efficient efficiency options are when they are implemented in the current economic, technological and social context. Policy-makers, however, often have additional objectives such as fairness, equity, long-term sustainability, competitiveness, employment, regional balance, etc. A full basis for a decision might require additional analyses to cover these issues.

Flexibility: decision-makers may feel that CBA results, by indicating the most efficient option, usurp the freedom of choice from the decision-makers. Here, it is important to remember that the CBA is a decision support tool, and that other effects or political considerations may not be encompassed in the CBA.

Participation: the CBA has been accused of not involving relevant stakeholders. By presenting one-dimensional results there is a risk that it closes the door for debate. Stakeholder participation and debates are important to resolve conflicts of interest. Without them, important stakeholder groups might not accept the option selected by the decision-makers. This can be a significant problem for controversial options such as the construction of a waste incinerator or an expansion of the source separation scheme. Since CBA does not resolve conflicts of interest, it cannot replace the decision-process but only provide input to this process.

However, although Nordic CBAs on waste management do not resolve conflicts of interest, they have highlighted some of them. Early CBAs on recycling highlighted the conflict between the benefits (environmental improvement) and costs (particularly the time spent on source separation). Subsequent CBAs on collection systems highlighted the conflict between the benefits of kerbside collection (less time required from households) and bring systems (less effective, lower collection cost). In this way, CBAs can contribute to an informed debate. The debate can be stimulated during the CBA by involving a steering committee or reference group that includes important stakeholders. The completed CBA report can stimulate further debate, if it highlights important methodological choices and data uncertainties.

Capacity: expertise in both economics and natural science is necessary to perform a CBA. Moreover, a certain level of expertise is also necessary among the people using or judging the results, for example to participate in a debate that is based on CBA results. If stakeholders are involved in the steering committee, they get the opportunity to learn about the individual study and about CBA in general. This will make them more able to participate in any debates spurred by the completed CBA.

Hence, the CBA does have limitations. Most of them can be overcome or alleviated through a few careful measures:

- start by generating and screening ideas for relevant options (see Section 3.2),
- involve a steering committee, to achieve learning and acceptance (see Section 3.1),
- write a transparent report highlighting important methodological choices and uncertainties (see Section 7.4), and
- carry out or recommend complementary analyses to achieve an improved basis for decisions (see Section 2.4 and Box 5.2).

2.4 Alternatives to CBA

In situations where a policy is focused on one main environmental effect (or benefit) and where this effect may be difficult to monetise, the cost-effectiveness analysis (CEA) could be a sensible alternative to the CBA. For example, the CEA method is useful if decision-makers want to know which policy is the most cost-effective for reducing CO_2 emissions.

The cost-effectiveness analysis measures the cost per physical unit of an environmental effect, e.g. EUR per tonne of CO_2 reduction, or alternatively, the maximal environmental effect for a given cost. In other words, the CEA does not include all environmental effects, only the one which is of major concern.

Contrary to the CBA, the cost-effectiveness analysis cannot reveal whether a project or policy is beneficial for society and hence worth undertaking, Pearce et al. (2006, 274). The cost-effectiveness analysis can only show which of several policies that has the lowest cost per unit given that it is already decided that one of the policies should be implemented.

If it is not desirable to monetise environmental goods (e.g. landscapes and biodiversity), the Norwegian Ministry of Finance (2005) suggests a method for a systematic approach to non-monetised impacts. In this way, impacts which have no monetary value could still be dealt with properly in a CBA. The approach is presented in Box 5.2.

Other methods, such as multi-criteria analysis, are further presented by CEC (2003b), Pearce et al. (2006), Söderbaum (2006) and eftec/DEFRA (2006).

2.5 Brief introduction to Life cycle assessment (LCA)

Life cycle assessment (LCA) is a methodology for evaluation of the environmental impacts potentially generated in the lifetime of a product, a system or a service.

It is an ambitious methodology, and it covers all stages in the life cycle of the products or systems investigated, from the cradle (material extraction) to the grave (final disposal). It attempts to cover all physical exchanges in the life cycle of a product with its surroundings, be it inputs of auxiliary materials and energy consumption, or outputs of emissions, waste and usable energy. Input data on physical exchanges are collected and compiled in a so-called life cycle inventory analysis. The results from this inventory may, for example, for a 40g paper sheet be expressed as the consumption of 100g of wood and of 0.25kWh energy, generation of 0.1kWh energy, emission of 64g CO_2 and 0.1g SO_2, along with a host of other inputs and outputs (see Figure 2.1).

In an ideal LCA, all inputs and outputs should be covered in the inventory. In practice, the completeness of the inventory is limited by the analyst's knowledge of the interactions of the system being investigated with society and the environment.

Once an inventory of all inputs and outputs has been completed, these are grouped into potential impact categories such as resource consumption, climate change, acidification or toxicity, and translated into environmental impacts following natural science causality knowledge. For instance, with respect to climate change, methane emissions are multiplied by a factor of 21 to convert them to carbon dioxide (CO_2) equivalents, since the climate change impact potential of 1 kg methane is 21 times that of a kg of carbon dioxide.

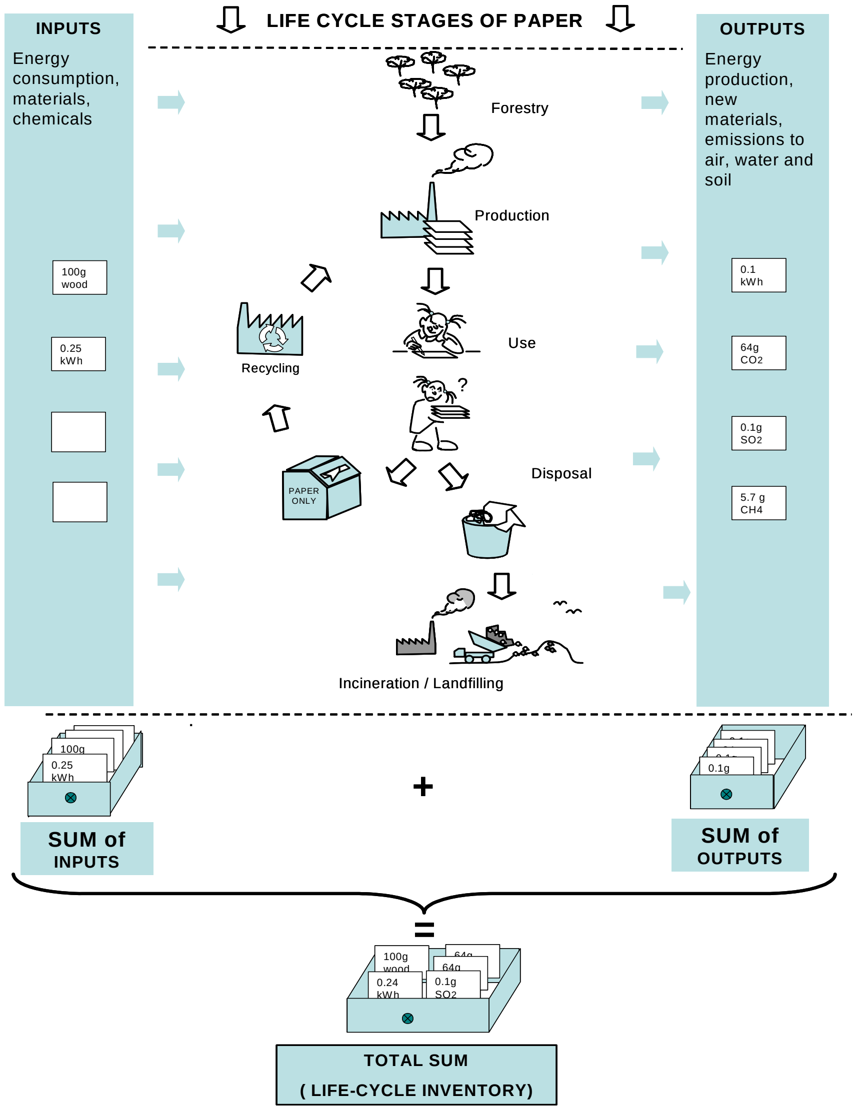

Figure 2.1. Illustration of the data collection phase of the life cycle assessment of (a sheet of) paper.

Source: Reproduced from Danish Topic Centre on Waste and Resources (2006)

The impacts may be grouped further and weighted against each other to derive a single impact potential score such as eco-points or ecological footprint, which is easier to communicate in a decision-making context. For this aggregation process, an LCA uses references for comparison based on the functionality of the products or services being compared, and units such as person equivalents (that is, the amount that a citizen emits of a given substance per year), to convert emissions to a single indicator. Such aggregation inevitably carries with it a certain degree of subjectivity, and requires decisions over which impact is more important in a policy context – for instance is climate change more or less important than human toxicity? This is a weighting process that has similarities with the monetisation process of the CBA.

In contrast to a CBA, LCA uses mostly physical and chemical data, both in the inventory and in the calculation of impacts. Only in certain areas such as the simultaneous production of two or more products (for instance, beer and yeast production), data about the economic value of the two products may be used in LCA to split between them the "responsibility shares" of the environmental loads occurring during production.

An international standard for LCA has been developed, and handbooks are available (e.g., Guinée 2002), as well as scientific reviews of recent developments, Rebitzer et al. (2004), and Pennington et al. (2004). Separate publications describe how to apply the method on waste management systems, Finnveden (1999) and Clift et al. (2000). The application of LCA to waste management, and a few methodological key issues, are discussed in Section 4.2.

3. Defining objective and scope

The decision on what to study and the scope of the analysis will affect the entire analysis and thereby the conclusions. Hence, there is plenty of reason why it is worth spending time and efforts in preparing the analysis carefully.

The issues discussed in this section are:

- definition of the problem to be examined
- construction of the baseline scenario
- description of relevant alternative scenarios
- the scope of the analysis

In this context, the baseline scenario is the “no policy intervention scenario” or business-as-usual scenario that the alternative scenarios will be compared to or measured against. The alternative scenarios describe the development that is anticipated if the policy intervention, investment in new technology, or another kind of project is implemented.

The scope of the analysis covers considerations of the reference unit for comparison, system boundary, geographical boundary, and time horizon.

3.1 Objective of analysis

A cost-benefit analysis may be conducted as part of the preparatory work for setting new targets in a waste strategy or waste management plan. It may also be conducted to examine the most appropriate way of implementing an EU directive, or whether a new waste treatment technology would be more beneficial to society than the current technology.

Before beginning a CBA, it should be ensured that the necessary resources in terms of staff and possibly consultancy are available to achieve the defined objective (and scope). If the objective of the analysis does not match the resources available for the study, it would be better to reconsider the study. Perhaps the objective needs to be constrained to studying just a few alternative scenarios; or to undertake a screening of the most important effects; or perhaps the study should be made using a different approach, e.g. a cost-effectiveness analysis. It is recommended to be explicit about any constraints in the analysis, especially if elements relevant for the analysis have been omitted.

3.1.1 Definition of the problem to be examined

The definition of the problem or subject to be examined should include a description of the purpose of initiating the policy or project as precisely as possible. The basic question is why the CBA should be undertaken and what kind of information it should provide, to whom and for what purpose.

Understanding the context in which the analysis is to be conducted is important in order to define the problem to be examined, and it should ease the definition of the baseline and alternative scenarios. For example, if the purpose of the CBA is to examine various ways of implementing an EU directive, the directive will contain certain legal requirements that the national regulation must comply with. Already existing national regulation might need to be amended before the new policy can be adopted. The historical development of the waste management system may also be relevant and influence the future policy, e.g. if systems to distribute energy from waste incineration plants exist already, or if a particular expertise has been developed which may support export of a technology.

An initial literature review of studies within the subject might provide inspiration and avoid duplication of work. Involvement of stakeholders in the process is also a way of collecting and making use of available information. In some cases the information and data necessary for the analysis are only available from private companies or local authorities. Establishing a steering committee enables stakeholders to obtain insight in the methodology applied and assumptions made whereby the result of the CBA may be accepted more easily.

Identifying the target audience, e.g. policy-makers, public administration, industry representatives and NGOs, can help define the problem to be analysed. Policy-makers, be it local, regional or national, may have different needs for information and decision support, depending on the scale of the policy or project, or the complexity of the problem.

It may also be desirable to describe the roles and responsibilities of the public authorities involved and affected by the policy or project. This would indicate which authorities should be involved in, or at least consulted during, the process. In addition, it might indicate where additional information could be gathered.

3.1.2 Construction of the baseline scenario

The baseline scenario serves as the reference situation, to which the alternative scenarios will be compared. This scenario is often a continuation of business as usual, i.e. a scenario without the policy or project that is assessed in the CBA. It may also be the case that the current development is not allowed to continue. In such a case, the baseline scenario can be either a hypothetical counterfactual, or adjusted to meet legal requirements. In any case, the purpose of the baseline scenario is to describe what would have happened in the absence of the policy or project.

The description of the baseline scenario should be as detailed as possible and should account for the conditions that are uncertain and the assumptions that have been made in the baseline. In the absence of a new policy, questions in this respect might be: how much waste would be generated and collected; how would it be collected and treated; would the waste be exported for recovery or would it be managed domestically?

The baseline scenario should account for foreseen developments that are independent of the policy or project. A cost-benefit analysis may have a tendency to become a static analysis by considering only conditions that are relevant at the time of appraisal. However, economic activity, population and number of persons per household may be on the increase, which will affect the use of resources and generation of waste. So may an ageing population. The behaviour of producers and consumers is also likely to change over the 20–30 years which is a typical time horizon for an analysis. Furthermore, there may be other regulatory changes in the pipeline or technology innovations that will influence the use of resources, management of waste and emissions during the life cycle. An outlook could illustrate the likely, future waste generation and management[4] as well as other demographic and economic variables. Different methods for constructing a baseline scenario are presented in Box 3.1.

[4] The European Topic Centre on Resource and Waste Management of the European Environment Agency is developing baseline projections for a series of waste streams.

Box 3.1 Methods for creating a baseline scenario

Various methods can be used to construct a baseline scenario. These include:

Trend extrapolation. A simple approach to constructing a policy baseline is to assume that trends visible prior to the policy change would have continued unchanged if the policy measure had not been implemented.

Econometric methods. Econometric models may be estimated which, for example, link pollution levels to various economic variables (*e.g.* the level of gross national product), and which include a 'dummy variable' for the date of introduction of the policy measure. The model can then be used to make a 'counterfactual' prediction of what would have happened to pollution levels if everything else had remained unchanged, except that the policy had not been introduced.

Linear programming techniques can be used to indicate how the decisions of firms might change in response to different constraints and incentives; the problem with these measures is that they assume some form of optimal decisionmaking, which may in practice be unrealistic.

Often it will be better to use ***'judgmental' methods*** rather than any of the other techniques to describe the baseline in the absence of policy. However, one problem with definitions of the baseline scenario constructed purely on the basis of judgment is that the outcome of the evaluation study will depend critically on the judgments made; there may easily be scope for doubts about the realism of such a baseline

What can be done depends partly on the availability of suitable data. In turn, data availability depends partly on the institutional setting of the evaluation. Issues of commercial confidentiality may obstruct access to some of the key data needed.

Source: OECD (1997, 97)

In describing the baseline scenario it is relevant to differentiate between policy objectives and policy measures. The policy objective may be to achieve a 50% recycling rate which can be achieved with various mixes of instruments, whereas the policy measure is a specific instrument, e.g. obligatory, separate collection of organic waste from households.

It is recommended to use the baseline scenario consistently throughout the study in order to avoid any confusion of what is used as the counterfactual.

3.2 Identifying alternative scenarios

The objective of a CBA could be to provide information on one of the following questions: should a given waste stream be recycled, incinerated or landfilled? What should the optimal level of recycling be for a given waste stream? Which policy measures would be better at achieving the targets?

All relevant options to the baseline scenario should be considered in the analysis. In order to be able to identify these alternative scenarios, it may be necessary to survey the issue for feasible and desirable options, and to study previous analyses within the field. A steering committee may also provide input to the definition of relevant scenarios. In this way, options that are not feasible for one or another reason (e.g. technical, financial, regulation, time constraints) can be excluded at an early stage. All the alternative scenarios should, of course, cover relevant options. The number of scenarios should not be too many, as the analyst or target audience may lose overview of the entire analysis.

The number of alternative scenarios needed differs according to the objective of the analysis. For many purposes two scenarios may be sufficient: the baseline and the situation where the policy or project has been implemented. On the other hand, if the policy or project can be implemented by using different (packages of) measures, more scenarios are needed.

When defining alternative scenarios it is necessary to have or obtain information on the effectiveness or efficiency of various waste management systems, policy measures etc. For example, how much paper and cardboard can a local authority expect to collect via a kerbside collection system compared to a bring bank system? And what is the efficient density of bring banks? Much of this information will be obtainable either from local authorities and their knowledge from the operation of a system, or from various studies. As for policy measures, some information on the effectiveness may also be available from studies, although the experience from one country or type of instrument can rarely be transferred directly to another country or policy area. Often, some specific circumstances may influence the effectiveness.

3.3 Definition of scope

The definition of scope comprises the decision on what the analysis should include and, consequently, exclude. This exercise regards a delimitation of: the system (technology) under study, the geographical boundary, and the time horizon. First, however, the reference unit for comparison should be defined. In this guideline it is suggested to apply the same term as in life cycle assessments: the functional unit.

3.3.1 Functional unit

The costs and benefits of waste management are in many cases calculated and presented per kg or tonne of waste treated. The denominator in the unit is called the functional unit, a term borrowed from LCA theory. If it is 1 tonne of waste treated, the CBA results imply that the quantity of waste is unaffected by the management measures investigated. Having identical amounts of waste treated in different scenarios makes it possible to simplify comparative analyses by neglecting the production and use of the materials (Finnveden 1999). This simplification is sometimes called the "zero burden assumption", suggesting that the waste enters the system boundary without burdens.

A CBA that presents the environmental burdens per kg or tonne of waste generated allows for comparisons of different options for dealing with this waste, but not for analysis of changes in the quantities of waste generated. It is inadequate for the identification and assessment of waste prevention strategies. It also fails to account for the serious challenges posed by a continuation of the short-term and long-term trends of increasing waste flows, and consequently do not give information on how large capacity for waste treatment is required.

In studies made to assess waste prevention, the functional unit needs to be adjusted. It can, for example, be the treatment of the annual quantity of waste generated (or expected to be generated) in a geographical area. Such functional units have been used in environmental assessments of waste management. Xará et al. (2004) used the annual quantity of waste in the city of Porto as the functional unit in a life cycle assessment. Olofsson et al. (2004) compared waste prevention to different waste management strategies, using the annual quantity of waste in Sweden as the functional unit. The quantity of waste varied between the scenarios because the analysis accounted for the reduction in waste quantity resulting from potential waste prevention measures.

Adjusting the functional unit is obviously a way to facilitate the assessment of waste prevention. This measure may appear simple enough, but if different scenarios include different waste quantities, the zero burden assumption is no longer valid. It is reasonable to demand that such studies include the environmental burdens associated with the production of all the materials that eventually become waste. This makes the assessment more complicated.

It can be relevant to use the annual quantity of waste generated in a geographical area as the functional unit also when the quantity of waste is identical in the different scenarios. With this functional unit, the results of the CBA will reflect the anticipated total costs and benefits of waste management in the region concerned. Such results can be easy to communicate and relate to, particularly if the CBA aims at contributing to waste management planning in this region.

3.3.2 System boundary - coverage of the life cycle

Another important decision at an early stage of the analysis is the choice of system boundary. This term is also borrowed from LCA theory, and it indicates which life cycle phases are included in the analysis.

In any analysis, CBA, LCA or other, the scope of the system to be studied will have to be decided. However, in this guideline it is recommended to pay special attention to the implications of a decision on the system boundaries, and how it might affect the result of the analysis. If the functional unit is defined as "the management of one tonne of paper waste", the relevant effects may not just arise from waste management of used paper, they may also arise from the avoided resource use, e.g. logging of wood and manufacturing of new paper due to recycling, or avoided use of fuels due to energy recovery. Thus, including or excluding effects that may not be considered to be a direct part of the waste management phase are likely to influence the outcome of the analysis.

There is a risk of making the analysis very comprehensive if *all* elements and environmental effects in the waste management system should be included and assessed. Thus, the question is what are the most significant ones from a socio-economic perspective? In order to answer this question, in reality a rough assessment will have to be made of which elements are most significant from an environmental and economic point of view. A literature review, e.g. Ekvall (1999), would reveal that the most important environmental effects stem from:

- treatment of waste: landfill, incineration and recycling activities,
- manufacturing of the material which is replaced through recycling,
- production of energy which is replaced through incineration of waste with energy recovery,
- manufacturing of the material which becomes waste (mainly relevant for waste prevention studies), and
- production of electricity and heat used in the waste management system and the manufacturing of material.

In the economic analysis the collection and treatment of waste entail the dominant costs. If the householders' time for separation of waste is included, it will generally be a significant cost as well. The valuation of environmental effects may also be of importance.

An illustrative way of presenting the system included in the study is to make a flow-chart showing how the resource or waste stream in question is managed. An example is shown in Figure 3.1 for the management of source separated waste and residual household waste. In a CBA, the economic and environmental system boundary should be identical. However, the prices used in the economic analysis give information about the upstream costs of producing the good or service or the downstream cost of waste management. For example, revenue from sale of energy produced through incineration is deducted from the incineration cost in order to obtain the net cost of incinerating one tonne of waste[5]. In other words, the price of materials and energy entering and leaving the waste management system can be used as a proxy for the economic analysis of activities outside the waste management system. As a result, economic data typically need only be collected outside the waste management system, when there are good reasons to suspect that the prices of materials and energy flows are poor approximations of the economic effects of these flows.

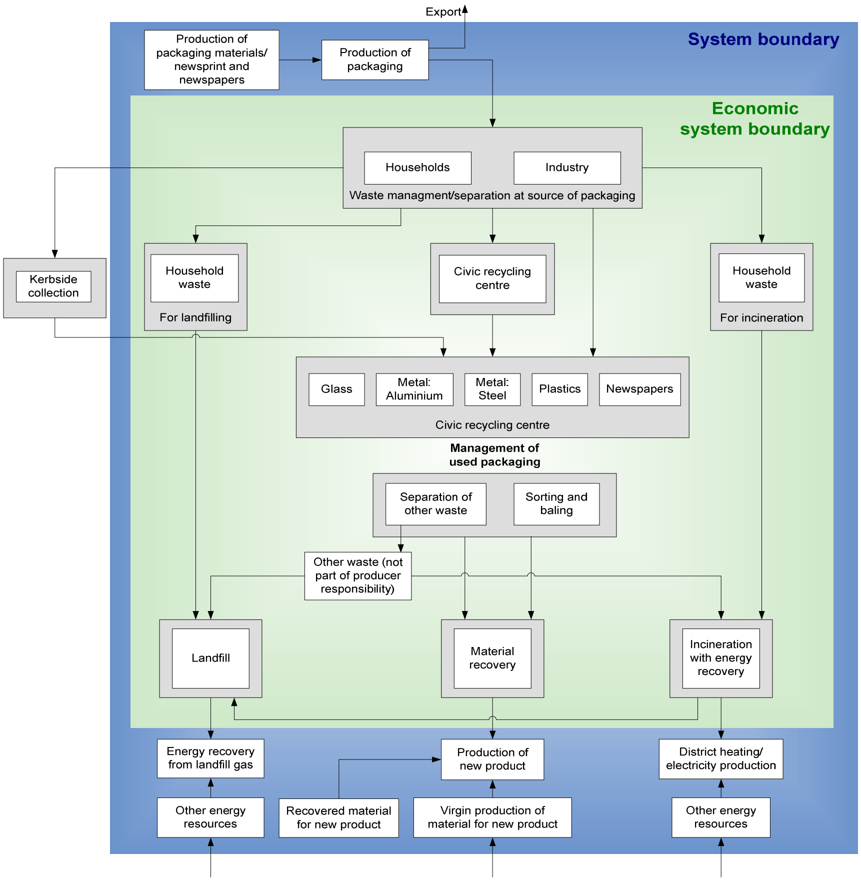

Figure 3.1 System boundary (for the environmental and economic data collection): management of source separated waste and residual household waste

Note: The environmental system boundary (marked in dark-shaded colour) is broader than the economic system boundary. Transport is not shown in the chart but is included in the analysis.

Source: Reproduced from Bäckmann et al. (2001, 58)

3.3.3. Geographical boundary

An often debated issue is the geographical coverage of the analysis. This is an area where there is usually a difference in approach of the CBA and the LCA. In the LCA the geographical boundary is typically global, i.e. it

[5] See Table 4.4

includes emissions taking place abroad, be it in China or Germany. In the CBA the geographical boundary is often national and focuses on domestic costs and benefits, because it is based on the preferences of the domestic population and aims at an efficient use of domestic, economic resources6.

Since many environmental effects are transboundary the environmental impact of an emission may occur not only in the country where the emission arises but in neighbouring countries as well.

In the case where an emission originates in the country performing the CBA, but causes unpriced external effects abroad, it is not disputed that the effects caused abroad normally should be included in the CBA, Møller et al. (2000).

However, when the project or policy analysed affects imports or exports, the problem is how to value the external effects related to such resources or products. Frequently, the problem is bypassed by using the price at the border, assuming either that there are no environmental costs or externalities in the extraction, manufacturing or use of these resources or that these externalities already are taken care of in the exporting or importing country. This approach is based on the idea that each nation is free to decide how to value its own environment.[7]

Having said that, it should be stressed that in today's global economy a good is often manufactured, consumed and disposed of in multiple parts of the world. Accordingly, ignoring environmental effects from production or extraction abroad (which may be the greater part of the total effects) may seriously bias the insights gained through the CBA. In a comparison between domestic recycling and export of waste the purely national approach would, as a paradox, clearly favour the export alternative, due to the lower environmental cost from a narrow national perspective.

Møller et al. (2000, 32) suggests that the environmental effects be described separately for the country in question and for foreign countries. This allows illustrating the weight of foreign environmental effects in the total CBA result.

According to Pearce et al. (2006, 55) the conventional focus on the nation-state in CBA needs to be extended, in particular with regard to impacts arising in other countries if:

---

[6] See Section 2 regarding the theoretical foundation of the CBA. The geographical boundary of the CBA does not have to be the entire nation; it can also be a municipality or a county.

[7] In principle this means that any environmental costs in the exporting country are supposed to be included in the export price. If an environmental tax is levied for example on the production of a good, then this tax is reflected in the price paid at the border. If the production of a good is subject to environmental regulations, that good is more expensive to produce than it would otherwise have been, and therefore, again, more expensive to import. If there are no environmental taxes levied on the good/resource, it is assumed that the exporting country values (additional) local environmental damages from the manufacturing of the good as equal to zero.

the impact relates to an issue where the effect in other countries has been acknowledged as being of joint interest and responsibility by means of an international treaty or international cooperation,
there is an accepted ethical reason[8].

This approach is recommended in this guideline, and it implies that greenhouse gases, transboundary pollution and many other environmental effects arising abroad should be included in a CBA.

Including all economic activities and environmental effects in a CBA is not straightforward and is likely to be a comprehensive task. While the inventory of environmental effects and the prices of traded goods abroad usually are readily available the only feasible approach for the preferences (monetary values) of the environmental (non-traded) goods is to transfer benefit estimates by use of purchasing-power-parities.

In general, a CBA is conducted to study the effects of one country's policy. As a result of this limited geographical coverage or scale of the study, most studies in the field of waste and resource management assume that the policy analysed will only induce a marginal change in economic activity, i.e. it will not affect the relative prices. If, on the other hand, the objective is to study the effects of a common European policy and the geographical scope is EU-27, it is clearly a possibility that the relative prices will change and that the currently observed market prices cannot be used in the analysis. For example, a new policy that aims at increasing the recycling rate in the European Union may result in lower prices for recycled material due to increased supply, and in higher prices for collection equipment due to increased demand. In principle, this dynamic can be included in a CBA although it is a complex task. Rather, in such cases it is recommended to use market equilibrium models or general equilibrium models to estimate the effects of changes in the market.

3.3.4 Time horizon

The choice of time horizon is important as it will affect the present values of costs and benefits, and thereby the result of the analysis. Nevertheless, there is no fixed rule for defining the time horizon, except that it should account for all the relevant differences in costs and benefits of the scenarios studied.

Pearce et al. (2006, 56f) argue that previously the time horizon was equal to the physical or economic life time of the investment. Here, the time horizon for infrastructure was typically between 30 and 50 years, and for housing it could be around 100 years.

Environmental effects have very long time horizons as it may take several years before an effect becomes apparent or has an actual impact

[8] Fx where the issue involves cultural assets such as a World Heritage Site, or where a nation feels a moral obligation to account for its impacts on others, Pearce (2001, 172).

on ecosystems and human health. The effects of climate change are estimated for a horizon of 100 years, and the health effect of several substances is still unknown (see also Section 4.2.6 on the time horizon for landfill emissions).

When estimating the environmental effects – and thus the monetary value – from landfilling waste the time horizon is particularly important as emissions of methane gas and leachate can occur for at least tens and hundreds of years, respectively, (COWI 2000). Factors that further complicate the estimation and valuation are that these emissions vary over time and depend on the composition of waste. An example of the emission of methane gas over time from various waste streams is shown in Figure 4.5.

Møller et al. (2000, 33) recommend that when environmental effects occur after the project or policy as such has been terminated, the time horizon should be chosen to account for such environmental effects. This may imply that the time horizon will be infinite. Another approach is to set the time horizon according to the uncertainty of future estimates, Pearce et al. (2006, 56f). The rationale for this approach is that it is difficult to know what the situation will be in 40 or 50 years' time.

4. Inventory

An inventory is a listing of all effects that arise from implementing a policy or project measured in their natural physical units, i.e. tonnes, m^3, km, etc.

The description of the baseline scenario and the relevant, alternative scenarios should include all important foreseen pressures, be it economic effects, environmental effects or other external effects.

The economic effects comprise typical, traded production input factors: capital, labour and various materials. In this context, the environmental effects comprise the external effects that are not traded, i.e. emissions from the relevant production and waste management processes; the use of land, water and other natural resources; noise, etc.

In order to estimate the economic and environmental effects, considerable information and knowledge about the system is necessary. Not only skills in economics but also skills in natural science are required to quantify and assess the pressures.

First, the inventory of economic effects is presented. This section includes both general considerations for the analysis as well as considerations mainly regarding the economic effects. The time spent by householders on waste management activities, and space for storing recyclables in households are also discussed in this section. Subsequently, the inventory of environmental effects is presented. This inventory is "the environmental assessment" of the CBA.

4.1 Economic effects

To allow for a proper calculation of the net present value, the inventory needs to include the resource use and the result of this resource use for each year the project runs.

At this stage, effects are listed in their physical units. Some effects will already be stated in monetary terms.

An easy way of getting an overview of all the effects is to present them in a table structured as in Table 4.1. Here, the resource use (inputs to waste management, production etc.) and the result of this resource use (environmental effects, health effects, etc.) are shown for each year of the policy or project time horizon. This guideline recommends illustrating how the resource use and the result of the resource use is expected to develop over the time horizon of the policy or project as environmental effects and thus monetary values may vary over time.

Table 4.1 Inventory of effects

	$Year_0$	$Year_1$	….	$Year_n$
Resource use				
Investment (money terms, no.)				
Operational costs				
- labour (man-hours, money terms)				
- natural resources (tonnes, m^3, GJ, money terms)				
- import of goods (no., money terms)				
Result of resource use				
- environmental effects (tonnes, m^3, no.)				
- health effects (no., in percentage)				

Note: To show the general nature of the inventory, the table includes also environmental effects and health effects, although the environmental assessment is discussed in Section 4.2.

Source: Adapted from the Danish Ministry of Finance (1999) and Møller et al. (2000).

The inventory can be made in actual quantities for each scenario or as changes from the baseline. The former has the advantage of being straightforward and transparent, whereas the latter has the advantage of showing how an alternative scenario deviates from the baseline. In any case, the baseline and the alternative scenarios will have to be quantified.

Before starting the inventory, it should be decided whether the policy or project is to analyse what is beneficial for society here and now, taking into account the investments already made, or whether it is to analyse what is beneficial to society in the long term. The first approach will consider investments already made as sunk cost, and therefore exclude these from the analysis. The second approach will include all investments regardless of current investments, as it aims to analyse the optimal solution in the long term when society is not tied to any current investments. Hence, the first approach will favour systems and policies already in place and is usually only adopted if the CBA is carried out from the perspective of an individual utility. The second approach which aims to analyse the long-term optimal solution for society, is more relevant in CBAs at the level of the national economy.

In order to enhance the transparency and reliability of the CBA as a decision support tool, it is recommended that all data sources, assumptions and methods used are specified in the analysis.

4.1.1 Effects from collection, separation and transport

Depending on the defined functional unit, the analysis would take its starting point in the generation of the waste stream under consideration. This waste is generated in households, private companies, public institutions, etc. The effects from the activities of collection, separation and transport will, among other things, depend on "the waste producer" (single-family/multi-storey households; manufacturing/commercial companies), the type of waste (mixed waste; separate collection/residual waste), and the waste collection system (kerbside; bring banks; recycling centres). Collection of waste from the "waste producer" should cover the investments in equipment necessary for the activity, e.g. indoor bins, outdoor bins/containers, trucks, as well as separation and pre-treatment facilities. Likewise, the effects from the operation of the activity should be included, and these may cover bags/sacks, manpower, fuel, electricity, administration costs and information campaigns.

It may be difficult to obtain complete information at a very detailed level of information about the collection costs (e.g. investment in trucks, manpower, fuel use, etc.) if such inventories do not already exist. A surrogate can be established by obtaining the public tender collection price from a selection of local authorities. In most cases, it would be fair to assume that waste collection is in competition and therefore the price reflects the marginal costs. It is true that the calculation of the tender price may have used a different discount rate than the one applied in the CBA, hence there is a divergence of assumptions, but the public tender collection price can be used as an approximation in want of better data.

An example of effects from separate collection of organic waste and residual waste from two different types of household is presented in Table 4.2.

Table 4.2 Example of annual effects from separate collection of organic waste and residual waste, per household

	Organic waste		Residual waste	
	Single-family	Multi-storey	Single-family	Multi-storey
Investment:				
bin, indoor	825 DKK	34 DKK	-	-
useful life	10 years	5 years	-	-
bin, outdoor	63 DKK	62.5 DKK	299 DKK	179 DKK
useful life	10 years	10 years	10 year	10 year
truck	-	-	-	-
useful life	-	-	-	-
Operation:				
bags, indoor	168 plastic bags	168 plastic bags	-	-
distribution, bags	4 DKK	4 DKK	-	-
sacks, outdoor	26 sacks	1.6 sacks	86 DKK	-
maintenance	22 DKK	1.7 DKK	-	-
collection*	231 DKK	25.91 DKK	359 DKK	140 DKK
information	3 DKK	3 DKK	-	-
administration	10 DKK	10 DKK	-	-

Note *: The collection reflects the price paid by the local authority to the operator for the service, and it includes the transport of waste to the treatment facility.

Source: Extract from Damgaard et al. (2003, 114)

4.1.2 Effects from treatment

In this guideline the term “treatment” describes the activities after the use and collection of the product, i.e. reuse, recycling, recovery and final disposal of waste.

The inventory of effects will depend on the kind of treatment included in the scenarios under study. Table 4.3 provides an overview of the various factors that influence the cost of treatment for incineration, landfilling, composting and anaerobic digestion.

The choice of waste treatment may also generate effects on energy production (heat and electricity from incineration; recovery of methane gas from landfill), on the production of recycled materials, and on the effects from avoided production of other energy and materials. These effects were also shown in Figure 3.1 on the system boundary in section 3.3.2.

Table 4.3 Factors that influence the cost of treatment

	Incineration plant	Landfill	Compost plant	Anaerobic digestion
Land acquisition	X	X	X	X
Capacity of plant	X	X	X	
Requirements for land per unit of capacity			X	
Plant utilisation rate	X	Rate at which landfill is filled	X	
Requirements for:				
Treatment of flue gas	X			
Engineering (affected by geology and proximity to sensitive aquifers)		X		
Collection of methane gas		X		
Conditions for utilisation of digestate and liquor (e.g. can digestate be applied directy to land or must it be stabilised via aerobic composting)				X
The choice of technology / process			X	X
Costs for daily cover		X		
The purity of source separation			X	
Choice of input materials used				X
The nature and length of contracts and the materials received			X	
Price for support of energy production				X
Efficiency of energy recovery / production	X			X
Recovery of materials and the revenues from this	Energy, metals	Energy (methane)		Energy
Revenues for sale of product (Compost: depend on quality of input material and maturity of end product)			Compost	Digestate
Treatment and disposal / recovery of ash residues	X			
Aftercare		X		

Source: Hogg (2002)

In Hogg (2002) detailed breakdowns are also available for a crate incinerator (Germany); landfill (the United Kingdom) and intensive composting (Italy). In addition, comparative costs or unit costs of the treatment methods are presented for a number of EU Member States. The Danish association of waste management companies, RenoSam, has prepared fact sheets for its members on the costs and revenues from collection, separation and treatment of various waste streams as part of a benchmarking study[9], (RenoSam).

A more detailed overview of technologies for diversion of biodegradable municipal waste from landfill is presented in Crowe et al. (2001). These include centralised composting, home and community composting, anaerobic digestion, incineration, pyrolysis and gasification. For each technology the advantages and disadvantages are presented, and so are the typical costs. Crowe et al. (2001) also discuss the conditions that are necessary for the establishment of a market for recycled products.

In practise, information about input of resources necessary for waste treatment is best obtained from collaborating with or contacting a state-of-the-art facility to get insights in the investment required and the operational costs. In some areas, information is also available from the facility's annual accounts or environmental reports.

Inspiration may also be found in other CBAs in the field of waste and resources. In Annex II, Nordic CBAs on waste are listed.

[9] The cost per tonne has been split into costs for machines, buildings, operational cost, costs and revenues from recycled materials for the years 2003 and 2004. Fact sheets have been made for: Collection of domestic waste; Collection of paper and cardboard; Operation of civic recycling centres; Separation of paper and cardboard; Incineration; Composting; and Landfill of mixed wastes.

Table 4.4 Inventory for the incineration process (excluding environmental effects)

Activity	Resource use	
Capacity of the facility	483,000	Tonnes
Expenditures, capital		
Investment: incineration plant	2,000,000,000	DKK
- Useful life	25	Years
Investment: machinery	700,000,000	DKK
- Useful life	40	Years
Resource use, operation		
Labour		
- Staff	149	Men
- Pension	3,000,000	DKK
Raw materials		
Lime ($CaCO_3$)	3,000	Tonnes
Chemicals	2,000,000	DKK
Lye	390	Tonnes
Energy		
- Electricity	42,300	MWh
- Natural gas	256,000	m^3
- Oil	526	m^3
- Peak load, natural gas and oil	1,000,000	DKK
Other	52,000,000	DKK
Maintenance		
- Machines	44,000,000	DKK
- Buildings	5,000,000	DKK
Disposal of residues		
- Landfill	15,000	Tonnes
- Recycling	107,000	Tonnes
Production		
- Electricity	115,300	MWh
- Heat	953,000	MWh

Source: Strandmark et al. (2002a)

For instance, detailed inventories of incineration, composting and anaerobic digestion can be found in Damgaard et al. (2003) and Strandmark et al. (2002a). Damgaard et al. (2003) also include inventories for collection and pre-treatment for anaerobic digestion. An example of an inventory for the incineration of 483,000 tonnes of waste is shown in Table 4.4. As indicated, the inventory includes disposal and recycling of residues and production of electricity and heat for sale. This information is used for the calculation of shadow prices in Table 5.2 in Section 5.1.

Inventories of material recycling or facilities for recycling are scarce. Data for recycling of plastics in Denmark are given in Strandmark et al. (2002a), and include: investments in buildings and machines, labour, maintenance, energy, raw materials and residues.

4.1.3 Householders' time consumption

Householders may spend leisure time on separating waste and transporting it to a recycling centre, bring bank or similar. If the time spent on this activity reduces householders' welfare, then it may be regarded as an additional cost. However, if householders undertake these actions on a voluntary basis, it can be excluded from a CBA (Smith 2005, 39).

Different estimates and assumption regarding the time consumption have been used in different studies. The Swedish Consumer Agency (1997) investigated source separation in households in four Swedish towns: Boden, Mark, Täby and Helsingborg. They concluded that source separation (including cleaning, home composting and transport to pick-up points) requires on average 15-25 minutes per household per week. Most of this time was spent on cleaning packaging and transporting the fractions.

According to Farm (1997), cleaning all packaging in a family of four requires 18 minutes/week. This is consistent with the results from the Swedish Consumer Agency.

Bäckman et al. (2001) and Ekvall and Bäckman (2001) calculated the time required for source separation based on measured values of the time required to clean packaging of different kinds, estimated values of the number of packaging per family, and estimated values of the number of trips and time required for transporting the separated waste stream to pick-up points. Their results were significantly lower than the results presented by the Swedish Consumer Agency and by Farm.

Sterner and Barteling (1999), Bruvoll et al. (2002), and Berglund (2006) carried out surveys using questionnaires and interviews to investigate the time spent on source separation by householders. Their values were much higher than the ones presented by the Swedish Consumer Agency and by Farm in 1997. The high results from the questionnaire may, at least in part, be explained by the fact that householders tend to overestimate the time needed for boring activities, (Bruvoll et al. 2002). Furthermore, the time reported in the questionnaires might have been valid for entire households rather than single individuals in the households (Berglund 2006).

In a recent study, Sahlin et al. (2007) disregarded the questionnaire and interview surveys and estimated the time required for source separation of paper and plastic waste by calculating the geometric average between the results from the Swedish Consumer Agency (1997), Farm (1997), Bäckman et al. (2001), and Ekvall & Bäckman (2001). The results are presented in Table 4.5. They can be interpreted as a best estimate of the additional time required to separate these waste streams in Swedish homes, compared to leaving the material in the mixed household waste.

Table 4.5 Estimated time requirement for source separation of paper and plastic waste streams in an average Swedish household

	Average total time	For cleaning	For sorting	For transport
	h/tonne	h/tonne	h/tonne	h/tonne
Paper and hard plastic packaging:	72	19.3	19.3	33.5
Other paper (newsprint, journals, note pads, envelopes):	53	0	19.3	33.5
Soft plastic packaging, diapers:	86	0	19.3	67

Source: Sahlin (2006)

4.1.4 Space in households

Source separation requires not only time but also space for storing the multiple waste streams. A plausible assumption is that source separation only takes place to the extent that there is room in the homes. In other words, reduced source separation would not result in smaller dwellings. This means that the space requirements of source separation do not entail any extra financial cost. However, as a result of source separation, less space is available in the homes for other purposes. When storage space in kitchens is a limited resource, the space requirement can be considered an external cost. Ekvall & Bäckman (2001) assume this to be the case in 20% of the households. They also assume the floor space required to store a waste stream (paper packaging) to be approximately 3 dm * 2 dm = 0.06 m^2.

4.2 Environmental effects

This guideline recommends that the environmental effects be estimated using the methodology for life cycle assessment (LCA). This methodology helps expand the perspective beyond the waste management system. When waste management includes recycling of materials, the boundary of the system is expanded to include the avoided production of that material which is replaced through recycling (see Figure 3.1). When waste management includes energy recovery, the system boundary is expanded to include the avoided combustion of fuel that is replaced by waste. This is important, since the indirect environmental effects caused by surrounding systems, such as energy and material production, often override the direct impacts of the waste management system itself.

As stated in Section 3.3, it is often relevant to use the annual quantity of waste in a geographical area as functional unit. Hence, the LCA is a calculation of the environmental effects (emissions, use of natural resources, etc.) that are caused by the annual waste management in this area. The functional unit can also be 1 tonne of waste treated, or 1 tonne of a specific waste stream, and in this case the LCA results present the environmental effects per tonne of waste.

To calculate the environmental effects, it is necessary to collect environmental data on the technological processes and transports in the waste management system. When the functional unit is annual waste management, data on annual emissions from different activities can typically be added directly to calculate total, direct environmental effects of the waste management system.

However, an LCA should also include the environmental effects arising outside the waste management system to the extent that they are caused by this system. In other words, the LCA should include environmental effects caused by the production and transport of electricity, fuel, materials and equipment used in the waste management system. As stated above, and indicated in Figure 3.1, the LCA should also include the environmental effects that are avoided when energy and materials from the waste management system reduce the need for producing energy and materials elsewhere.

As a consequence of the broad perspective, an ideal LCA model includes environmental data from a very large number of activities. Simplifications are required to make the study feasible. The general rule is, of course, to focus on getting good data on the activities that are most important for the total LCA results. The production of equipment (factories, vehicles, etc.) is typically environmentally insignificant and can be excluded from the LCA (Lindfors et al. 2005). For many other activities, it is sufficient to use data from literature or existing databases, which reduces the effort required for data collection.

A few methodological key issues are discussed in the following subsections:

- identification of the material replaced through recycling,
- choice between modelling average environmental performance or marginal changes in production volume,
- identification of the technology used for marginal electricity production,
- identification of the heat production technologies that are replaced through waste incineration,
- identification of the source for marginal production of fuel,
- choice of time horizon in the modelling of environmental effects from landfills.

Further information on how to apply LCA on waste management systems has been published, for example, by Finnveden (1999) and by Clift et al. (2000).

4.2.1 Alternative material production

If a material is recycled it often replaces other material, recycled or virgin, in new products. The environmental benefits depend strongly on what material is replaced. The ideal method to account for this effect is to expand the system under study to include the unit processes that are actually affected by an increase or reduction in the flow of recycled material from the waste management system. In practice, it can be difficult to identify what unit processes are actually affected by a change in this flow. Recycled material (from our system) can replace material of the same type, i.e. virgin material or recycled material (from other systems). It can also replace completely different types of material or no material at all. For example, printing paper can be recycled into new, recycled printing paper or into other paper products such as tissue paper (downcycling). Simplifications are required to make the methodology operational. The discussion on simplifications presented in this section is based on Ekvall & Weidema (2004).

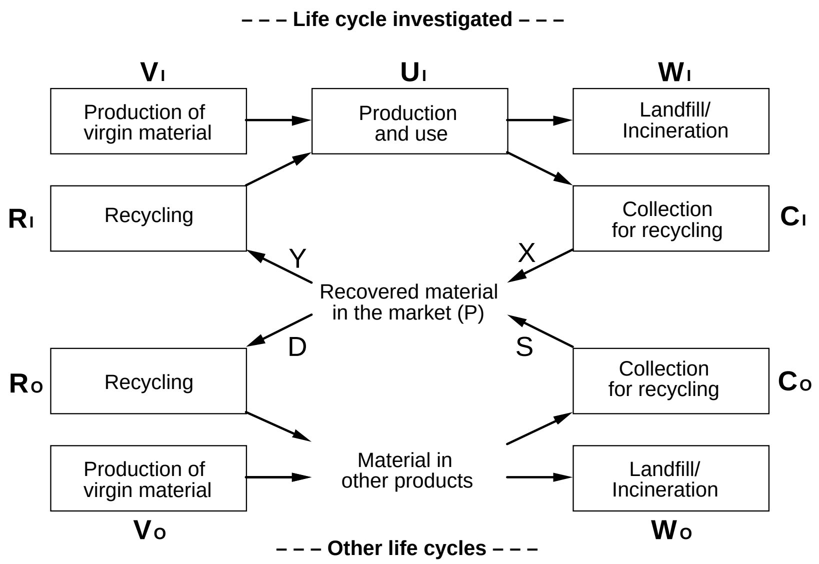

Figure 4.1 A conceptual model of open-loop recycling through a market for recovered material

Source: Ekvall (2000)

A first line of simplification is to assume that the recycled material mainly substitutes virgin or recycled material of the same type. This assumption is, of course, valid unless the recycled material substitutes other types of material. With this simplification, it is still necessary to establish to what extent recycled material from our waste management system replaces virgin material and to what extent it replaces recycled material from other systems. The static, conceptual model in Figure 4.1 can be used for this investigation. In this model, Y, X, D and S are flows of a specific type of recovered material to and from the market for that recovered material, and P is the price of the recovered material. The environmental inputs and outputs of different parts of the life cycle investigated are denoted V_I, R_I, U_I, W_I, and C_I. The corresponding inputs and outputs from other life cycles are denoted V_O, R_O, U_O, W_O, and C_O.

If the amount of recycled material from our waste management system is changed by ΔX, the effects on other life cycles can be calculated from the price elasticity of supply and demand in the market for recovered material. If X and Y are small compared to D and S, the price elasticity of supply (η_S) and demand (η_D) respectively is:

(1) $$\eta_S = \frac{\Delta S/S}{\Delta P/P}$$

(2) $$\eta_D = \frac{\Delta D/D}{\Delta P/P}$$

The effects on the demand (ΔD_X) and supply (ΔS_X) of a change ΔX can be calculated as follows, (Ekvall 2000):

(3) $$\Delta D_X \approx \frac{\Delta X\,\eta_D}{\eta_D - \eta_S}$$

(4) $$\Delta S_X \approx \frac{\Delta X\,\eta_S}{\eta_D - \eta_S}$$

This elasticity approach requires that the relevant price elasticity values be identified. Values for price elasticity are generally identified through the use of time series and econometric models. The price elasticity strongly depends on the time horizon of the study. In general, the price elasticity is larger in a long-term perspective than in a short-term perspective since in the long-term perspective, decision-makers are able to adapt to changes in the price when making investments. The price elasticity also depends on, e.g., the collection schemes and the legislation in place at that time and in that location. As a result, the price elasticity should ideally be separately identified for the context of each individual CBA. Unfortunately, this is not likely to be feasible. Instead, further simplifications are necessary.

For this second line of simplifications, several alternatives exist:

- Use default values for the price elasticities, for example the values that are presented by Palmer et al. (1997) and summarised by Ekvall (2000); these values can be inserted as η_S and η_D in equations 3 and 4 above to calculate estimates of how the flow of the material to and from other life cycles is affected,
- Assume that the demand and supply are equally elastic ($-\eta_D = \eta_S$),

- Assume that the demand or the supply is completely inelastic (η_D or η_S is zero),
- Develop multiple scenarios based on different choices among the above approaches.

The alternatives are shown in Figure 4.2.:

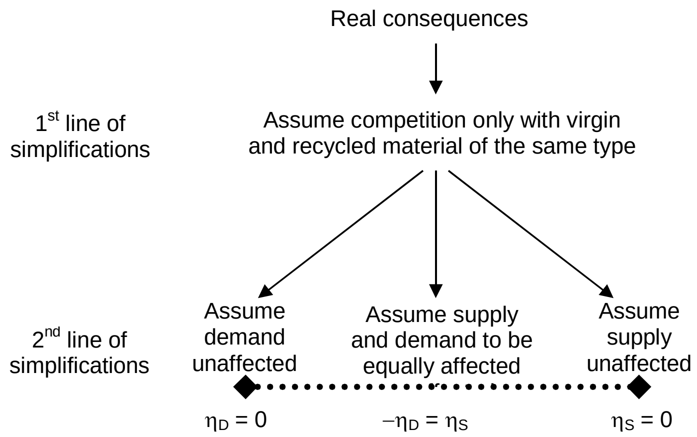

Figure 4.2 Example of possible simplifications in modelling of recycling. Additional alternatives for the second line of simplification are to use default values for the price elasticities or to develop multiple scenarios based on different values for the price elasticities

Source: Ekvall & Weidema (2004)

The advantage of using default values is that the CBA utilises available knowledge. The main danger lies in a sense of false security. The actual elasticity in a particular recycling case can differ a great deal from the default values. For the price elasticity for the supply of old newsprint, literature includes estimates ranging from 0.06 to 1.70, (Palmer et al. 1997). This extremely large span is caused, at least to some extent, by case specific factors such as the time horizon of the study and the time and place where the material was collected for recycling.

As an alternative to using default values, it can be assumed that supply and demand are roughly equally elastic. The consequence of such an assumption is that 50% of the recovered material from the waste management system is assumed to replace material recovered from other life cycles and the remaining 50% is a net increase in total recycling. With this alternative approach, the CBA does not utilise all available knowledge, but it takes into account the fact that recycled material from the system can replace virgin and recycled material from other systems. This approach has been used for example for modelling the effect of PET recycling in an LCA on beverage packaging, (Ekvall et al. 1998).

The third option is to decide whether the supply or the demand is the most inelastic and set this elasticity to zero. In a model based on this approach, recovered material from the system investigated replaces only virgin material or only material recovered from other systems. This approach might be easier to apply than the approaches above, because the system investigated becomes less complicated. Ekvall et al. (1998) used this approach to model the consequences of glass, steel and aluminium recycling. This simplification will probably not affect the results significantly if the difference between the actual price elasticity of supply and demand is large for the recovered material.

Apparently, there is large uncertainty in the price elasticity of supply and demand. When this uncertainty appears significant for the conclusions of the CBA, it is recommended that this uncertainty is accounted for through the use of different scenarios based on different assumptions related to price elasticity.

4.2.2 Marginal vs. average data

Marginal and average data convey different and complementary types of information. Marginal data represent how a small change in the output of products and/or services from a system affects the environmental effects from this system. To calculate marginal data, it is necessary to identify the technology that is affected by a change in demand for the products and/or services. Average data represent the average environmental effects for producing a unit of products and/or services in the system. They are calculated by dividing the total environmental effects from the system with the total output of products and/or services.

Marginal and average data can both be used in environmental assessments for learning purposes as well as decision-making, (Ekvall et al. 2005). However, the CBA aims to estimate and monetise the effects of a policy or project, and the choice of data should be consistent with this aim. This, in many cases, implies the use of marginal figures; however, Azapagic & Clift (1999) distinguish between marginal and discrete effects on an activity. Marginal effects are infinitesimal effects on production volume, reflecting the environmental effects of the technology affected by a marginal change. Discrete effects are substantial changes in production volume or complete changes from one activity to another. Substantial effects should be estimated using incremental data, (Azapagic & Clift 1999). These are likely to depend on the scale of change, as seen in Figure 4.3. Complete changes should be modelled using average data.

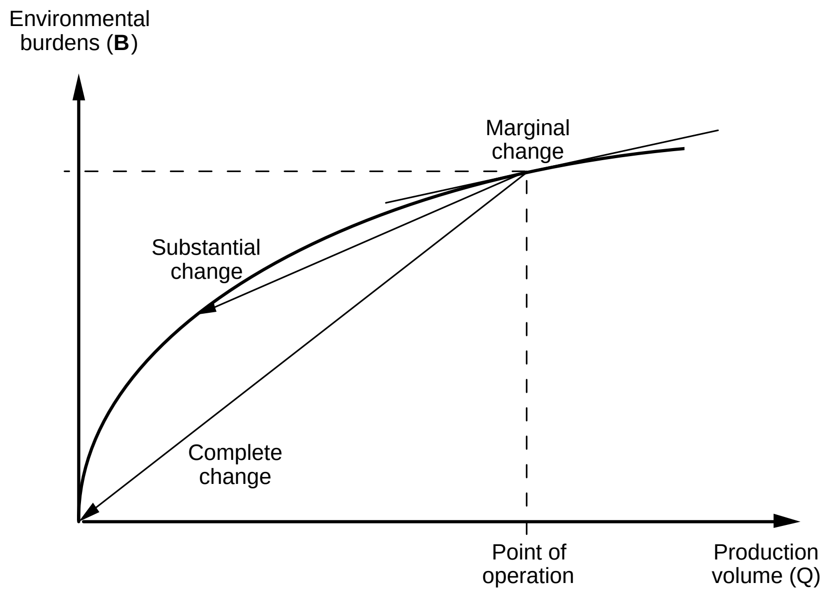

Figure 4.3 Marginal, substantial, and complete changes in a production process

Source: Azapagic & Clift (1999)

Most waste management policies and projects can be expected to have marginal effects on the production of bulk materials (e.g., steel, aluminium, polyethylene) based on virgin or recovered raw material. They can also be expected to have marginal effects on the production of most energy carriers: electricity, heavy fuel oil, petrol, etc., for which total production volume is very high. They can have substantial effects on part of the waste management system and on the production of district heat (see Section 4.2.4). They can, however, also cause complete changes in parts of the waste management system, such as the initiation of a new system for waste collection. This implies that the environmental assessment in the CBA should ideally be based on:

- average and incremental data for most of the waste management system, depending on the expected effects of the project or policy,
- incremental data for the production of district heat, and
- marginal data for recycling processes as well as for most production processes outside the waste management system.

Weidema et al. (1999) presented a five-step procedure for identifying technologies affected by marginal changes. Each step essentially aims at answering a single question:

- What are the relevant time aspects?
- Are specific processes or overall markets affected?
- What is the trend in the market?

- For what technologies is the production volume flexible?
- What technology is actually affected?

Regarding the first step, it is useful to distinguish between the long term and short term. Long-term marginal data reflect the environmental effects of the technology, if any, where the production capacity is affected by a marginal change in demand. Short-term marginal data represent the technology where a marginal change affects the utilisation of existing production capacity. As indicated in Section 4.1, the long-term perspective is probably relevant in most CBAs.

Despite the five-step procedure, the identification of marginal technologies - and the corresponding, marginal environmental data - is still not always straightforward (Åström 2004). The identification of scale-dependent incremental data can also be difficult. For these reasons, the CBA practitioner often needs to use other data as approximations or substitutes for the ideal data.

4.2.3 Marginal electricity

Marginal electricity is electricity from power plants that are affected by a change in electricity demand. In CBAs of waste management in the Nordic countries, the choice between marginal and average data is particularly important for electricity production. The CBA can include production of electricity that is used in the waste management systems and electricity used for producing materials that are replaced through recycling. It can also include electricity production that is avoided through energy recovery in the waste management system. The choice between marginal and average data is important because of the large difference in environmental impact of average and marginal electricity production in the Nordic countries. The electricity mix includes hydropower in Norway and Sweden as well as Swedish nuclear power. These technologies contribute to the relatively small emissions to air of average Norwegian, Swedish and Nordic electricity production. Marginal electricity production, on the other hand, is typically argued to be based on fossil fuel and to result in large emissions to air.

The specific case of marginal Nordic electricity production plays a significant role in the Swedish energy debate. In this context, marginal effects often denote short-term marginal effects only, i.e. effects that occur before the production capacity can be adapted to the change in electricity demand. With this narrow definition, marginal electricity in all Nordic countries is currently produced by coal power, i.e., coal-fired plants for separate electricity production, see for instance Werner (2001) and ECON (2002a). In the future, the short-term marginal electricity is likely to be produced by natural gas-fired plants with combined cycle technology (Gustavsson & Börjesson 1998, Werner 2001). However, it has also been argued that the dismantling of a further nuclear reactor would be compensated not only by coal power or natural gas but by a mix containing, for example, hydropower and wind power as well as combined heat-and-power production (CHP) (Swedish National Energy Administration 2001). Such an analysis goes beyond the concept of short-term marginal effects.

Frees & Weidema (1998) applied the five-step procedure described in Section 4.2.2 to identify long-term marginal electricity, i.e., electricity produced with a technology where the production capacity is affected by a change in electricity demand. They concluded that long-term marginal electricity in the Nordic countries will be produced either in new coal-fired power plants or in new power plants fuelled by natural gas. However, Ekvall et al. (1998) argue that the long-term marginal technology can be a mix of new plants for fossil fuel and existing Swedish nuclear power plants, if the analysis takes into account that a change in electricity demand might affect political decisions on the energy system.

Most or all of the above arguments are based on reasoning and cost estimates for the different technologies. It is also possible to use a model of the energy system and investigate how it reacts to an externally induced change in electricity demand or supply. Several energy system models exist. For example, Mattsson et al. (2003) present Nelson, a dynamic optimising model of the production of electricity and district heat in the Nordic countries. They use it to investigate how the energy system reacts to a change in Nordic electricity demand or Swedish nuclear power production. Such a model gives a more complete description of the consequences of using or delivering electricity, because it takes into account effects on the utilisation of existing production facilities as well as effects on investments in new production facilities. The results from Mattsson et al. (2003) demonstrate that marginal electricity production in the Nordic countries is complex in the sense that it involves several different energy sources: natural gas, coal, wind power etc. The mix of technologies is uncertain because it depends heavily on assumptions regarding uncertain boundary conditions, future fuel prices etc. However, the results of Mattsson et al. (2003) provide some basis for recommending what data to use to model Nordic marginal electricity production:

As a first approximation, we recommend to assume that marginal electricity is produced in combined heat-and-power or power plants utilising natural gas. Natural gas CHP is an important element in most of the model results. The environmental performance of natural gas lies in between the environmental performance of coal and wind power in important aspects (notably CO_2 and NO_X emissions).

To reflect the uncertainty in marginal electricity production, this guideline recommends the use of two extreme scenarios, for example brown coal and nuclear power. These technologies are likely to have very high and very low impact respectively, at least in terms of the environmental effects accounted for in an LCA and CBA. Brown coal power would be the marginal technology in a scenario where a stagnation in electricity demand in northern Europe makes it possible to shut down such power plants in Germany or Poland. Nuclear power would be the marginal technology in a scenario where a change in electricity demand affects political decisions to close such power plants in Sweden or Germany.

4.2.4 Alternative heat production

Incineration is a widely used technology for waste management in Sweden and Denmark. Through waste incineration, energy in the waste is utilised for production of district heat and, to some extent, electricity. This energy recovery reduces the demand for other energy sources. The environmental benefits of waste incineration depend heavily on what energy sources are replaced, (Ekvall 1999), and this depends on the time perspective of the study (Ekvall & Finnveden 2000).

Waste incineration plants require heavy investments (see for example Table 4.4). On the other hand, operational costs for producing heat and electricity from waste are low, because the cost of the fuel is below zero: a gate fee is paid to dispose of the waste. This makes it reasonable to assume that any installed waste incineration plant is used to its maximum capacity, as long as the district heating system can use the energy delivered. A policy or project that results in more local, regional or national waste being sent to waste incineration will not affect the utilisation of waste incineration. Instead, the short-term effect will be that local, regional or national waste replaces other waste streams, possibly waste from other countries that can be imported for incineration in the waste management system. Waste that is displaced from the incinerator is likely to end up in landfills (see Figure 4.4), because landfilling is the predominant option for waste management in several countries outside the Nordic area. Thus, the environmental benefits of sending Nordic waste to incineration are insignificant in the short-term: it is not likely to increase energy recovery from waste incineration, but it is likely to increase landfilling in other countries and the associated environmental effects.

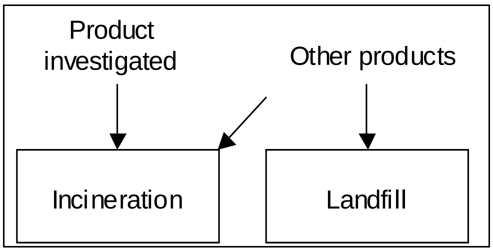

Figure 4.4 When waste incineration is restricted by incinerator capacity, the alternative fuel is other waste flows that are likely to end up at landfills when displaced from the waste incinerators

Source: Ekvall et al. (2006)

As previously stated, the long-term perspective is probably more relevant in most CBAs. The long-term effect of sending more waste to incineration can be that the capacity for waste incineration increases. This is particularly probable for countries where landfilling of combustible waste is prohibited by law.

When the capacity for waste incineration is expanded, additional energy in waste is utilised and replaces other energy sources. Electricity produced from waste replaces marginal Nordic electricity production (see Section 4.2.3). District heat will replace different energy sources in different district heating systems. In some cases, investments in waste incineration will replace investments in other technologies for baseload production of district heat, i.e, technologies with low or moderate operational costs. In other cases, investments in waste incineration will reduce the utilisation of existing plants for district heat production.

Sahlin et al. (2004) conclude that heat from Swedish waste incineration replaces mainly heat from biofuel (see Box 4.1). The effects of Danish waste incineration will be different, because the Danish heat production is predominantly based on fossil fuel. Ekvall et al. (1998) assume heat from waste incineration to replace heat from individual Danish household boilers using oil (60%) and natural gas (40%). Frees et al. (2005) assume that heat and electricity from Danish waste incineration replace heat and electricity from CHP plants with natural gas.

Box 4.1 Example of investigation into alternative heat production

A rapid expansion of waste incineration currently takes place in Sweden. Sahlin et al. (2004) use a questionnaire and a model of the Swedish district-heating systems to investigate the effects of the expansion in Swedish waste incineration that was planned five years ago. Their results indicate that expansion of waste incineration in Sweden replaces investments in district heat production from biofuel and in reduced utilisation of a mix of technologies. This investigation is valid for assessing past plans for waste incineration. It is not directly applicable for assessing projects and policies that affect future plans. Still, the results of Sahlin et al. probably represent the best available knowledge on the long-term effects of Swedish waste incineration. Based on these results, we recommend the following:

As a first approximation, assume that district heat from Swedish waste incineration replaces district heat produced from biofuel. Biofuel is the largest single energy source for production of district heat in Sweden. This means that it is used in many of the existing plants that will be affected by waste incineration. It is also expanding, and it is a baseload or midload technology, which means that investments in this technology can be expected to compete with investments in waste incineration.

As an alternative scenario or uncertainty analysis, assume that district heat from Swedish waste incineration replaces a mix of technologies that coincides with the results of Sahlin et al. (2004) that include the most fossil fuel: 62% biofuel, 11% oil, 9% natural gas, 7% industrial waste heat, 6% heat pumps, and 4% peat. The effects of any future expansion of waste incineration are likely to be in-between these two scenarios, since the use of biofuel continues to expand.

For Norway the situation is different, as Norway has a very low incineration capacity and presently is exporting waste for combustion in Sweden. A project or policy including combustion will always include expanded incineration capacity, and the marginal energy source at a national scale will be electricity[10] (but at a local or regional scale it can be other fuels, and most likely fossil fuels), see for instance Ibenholt and Lindhjem (2003) and ECON (2002b). This example, with the very differing situations in different countries, shows the importance of a thorough assessment of the context in which the policy or project are being implemented.

4.2.5 Marginal fuel

The previous section dealt with energy replaced by waste incineration. However, it did not account for the fact that fuel displaced by waste in a district heating system might, in turn, replace another fuel elsewhere in the energy system. Biofuel that is not used to produce district heat in Sweden might, for instance, replace part of the coal in Danish or German

[10] Electricity is the main heating source in Norway, see www.ssb.no

power plants. If the demand for fuel in marginal electricity production is reduced, this fuel might also be used elsewhere.

It is reasonable to assume that fuel that is ultimately replaced in the energy system is the marginal fuel, i.e., the most expensive fuel. In a similar way, any fuel use in the waste management system can be assumed to affect the production of marginal fuel. Unfortunately, the identification of marginal fuel is difficult.

Frees et al. (2005) argue that marginal fuel in Sweden is fossil fuel and assume it to be 50% oil and 50% natural gas. Gustavsson et al. (2006) also assume that it is a fossil fuel, with the uncertainty ranging from natural gas to coal. The basis for this assumption is the argument that all available biofuel will be used in the effort to reduce climate change. Most biofuel is also byproducts from forestry and agriculture. This means that the production of biofuel does not mainly depend on the demand for biofuel but on the demand for other products from forests and farms. If the production of biofuel is inflexible, and all available biofuel is needed to fight climate change, the only alternative available to cover an increase in fuel demand is fossil fuel. Hence, the marginal fuel is a fossil fuel, and any energy recovered from waste will reduce the extraction and use of fossil fuel.

However, a discussion of the policy against climate change can also result in the opposite conclusion. As a result of the Kyoto protocol, the Nordic countries and the EU have accepted a limit on their emissions of fossil CO_2. This limit will decide the level of fossil CO_2 emissions and restrict the use of fossil fuel. If the policy or project results in increased CO_2 emissions in the waste management sector, the CO_2 emissions need to be reduced in another sector or, through flexible mechanisms, in another country. There will be no net change in total CO_2 emissions, only a change in the costs required to meet the CO_2 target. Unless CO_2 sequestration and storage is available, the same is likely to hold for fossil fuel: the use of fossil fuel will not be affected by marginal changes in the fuel demand. Instead, any change in fuel demand will be met by changes in the production of biofuel.

And the production of biofuel can, to some extent, adapt to changes in demand. The extraction of stamps and boughs after forest cuttings can be affected by the price of biofuel. Forestry machines are frequently reprogrammed to change the share of each tree that becomes sawmill logs, pulpwood, and fuel. The production of biofuel from forests can also change fairly rapidly through changes in thinning activities. In the mid-term perspective, production of biofuel can increase through the establishment of energy plantations. In the very long-term, the production of biofuel from forests can increase significantly through fertilisation and other measures to increase forest growth (Ivarsson 2004). This discussion indicates that biofuel is the marginal fuel and that any energy recovered from waste will reduce the production and use of biofuel.

Studies of marginal fuel can be made using energy system models such as MARKAL-Nordic (Unger 2003), RAMSES (DEA 2006), or BALMOREL (2001). Such models can give detailed insights into how the energy system might react to a change in fuel consumption. However, the results from such models will depend strongly on the boundary conditions and input data, and these will, in turn, depend on assumptions and argumentation similar to the discussion above.

It is possible that further analysis and discussion can clarify the issue regarding what the marginal fuel is. Until then, a CBA does not gain from making simple assumptions on how the fuel markets are affected by the waste management system. Most LCAs exclude this third-order effect, and for the time being it is recommended to exclude it also from CBAs. If so, the CBA will include only the production and use of fuels that are actually used in the waste management system and the production and use of fuels that are directly replaced by energy from the waste management system.

4.2.6 Time horizon of landfill emissions

Emissions from most processes occur immediately when the process is active. Deposition at landfill is different in that the emissions from landfills occur over many years after the deposition. Figure 4.5 illustrates how methane emissions from biodegradable waste that is deposited at landfill in year 1 continues for several decades, although at an exponentially decreasing rate. In this simplified illustration, the decay rate varies from material to material; where the methane emission from food waste is halved in four years, the emission from wood is halved in 23 years (Skovgaard et al. 2006). Methane emissions do not begin when the waste is deposited, but some time afterwards (Finnveden et al. 1996; ECON 2000). The actual rate of emissions will also depend on the local conditions in the landfill: waste composition, precipitation, temperature, etc.

Emissions from landfill to water also occur over several years. Finnveden (1996) estimates that metals which are landfilled will eventually be fully emitted through leachate from the landfill. If 1 kg of lead is deposited at the landfill, 1 kg of lead is eventually emitted; however, the environmental impact of metals emissions depends not only on the quantity emitted but also on the concentration of the pollutant. A slow leaching process is likely to have much less impact than a sudden emission of 1 kg of lead.

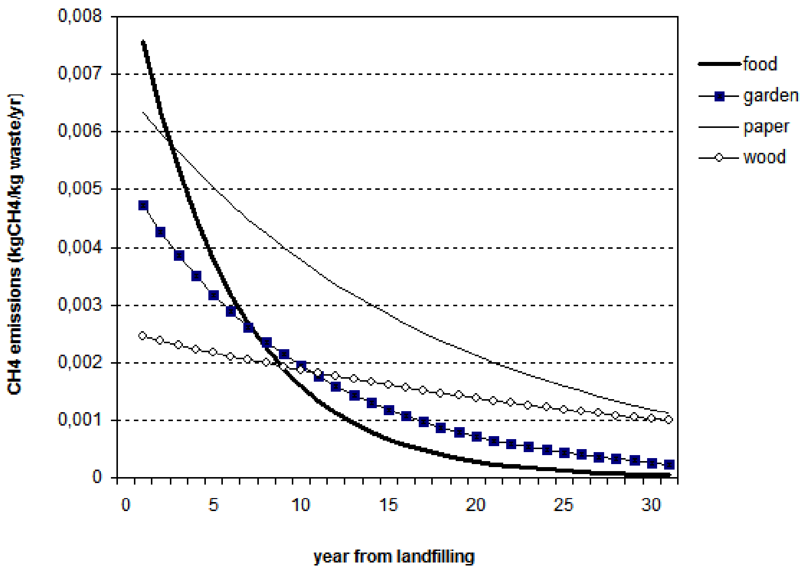

Figure 4.5 Example of methane emissions evolution over time using a first-order decay model

Note: The degradation of 1kg of different waste materials is presented, each material having a specific organic content and degradation rate (represented by the half-life degradation times, which are food: 4 years, garden waste: 7 years, paper waste: 12 years, wood: 23 years.

Source: Skovgaard et al. (2006)

Should the CBA account for all emissions from the deposited waste, regardless of how far into the future the emissions occur? In his handbook on LCA, Guinée (2002) recommends LCA practitioners to distinguish between controlled and uncontrolled landfills. Uncontrolled landfills are not an option for Nordic waste, but uncontrolled landfills in other countries might be included in the CBA if it accounts for waste flows across national borders. For waste deposited at controlled landfills, Guinée (2002) recommends that an LCA should include emissions of organic substances during the first 200 years. All emissions of inorganic substances, such as metals, should be included regardless of when they occur. If waste is deposited at uncontrolled landfills, Guinée recommends that no emissions from the landfill are included in the LCA. Instead the waste as such should be recorded as an emission.

If this method is used in a CBA there is a risk that the monetisation of waste as an emission does not fully reflect the impact of the substances leaching from the waste. An uncontrolled landfill typically causes much more severe leachate and air emissions compared to a controlled landfill. If the monetisation does not fully account for this impact, the results might wrongfully indicate that deposition in an uncontrolled landfill is environmentally superior to a controlled landfill. To avoid this risk, it is recommended that the CBA practitioner uses the same method for all landfills, and accounts for the actual leachates into the natural surroundings[11]. One possibility is to include organic emissions during the first 200 years and all inorganic emissions whenever they occur.

[11] Leachates being caught in a controlled landfill and treated in a secure manner should not be included in the CBA.

5. Monetary valuation

In principle, all the effects identified in the inventory should be assigned a monetary value. In this chapter it is discussed how to estimate the social costs and benefits of a project. The chapter is divided into two sections: in the first section valuation of marketed goods and services, i.e. goods that are traded in a market and hence have a market price, are discussed. The second section discusses valuation of goods that are normally not traded in a market, and where a market price does not exists. The latter includes environmental benefits.

5.1 Marketed goods and services

The project or policy being analysed will induce investment/capital and operational costs (both fixed and variable), as well as generate income from produced goods or services (for instance energy or recycled materials). These are costs and receipts that are normally included in a financial analysis of private costs for a project or investment plan. These resources should also be included in a CBA, but not necessarily in the same manner and with the same costs as in a private analysis. As stated in Chapter 2, CBA is supposed to account for the social costs and benefits, which is why private costs usually have to be adjusted in order to show the social costs. The social benefit is expressed as the affected people's willingness-to-pay for these benefits whereas the costs are expressed as the opportunity costs, i.e., the value the resources used would have received in the best alternative use.

Some guidelines recommend that the analysis of social costs starts with a straightforward financial analysis, and that the market prices in this analysis thereafter are converted to *shadow prices,* i.e. the price that reflects the opportunity cost, e.g. CEC (2003b). Other guidelines recommend that the effects are expressed in physical units (e.g. km, m^3 or man-hours) and that these effects are directly valued at their social costs (e.g. Norwegian Ministry of Finance 2005). What method to choose is partly a question about available data and the resources set aside for the CBA. In many cases it is likely that a financial analysis of the policy or project in question already exists, and that this can be used as the basis for the social cost analysis.

5.1.1 Private costs

Private costs are the costs for use of marketed resources in all the stages identified in the process inventory. For a waste management policy or project this might typically include:

1. Households: costs for water and energy use for cleaning the waste streams, and the cost of storage systems. In case of a bring system the households may have costs for transporting the waste to bring banks or civic recycling centres. The calculation of private costs should not include the leisure time spent for handling the waste or the costs of the space used (see Section 5.1.9). Moreover, it should not include the waste fees paid to the actor collecting and treating the waste, which is a transfer within the waste management system (see below).
2. Collection of the waste stream: includes transport and labour costs, and investment in and maintenance of containers and transport equipment.
3. Treatment of the waste stream: includes labour and other operational costs, as well as investments in the treatment plant and facilities.
4. Income from the recycled fraction or the energy produced.
5. Other costs: this might be costs for the owner of a building, if this is not the same as the household (for instance investments in storage system, costs for maintenance of the waste area (e.g. bring bank area or civic recycling centre), and public administration costs of the system (not included in stages 2 or 3).

5.1.2 Distinction between real costs and transfers

It is important to distinguish between the real economic effects and pure transfers of money. Pure transfers have no net effect in a social cost analysis, and should therefore not be included in the calculation. For example, person A giving EUR 100 to person B, meaning that B has a gain of EUR 100 while A loses the same amount, will not change the net benefits for society as a whole. The same applies if person A pays EUR 100 to the municipality for the waste management services. Waste management fees, or user charges, are not real social costs as such, but they can of course affect the behaviour of the actors (households, companies, etc.) and thereby cause real economic effects (they can for instance affect the amount of waste managed through the municipal system). Hence, the waste fee or user charge should not be included in the CBA calculation, but it could be noted in a "side account" in order to identify distributional effects of the policy or project being analysed.

5.1.3 Calculating shadow prices

As mentioned above social costs and benefits are expressed in shadow prices. For a good or resource that is traded in a well functioning market, i.e. a market without distortions, the shadow price is equal to the market price. The price of the resource in such a market is determined by equilibrium between demand and supply, i.e., where the willingness-to-pay for one extra unit of the resource is equal to the cost of producing this unit.

Markets can be distorted by external effects[12], taxes, imperfect competition, and disequilibrium like unemployment. Some of these effects are discussed below, along with some other important considerations when calculating social costs.

In distorted markets the market price does not reflect the social cost or benefits. Therefore, the market price has to be corrected in order to estimate the shadow price. For a more thorough discussion of social costs in a CBA, see for example US EPA (2000), Norwegian Ministry of Finance (1997), Danish Ministry of Finance (1999) and Londero (2003).

It is, however, important to be aware that it may be difficult to correct the market price so that it reflects social costs. It may thus be appropriate to perform this correction only if the market distortions are believed to be of importance (Norwegian Ministry of Finance 2005).

5.1 4 Shadow prices and taxes

Choosing what shadow price to use for taxed input factors is not straightforward in a CBA, and this issue or problem is treated differently in various guidelines as shown in Table 5.1. Taxes on input factors and other goods create a wedge between the willingness-to-pay and the opportunity cost for these products, and this wedge should be accounted for in a CBA.

The Norwegian Ministry of Finance recommends different approaches to correct for this tax wedge, based on whether the production activities take place in a sheltered sector or a sector exposed to competition. For public production exposed to competition from private production the Norwegian guidelines recommend that the prices used for input factors in the private production are also used in public production (that is inclusive the same taxes that are used in calculations for private production). For public monopolies, i.e., where there is no competition from private producers, the price for input factors should be exclusive of all taxes and duties.

The Danish Ministry of Finance recommends using a method called the accounting price method to correct for the tax wedge. Here, a so-called *net-duty factor* [13] is used to reflect the difference between market

[12] The main external effect in a CBA for waste management is environmental costs and benefits. These goods are normally not traded in an ordinary market, and the social costs and benefits will have to be estimated by using other methods. This issue is discussed in Chapter 5.2.

[13] This net-duty factor is calculated as the proportion between gross domestic product (GDP) at market prices (measured as the sum of all private and government consumption of goods and services, private domestic investments and the difference between exports and imports of goods and services) and gross domestic income (GDP at factor prices), measured as the sum of income earned by those providing the factors of production (i.e. wages, salaries, rent, interest and profit. See Møller et al. (2000).

prices (including all taxes and duties) and factor prices (market price of input factors exclusive of reimbursed taxes and duties).[14] For Denmark this net-duty factor is equal to 1.17 for domestically traded goods, and 1.25 for internationally traded goods (Møller et al. 2000).

In Norwegian Ministry of Finance (1997) the issue of using a weight such as the net-duty factor is discussed, but the guideline recommends, partly due to the need for simple rules, using the same prices in the public and private sector as the shadow price is a sufficient estimate.

Table 5.1 Shadow prices and taxes

		Public monopoly	Competitive production
Labour	Market wage incl. taxes and employers' contribution * net-duty factor	Market wage incl. taxes and employers' contribution	Market wage incl. taxes and employers' contribution
Input factors	Market price excl. reimbursed taxes * net-duty factor	Price excl. taxes, but incl. taxes correcting for external effects	Price incl. the same taxes as in the private sector

Source: Møller et al. (2000) and Norwegian Ministry of Finance (1997)

In 1999, the Swedish Institute for Transport and Communications Analysis (SIKA 1999) issued a guide for cost-benefit analyses in the transport sector. This guide - which was made after a large project (ASEK) concerning socio-economic analytical methods in which also the Swedish Ministry of Finance was involved – recommends using a tax factor that "takes into consideration that resources that are taken into use have a value determined by the amount, which the consumers are willing to pay for the lost consumption goods". The tax factor is set at the value of 1.23. Thus, the factor is added to the costs in factor prices in order to consider the value (expressed as willingness-to-pay for consumer goods) that the production factors alternatively could have created. This tax factor is more or less equivalent to the net-duty factor in Møller et al. (2000) and COWI (2002).

A general recommendation is to use the same method as other CBAs in the country the analysis is performed for (e.g. as presented here for Norway, Denmark and Sweden). If no national guideline exists the method recommended by the Norwegian Ministry of Finance is probably the "simplest" alternative. Using different approaches in different countries will make comparison of CBAs between countries more difficult, but since a CBA normally has a national perspective and is being used to compare different national options or policies it would probably be more erroneous to use different methods for CBAs within in one country, than using different methods in different countries.

As an example, calculation of the shadow price according to the Danish guideline, Møller et al. (2000), is shown in Table 5.2. The example is shown for incineration and is based on the inventory in Table 4.3.

[14] Gross domestic income is equal to gross domestic product minus all taxes plus subsidies paid at all levels of production.

Table 5.2 Example: calculation of the shadow price for incineration in Denmark (excluding environmental effects)

	Resource use		Unit price	Net-duty factor	Shadow price	
			DKK/unit		DKK/unit	Mill DKK
Amount incinerated	483,000	Tonnes				
Capital cost						
Investment incineration plant	2 bn	DKK	0.0874*	1.21	0.1	211.58
- lifetime	25	Years				
Investment machinery	700 mill	DKK	0.0733*	1.21	0.085	62.05
- lifetime	40	Years				
Total capital cost						273.62
Operational cost						
Labour						
- staff	149	Men	315.436	1.17	369.060	54.99
- pensions	3,000,000	DKK	1	1.17	1.17	3.51
Raw materials						
- Lime	3,000	Tonnes	333	1.17	390	1.17
- Chemicals	2,000,000	DKK	1	1.25	1.25	2.50
- Lye	390	Tonnes	1,282	1.25	1,603	0.63
Energy						
- Electricity	42,300	MWh	240	1.25	300	12.69
- Natural gas	256,000	m^3	1.95	1.25	2.44	0.63
- Oil	526	m^3	950.52	1.25	1,211	0.63
- Peak load	1,000,000	DKK	1	1.25	1.25	1.25
Other						
- environmental control	13,000,000	DKK	1	1.17		15.21
- administration	9,000,000	DKK	1	1.17		10.53
- maintenance peak load	10,000,000	DKK	1	1.17		11.70
- R & D	2,000,000	DKK	1	1.17		2.34
- pre-treatment and transport	8,000,000	DKK	1	1.17		9.36
- consultancy etc.	10,000,000	DKK	1	1.17		11.70
Maintenance machinery	44,000,000	DKK	1	1.17		51.48
Maintenance buildings	5,000,000	DKK	1	1.17		5.85
Total operational cost						196.15
Management of residues						
Landfill	15,000	Tonnes	445	1.17	521	7.82
Recycling	107,000	Tonnes	158	1.17	185	19.80
Total management of residues						27.61
Total cost, mill DKK						497.39
Total cost per tonne incinerated, DKK						1,030
Revenue from production						
Production of electricity	115,300	MWh	234	1.25	293	33.73
Production of heat	953,000	MWh	169	1.17	198	188.44
Total revenue						222.16
Net cost incineration, total						275.23
Net cost incineration, DKK per tonne						570

Note: * The unit price of capital is the amortization factor multiplied with a factor that represents the alternative rate of return on investment, Møller (2001). For further explanation see Strandmark et al. (2002a, 36 and 71ff). The net-duty factor of 1.21 is an average of the factor for domestically produced goods (1.17) and factor for imported goods (1.25).
Source: Damgaard (2003) and Strandmark et al. (2002a)

5.1.5 Internalising taxes

The section above holds for taxes that are being used to generate revenue for the authorities and that create a wedge (or market distortion) between the price the buyer of a good or service pays and the income the producer of this good or service receives. For environmental taxes that are implemented in order to internalise an external effect (that is, to correct for a market distortion) there should be no tax wedge correction, i.e., there is no need for using a net-duty factor or similar as described above.

If these environmental taxes are believed to be a good estimate of the actual damage, they can be used as a proxy for the environmental cost, but then estimates for these external effects should not be included in the separate analysis of external effects, as this would be to double-count for the environmental damages. This can be illustrated with the Norwegian charge for emission to air from combustion of waste: in a Norwegian CBA there is a choice between either including this charge in the cost for combustion, or excluding the charge and use other estimates for the external costs from the emission to air, but one cannot use both the charge and have a separate estimate for external costs.

5.1.6 Imperfect competition

In a market with imperfect competition, e.g. a monopoly, the market price will not reflect the true opportunity cost. Hence, it could be argued that the market prices should be adjusted for the monopoly profit. However, it can be difficult to identify and quantify the monopoly profit, and as a result most guidelines recommend using the market price as the shadow price. An additional argument for using the market price is that the government should not, as a principle, reward monopoly profit by using a lower shadow price thereby generating a larger demand than the market would have done (Danish Ministry of Finance 1999).

5.1.7 Labour and unemployment

In general, the opportunity costs for labour should be the market wage, inclusive taxes and employers' contribution. However, for some projects or policies it could be argued that the alternative for some, or all, employed is unemployment. For these people the market wage is not the true social cost, but rather the so-called reservation wage which is equal to the value of (leisure) time the unemployed has to give up. In most instances it can be difficult to see whether the alternative for the people who are actually employed in the project is unemployment. Most guidelines therefore recommend that the market wage is used as the shadow price for labour, and that this principle is only adjusted if the project or policy is especially targeted towards long-term unemployed or certain geographical areas with high unemployment rates.

5.1.8 Financing projects and policy (marginal cost of public funds)

How the policy or public project is financed can have consequences for the CBA. If the project results in a marketable product, then the income from this can be enough to finance the project (i.e., that the project is privately profitable), but for many public projects this is not relevant or feasible. A project can be socially profitable but still give a large financial deficit. Whether the public sector finances this deficit by user charges or general taxes is of importance. Both user charges and general taxes can give an efficiency loss (deadweight costs or marginal cost of public funds, MCPF), because they create a wedge between private and social costs.[15] The tax cost for using general taxes to finance the policy/project will probably differ between countries but not between projects/policies in the same country. For Norway the tax cost is estimated to be 20%, and this figure is also applied by the Danish Ministry of Finance and the Government Institute for Economic Research in Finland. SIKA estimates a (second) tax factor for Sweden[16], which "takes into consideration that the increase in the tax revenues marginally gives rise to losses in welfare e.g. through individuals not working in their most effective occupations". This factor has a value of 1.30, see SIKA (1999).

User charges that do not reflect the user costs, i.e., where there is no direct link between the costs the user actually inflicts and the charge the user has to pay, will also create a wedge between social and private costs, that is a deadweight cost.[17] This deadweight cost is project specific, and must be calculated separately for each project.

User charges, in the form of waste fees, are used to finance many waste policies. For most purposes it can be assumed that user charges are set to reflect the actual costs, and hence there is no need to adjust for the tax costs. If this for some reason should not be the case, the costs financed through the public budget should be "charged" with a tax cost.

5.1.9 Valuation of householders' time and space

Time

There is no consensus on how to monetise the time spent on source separation in households. One method is to set the value of the leisure time of citizens equal to the full salary cost, including tax. The salary cost reflects

---

[15] A simple example of this efficiency loss is if person A is willing to perform a service for person B for EUR 100 and B is willing to pay EUR 110 for this service, then both will benefit from this transaction. But if A has a marginal tax equal to 50% he will only get EUR 55 of the EUR 110 person B is willing to pay, and thus the transaction will not take place and the potential gain of EUR 10 will not be realised.

[16] In addition to the tax factor of 1.23 mentioned in section on shadow prices and taxes.

[17] A commonly used example is a toll system for financing a road, and where the toll is independent of actual traffic and congestions. If there are no congestions the marginal cost for an extra vehicle is more or less negligible, and hence there will be no connection between the price the driver has to pay (the toll) and the actual cost he inflicts. The road will be used less than what is optimal, resulting in a social loss. The toll also inflicts administrative costs.

the value of the time for employers: this is what the marginal employer is prepared to pay for the time of citizens (Brännlund 2001). Another method is to set the value of the leisure time equal to the net salary, excluding tax. The net salary can be described as reflecting the value of the time for employees: this is what the marginal employee is prepared to accept working for. However, both of these methods are based on the assumption that each hour spent on household chores means one hour less is available on the employment market. This is not true for students, retired people and unemployed. It is probably also not true for most citizens working full time.

A different approach is to estimate how much it would be worth for consumers to avoid spending time on source separation. Here, time is treated as a non-marketed good and the value of the time is identified through WTP methods. Radetzki (1999) assumed that the WTP to avoid spending time on source separation is similar to the WTP to avoid household duties in general. This, in turn, is set to nearly EUR 7 per hour, based on an official estimate of what is paid for assistance for household duties.

It is reasonable for two reasons to assume that the WTP to avoid source separation on average is lower than the estimate by Radetzki:

- the minority that pays for such assistance is likely to have less time and more money than the average population, and
- source separation is perceived as more meaningful than other household duties (Dengsøe 2003).

Bruvoll et al. (2002) interviewed 1,162 Norwegian residents to investigate the time required and the WTP to have someone else separating the waste streams. Responses varied greatly between individuals: 27% preferred to separate the waste themselves, even if the cost for having someone else do it was zero; 35% had a WTP equal to zero; 6% had a WTP over USD 111 per year. The average response was USD 20 per year or 39 US cents per hour, which is considerably lower than the estimate by Radetzki.

The response of the 27% who preferred to separate the waste themselves was set at zero when Bruvoll et al. calculated the average result of USD 20 per year. It seems reasonable to interpret a stated preference to separate the waste as a negative WTP rather than a WTP equal to zero. This indicates that the average WTP is actually less than USD 20 per year.

The average result of 39 US cents per hour was calculated using the annual WTP and the estimated annual time spent on source separation. Both of these figures are likely to be overestimated (see previous paragraph and Section 4.1.3) and the uncertainty of the result is large.

Berglund (2006) did a study similar to Bruvoll et al. by sending a questionnaire to 850 residents in Piteå in Sweden. The results were also quite consistent with the study by Bruvoll et al.: the average response was USD 25 per year or 38 US cents per hour. The risks of overestimations were similar in this study, compared to the study by Bruvoll et al. The fact that only one third of the 850 residents stated their WTP also contributes to the uncertainty in the results of Berglund.

Based on the results presented by Bruvoll et al. and by Berglund, a best estimate for the average WTP to avoid source separation is approximately EUR 0.3 per hour. As should be clear from the above, the uncertainty in this estimate is large.

Since the WTP varies greatly between different individuals (Bruvoll et al. 2002), it is reasonable to use different time-costs in a CBA depending on the purpose of the study. The average WTP of 0.3 Euro per hour might be a valid value in a study designed to assess the costs and benefits of the current source separation schemes as a whole.

If the purpose is to assess a moderate increase or reduction in the separation and recycling target, a higher estimate of the external cost is reasonable. A change in the recycling target would probably have little effect on the source separation of the most motivated individuals. Instead, it would affect persons that are slightly more reluctant to participate in the source separation schemes.

If the CBA aims at assessing a radical increase in the collection and recycling rates, the WTP increases greatly. A radical increase in the source separation rate requires the involvement of individuals and households with little interest in the environment and/or waste management. As illustrated by the high-end results from Bruvoll et al. (2002), the cost for the time those persons spend on source separation can be very high.

A third approach is to account for the marketed as well as the non-marketed aspects of time. This requires information on what share of the source separation time would be spent at work. No such data are easily available. However, less than half of the Nordic population is currently employed - most of the remainder are children, students, or retired people. In the light of the large uncertainty, Ekvall & Bäckman (2001) assumed that 3–30% of the time spent on source separation would otherwise be spent at work, and that the average salary cost is approximately EUR 20 per hour. Based on these assumptions, the cost of the marketed share of source separation time can be calculated to be EUR 0.6–6 per hour.

Given that 3–30% of source separation time would otherwise be spent at work, 70–97% of the time is a reduction in the leisure time available for other activities. The non-marketed part of the time cost stems from this reduction.

Using the non-marketed cost estimate of EUR 0.3 per hour (see above) and the marketed cost of EUR 0.6–6 per hour, the average total cost for the time currently spent on source separation is EUR 0.9–6 per hour, where most of the cost and uncertainty lies in the assumption that 3–30% of the time spent on source separation would otherwise be spent at work.

Space in households

When storage space in kitchens is a limited resource, the space required for storing separate waste streams may be considered an external cost (see Section 4.1.4). For example, Ekvall & Bäckman (2001) assume the external value of floor space to be equal to the actual cost and assumes this to be SEK 800 (EUR 85) per m^2 and year for Sweden. Given the assumptions that storage requires 0.06 m^2 per waste stream and intrudes on the floor space in 20% of households, the annual cost of the floor space is: 0.2 * 0.06 * 85 = 1 EUR per waste stream per household.

Ekvall & Bäckman (2001) estimated the uncertainty in this value to be of a factor 2. Figures that are more up to date, accurate and/or valid for other Nordic countries might be available from other sources.

5.2 Valuation of environmental benefits and costs

External effects consist of several types of effects: emissions and discharges to air and water, noise, traffic congestion, disamenity, etc.

The environmental effects caused by policies cannot readily be measured and compared to effects which are directly valued in markets. Economists try to value non-market benefits, such as environmental benefits and costs, in order to be able to compare all effects of a project. Even if the valuation of benefit and cost streams of certain policies is often fraught with methodological and practical difficulties, it is usually better to include an uncertain estimation rather than leaving it out of the CBA altogether. A decision made without a measure of the value of environmental effects, will *implicitly* assign a value to these effects. Therefore, most CBA guidelines recommend that environmental costs and benefits should be included explicitly in the CBA, and as far as possible measured in monetary terms (e.g. US EPA 2000, Norwegian Ministry of Finance 1997, 2005).

The major environmental effects of waste management, from generation to the point where the waste is recycled, incinerated or landfilled, are emissions and discharges to air and water from the different waste treatment options and emissions to air from transport. The environmental effects include effects on health and ecosystems. Waste may also be considered a resource, as recycling and incineration will generate an output (materials or energy) that can substitute virgin raw materials and fossil energy. The environmental gain herefrom should also be included in the CBA as discussed in Section 4.2. Emissions and discharges are assumed to be the most important effects from waste management, which is why it has been decided to concentrate on these external effects in this guideline.

5.2.1 Total Economic Value

The concept of Total Economic Value (TEV) is a conceptual framework for keeping track of the wide range of complex and interrelated physical and value flows involved in valuing the natural environment, see also Eftec/DEFRA (2006). It reflects the use humans make of the natural environment, that is use value, and the value humans may attribute to it unrelated to their current or future use, the so called non-use value.

Use value involves some interaction with the resource, either directly or indirectly. *Direct use value* reflects the use of a resource in either a consumptive way (e.g. the fishing industry and agriculture) or a non-consumptive way (e.g. rambling). *Indirect use value* reflects the benefits individuals gain from ecosystem services supported by a resource rather than actually using it (e.g. watershed protection for flood mitigation, cycling processes for agriculture or carbon sequestration).

Non-use value refers to benefits arising from the knowledge that the natural environment is maintained. By definition, non-use value is not associated with any use of the resource or tangible benefit derived from it, although users of a resource might also attribute a non-use value to it. Non-use value can be split into three basic components; *Altruistic value* which reflects the knowledge that contemporaries can enjoy the goods and services the natural environment provides; *Bequest value* reflects the knowledge that the natural environment will be passed on to future generations; and *Existence value* which reflects the satisfaction of knowing that ecosystems continue to exist, regardless of any use made of them by oneself or others now or in future. The latter component is associated with the environment's "intrinsic value".

A third category, not immediately associated with use or non-use value is the *option value*. This value reflects the benefit an individual derives from having the option to make use of some aspect of the natural environment in the future, even though he or she does not currently plan to make such use. *Quasi-option value* is a related value associated with avoiding or delaying irreversible decisions, where technological and knowledge improvements can alter the optimal management of a natural resource. It is particularly relevant to the precautionary principle. A common example is the potential for genetic information in biodiversity to be used for creating pharmaceuticals or improved crop varieties.

5.2.2 Methods for valuation

The main principle when valuing environmental costs or benefits is people's willingness-to-pay (WTP) for avoiding these costs or gaining the benefits. This principle is based on the thesis that WTP expresses the social value of any good or service, see Section 5.1 and Chapter 2.

There is a large amount of literature and a significant number of case studies dealing with various monetary valuation methods. For comprehensive coverage and more detailed discussion of how to use the methodologies see for instance Pearce & Markandaya (1989), Pearce (1993), Constanza, et al. (1997), Garrod & Willis (1999), Freeman (2003) and Eftec/DEFRA (2006).

Table 5.3 Different methods for valuation of environmental effects

Demand curve approaches: Revealed preference	Hedonic pricing Travel costs (Avertive behaviour, Defensive expenditure)
Stated preferences	Contingent valuation Conjoint analysis (Choice experiments, Contingent ranking/rating)
Market valuation of physical effects	Dose-response approach Replacement cost method (Human capital approach)
Non-demand curve approaches	Clean-up cost Avoidance cost Abatement cost (Linked environmental values)

Source: Cowi (2000)

The different methods for valuation of environmental effects can be categorised according to different criteria. In this guideline the categorisation in COWI (2000) is applied and presented in Table 5.3. Below is given a short description of these methods.

The various methods will often give different results. This reflects the fact that they measure different value components (use value, non-use value etc.). It also indicates that estimates of environmental value (benefits or costs) are uncertain. This uncertainty can be caused by lack of knowledge about short and/or long-term negative effects, and difficulties in assessing how people and authorities value these effects or how much they are willing to pay to avoid negative effects. The costs are also likely to change over time due to changes in willingness-to-pay. It is plausible that increased income and increased scarcity of environmental goods will give higher WTP for the environment in the future, and hence the WTP ought to be adjusted over the time horizon of the project or policy studied. The cost of an effect, such as a certain emission, can be dependent on either the size of the actual effect or the “background” values for the emission. This is due to the existence of threshold values where emissions over a certain level are more harmful than below this threshold.

5.2.3 Revealed preferences

Hedonic pricing methods

The hedonic pricing method estimates the value of a non-market good by examining the relationship between the non-market good and the demand for some market priced complementary good (Freeman 1993). The most commonly used method is the hedonic pricing property approach, which is based on the fact that the price of a property is determined, in part, by the specific characteristics of the property's structure, location and environs. It is reasonable to assume that environmental goods and services such as landscape amenity, noise and air quality may be included in these characteristics. A property which features higher levels of desirable environmental characteristics will presumably command a higher market price than a similar property with lower levels of those same characteristics.

The method estimates the direct use value component of total economic value, and it is grounded firmly in the principles of economic theory, relying on the derivation of demand curves and elasticity estimates. It provides a basis for estimating the value of a wide range of environmental characteristics, especially on localised and site specific impacts, e.g. the effect of road traffic noise, or the impact of waste facilities etc. However, the method is less applicable to environmental goods/bads which are not typically perceived by the buyer, such as chemical hazard, radiation, etc.

An advantage of the hedonic pricing method, and revealed preference techniques as such, is the use of actual market data. The main disadvantage is the requirement for large amounts of data and specialist econometric expertise.

Travel costs

The travel cost method uses the cost incurred by individuals travelling to reach a site, and the costs of staying at the site (for a certain amount of time), as a proxy for the recreational value of that site. The method estimates the direct (often non-consumptive) use value component of total economic value.

Travel costs include both actual travel expenditure (e.g. petrol, fares, accommodation, food, etc) and the value of time, valued at the opportunity cost for this time (see the discussion about valuation of time use in Section 5.4). The collection of data is done through surveys among visitors to a site. In addition to collecting appropriate data, the method also requires econometric expertise, and it can take a long time to perform, especially if data have to be collected over a whole year in order to account for seasonal variations in visitor patterns and number.

The method provides estimates of the use value derived from well-defined recreational sites, or separable, well-perceived environmental or cultural attributes within such a site. For typical waste projects/policies this kind of environmental good is not of central importance.

5.2.4 Stated preferences

Contingent valuation (CV)

The contingent valuation method is a survey-based approach to valuing non-market environmental goods and services. The approach entails the construction of a hypothetical, or "simulated", market via a questionnaire methodology where respondents answer questions concerning what they are willing to pay - or willing to accept (WTA) as compensation - for a specified environmental change.

The method can be used to estimate the total economic value of an environmental good or service, i.e. both use value and non-use value components, or the economic value held by users and non-users separately. The method is based on the consumer demand theory which explains the factors determining demand, in this case, for environmental goods and services.

Reliable CV studies are both complex and resource demanding, and proper practice requires time to develop the survey instrument and to ensure that the non-market good or service to be valued is clearly explained along with the constructed market and payment method.

The principal output from contingent valuation studies is estimates of WTP/WTA for changes in the provision of non-market goods and services. These estimates of value are consistent with measures of welfare economics, the underlying basis of cost-benefit analysis. The CV approach enables the total economic value of an environmental good or service to be valued, i.e., including both use value and non-use value. The method is flexible and facilitates the valuation of a wide range of environmental goods and services. However, the method can be problematic for goods that are not well-known and have long-term and uncertain impacts. Practical examples of application of the method are numerous, and a bibliography of contingent valuation studies in 1994 revealed almost 2000 separate studies, (Carson et al. 1995). A study by Horowitz and McConnell (2002) finds that estimates of WTP and WTA for the same good often diverges, and that WTA is generally higher than WTP. In Pearce et al. (2006) different explanations for this divergence are discussed, and the conclusion is that if the difference only is due to the questionnaire design WTP can be used, but if the difference is due to some legitimate reason (actually reflects different values) both WTP and WTA ought to be calculated.

Stated preference techniques (including CV) are the only approaches that can estimate non-use value associated with environmental goods and services.

Choice experiments

An alternative to contingent valuation is the more indirect methods where the respondents are asked to choose between different hypothetical, but at the same time realistic, alternatives. The most commonly used methods are conjoint analysis and contingent ranking. The difference between these methods is that in a conjoint analysis the respondent has to choose an alternative, whereas the alternatives should be ranked against each other in contingent ranking.

The main advantage of choice experiments as compared to contingent valuation is that one avoids direct questions about payment, which for some respondents will be an unfamiliar and sensitive theme. People are often more accustomed to choice situations, especially if these are in accordance with choices the individuals have had to make earlier. On the other hand the main disadvantage is that the valuation is indirect, and that the respondents make no conscious monetary valuation and this can in fact conceal more than it clarifies. For a description and discussion of this method, see Adamowics et al. (1997) and Bateman and Willis (1999).

5.2.5 Market valuation of physical effects

Market valuation methods are, as the name indicates, based on market prices or shadow prices. In the literature these methods are sometimes denoted as the damage cost approach. The first method mentioned in Table 5.3, the dose-response approach, is actually not a valuation method, but it is a common method for linking actual damages with environmental values. This method is described in Box 5.1. The other methods mentioned in Table 5.3 are the replacement cost method, where the cost damage is estimated using the costs of restoring the damage, and the human capital approach, where the focus is on how air pollution affects the productivity of employees.

Box 5.1 Dose-response approach

The dose-response approach for valuing environmental costs can be used when the impacts from the pollution are known, i.e., when one knows how an extra unit of the polluting substance affects for instance human health or the eco-system. The approach establishes a link between a given pollution level (the dose) and physical impacts (the response). It is sometimes referred to as the production function approach, as it relates changes in environmental quality to changes in production relationships, (Eftec/DEFRA 2006).

The method is a straightforward way to value environmental change, as it observes physical changes and estimates how this will affect the value of goods and services. The method involves four stages:

1. identification of emissions from a given activity,
2. exposure to the receptors over time is quantified,
3. impact on the receptors is quantified (e.g., changes in output or in the risk of cancer), and
4. the economic value of the change is estimated (e.g. using market prices, the value of a statistical life, estimates of WTP or WTA).

The dose-response approach is often used to describe the relation between emissions to air and different health aspects. A dose-response approach for human health is often stated in terms of number of extra cases, for instance 'number of extra deaths– or life years lost – due to cardiovascular diseases'. In order to express these impacts in economic values, 'number of cases' must be multiplied with a cost estimate for each case. The cost of one case will normally consist both of components with a market price and components without a market price.

The process of estimating dose-response functions is exposed to many potential sources of error and uncertainty. A special problem is the fact that different emissions often vary in correlation with each other, and it can be difficult to distinguish the impacts from the different components. A related problem is that different emissions can affect each other, and that high emissions of a component can worsen the damages from another. These various effects can be captured in suitable models which are readily available, such as RAINS/GAINS and NERI's EVA-model.

5.2.6 Non-demand curve approaches

The non-demand curve approaches are not based on the fundamental principle of using individual willingness-to-pay to estimate monetary values, and give no true welfare measure but only an approximation. The approach can be to estimate clean-up costs (e.g. what it would cost to restore the damage in order to achieve the pre-damage situation), avoidance costs (what it would cost to avoid the emissions or impacts) or abatement cost (what it would cost to reduce the emissions to a certain level). Clean-up costs are based on a "total" clean-up of the environmental damages. This is not necessarily a social optimum, since there will probably be a point where the costs for further cleaning will exceed the damage costs for the current pollution level. Hence, clean-up costs tend to overestimate the environmental value.

Avoidance and abatement costs

Avoidance and abatement costs can, instead of presuming a total avoidance or abatement, be based on a politically set target for the emissions or impacts, or a standard for pollution set by the authorities. If the actual damage of an emission is unknown (but the authorities nevertheless put a charge or restriction on the emission), this charge can be interpreted as the decision-makers' valuation of reduced emissions. However, the charge or restriction depends not only on the perceived environmental cost but also on other political ends such as competitiveness, employment etc. These ends tend to call for lower charges and more lax restrictions compared to the charges that would exist if they were based on pure environmental concerns. On the other hand, environmental charges also generate funding for public spending, which means that they might be higher than the environmental value calls for. Apparently, environmental charges are uncertain estimates: they might underestimate or overestimate the environmental value significantly.

The profitability of measures to reduce or avoid emissions depends on the level of environmental charges and emission restrictions. The discussion above indicates that the marginal costs of exhaust gas cleaning etc. required by restrictions can be expected to underestimate the environmental value. The marginal costs of cleaning justified by environmental charges might be lower as well as higher than the environmental value.

The costs of implementing measures to abate or avoid an environmental damage can also be interpreted as implicit values of the environmental gains. This is an implicit valuation based on the political preferences, which can be seen as a lower-bound proxy for the inhabitants' willingness-to-pay.[18] Again, purposes other than environmental concerns can affect the decisions, making it an uncertain measure of the lower bound.

Implicit valuation is much debated, since it gives no information about the inhabitants' willingness-to-pay for the environmental benefit, but only what the authorities are *at least* willing to pay. This means, for example, that the cost of achieving different environmental standards can be the same, whereas the avoided damage might differ substantially.[19]

[18] If the parliament is willing to spend EUR X extra on highway stretch A instead of stretch B only to assure that Y individuals are not exposed to noise, then they value the fact that Y individuals avoid noise to be *at least* equal to X.

[19] Consider a situation where the costs of installing a landfill liner in order to avoid lechate, happen to equal the cost of safely containing nuclear waste. Then it might be that the costs of installing a liner are more or less equal to the damages avoided, whereas for nuclear waste the damage costs avoided by far exceed the costs of containing this waste.

5.2.7 Non-monetary valuation of environmental effects

Some environmental goods are not always desirable to monetise, either due to their complexity or ethical concerns. Typical examples are goods without use-value (like certain natural or cultural landscapes) and biodiversity. For example, issues concerning biodiversity can be of importance when assessing options for waste paper management, since paper recycling can affect forestry operations. Even if these impacts have no monetary value they should be properly dealt with in a CBA. The Norwegian guideline for CBA (Norwegian Ministry of Finance 2005) includes a method for a systematic approach to non-monetised impacts, which is presented in Box 5.2.

Box 5.2 The Norwegian method for assessing environmental effects without valuation

The Norwegian guideline for CBA (Norwegian Ministry of Finance 2005) includes a method for a systematic approach to non-monetised impacts. This method is originally compiled by the Norwegian Directorate of Public Roads, who has a long tradition for preparing CBA.

The method includes the following steps:

- Importance of the present good, in a given category ranking from low to high. The good could for instance be a recreational area with medium importance.
- Effects on the good or area caused by the project/policy, on a qualitative scale from small to large, differentiating between positive and negative impacts. There will be a deterioration in the use of the area as a recreational area, e.g. a medium negative effect.
- Consequences for the goods, where the importance and dimension/scale are judged together. The consequences, or impacts, will also be valued on a qualitative scale, from very large positive (++++) to very large negative (----). The impacts on the recreational area will be medium to large negative (--/----).

By comparing signs for the impacts on the different non-monetised goods, and if needed adjusting this for their relative importance, the total impact for these goods can be assessed. This qualitative assessment should then be compared with the monetised assessment. If the decision-makers end up choosing a different alternative than the economically most profitable, they have indirectly put a value on the non-monetised goods.

5.2.8 Benefit transfer

Valuation studies are often very resource demanding, both in financial resources and time, and thus it is unrealistic to make new estimates for every CBA performed. In practice, to save time and costs of conducting primary valuation work, value estimates from other sites (domestically or internationally) are sometimes transferred in adjusted form to the study setting, known as *benefit or value transfer* (Navrud 2004).

Benefit transfer is much used internationally, although the accuracy of this approach sometimes is questionable, and the pros and cons of transferring valuation estimates from one site to another are much debated. Questions that must be dealt with in a benefit transfer are for instance which parameters from an earlier valuation study can or should actually be transferred and how to account for differences between the earlier study and the present project (e.g. differences in culture, economic, social and environmental backgrounds). Can dose-response functions estimated in the US be used to express the connection between the levels of a certain pollution and mortality in a Nordic city? Can the WTP from a contingent valuation survey for a specific forest area be transferred to another forest area in another country?

Using benefit transfer to evaluate the environmental impacts requires expertise in making proper adjustment when transferring the coefficients to the current project/policy. There are three main methods for transferring valuation estimates from one or several studies to the actual project or policy:

- Unadjusted transfer of WTP, just using the estimated WTP from one study. This is a very simple approach, demanding that the study site and the policy site are more or less similar in several important characteristics (e.g. socio-economics, demographic, market conditions).
- Adjusted transfer of WTP, where the original WTP is adjusted for at least income differences between the sites.
- WTP function transfer, where the benefit or value function is transferred from the study site, and applied for the project site. This allows adjustment for several characteristics.

A more ambitious approach is meta-analysis of several original studies, and this can be done both for estimates and damage (dose-response) estimates, Pearce et al. (2006) and the Norwegian Ministry of Finance (2005). A meta-analysis covers many studies of WTP or damages, and uses statistical methods to analyse how the estimates vary depending on population, income level, pollution concentrations and other variables that are supposed to affect the WTP or damage estimate. Such an analysis will give an estimate of the relation between the damage or WTP estimate and the different factors that can affect these, and can make the estimates more suitable for benefit transfer. In order to reach sound estimates the meta-analysis should be based on a rather large sample of primary studies. In practice the number of primary studies is often limited, and many of the studies may be poorly documented. The more similar the present project/policy is to the primary valuation studies, the less uncertain the benefit transfer will be.

It should be noticed that sometimes a valuation estimate is used - and reused - in several studies and for many years. Many studies refer to an earlier study using this estimate. Andersen et al. (2003, 30) have traced back the estimate for CO used by the Danish Ministry of Finance (2001) to a study from 1977 for the medical evaluation of consequences to health from CO. Since then the knowledge has improved and so has the methodology for valuation.

5.2.9 Examples of valuation

Several valuation studies exist that can be used as a basis for a benefit transfer for use in a CBA. Very few of these are initially prepared for valuing environmental impacts from different waste management options. Nevertheless, many studies value the impacts of certain pollutants, for instance NO_x to air or nitrogen to water, and these estimates can also be used for emissions from waste incineration or landfilling.

COWI (2000) gives a good overview over valuation studies that have either been developed for calculating the external effects from different waste management options or have been used for this purpose. Some of the studies referred to in COWI (2000) are: Rabl et al. (1998), where the impacts on human mortality and morbidity caused by air emissions from waste incineration are valued using the dose-response method (also called impact pathway); CEC (1996), which is a benefit transfer of results from dose-response, clean-up costs, contingent valuation and averting behaviour approaches, including both health and environmental effects; and ECON (1995) which is based on Tellus (1992), but where the costs are adjusted for Norway. This study has later been updated, (ECON 2000), and in the later version several emissions are valued using dose-response for impacts on human health. The valuation of heavy metal and health damaging chemicals is based on the LCA indexes CML/RIVM (Heijungs 1992) and Eco-indicator 99 (Goedkoop & Spriensma 2000).

Other studies or projects presenting economic values for environmental damages that can be used in a waste management CBA are for instance:

- ExternE, which was a project funded under the JOULE Programme during the 1990s, where a detailed bottom-up “impact pathway” (or damage function) approach was developed to quantify external costs from energy conversion resulting from impacts on human health, crop losses, material damage and global warming. The ExternE external costs accounting framework is widely accepted and has been used to support decision-making in the field of energy and environmental policy, e.g. (CEC 1995 and 2005b). For further improvement of the existing framework, see IER (2004). Another follow-up of ExternE is the research project NEEDS, New Energy Externalities Development for Sustainable Development, see www.needs-project.org.

- Environmental priority strategies in product development (EPS), a system for weighting across impact categories that was developed for LCA during the 1990s (Steen 1999a; Steen 1999b). Although not intended as a method for monetising, it applies monetary values as a basis for comparing and aggregating different environmental effects.
- The Benefits Table database[20], BeTa, which was published in 2002 by the European Commission, including a series of estimates of marginal external costs of air pollution as well as a number of health and damage effects in Europe, (Holland & Watkiss 2002). The four air emissions covered by BeTa are shown in Table 5.4. Revision of these estimates has taken place in the context of the RAINS/GAINS model.
- The Ecotax method, which presents estimates of the economic value of different environmental effects. These estimates are transferred from Swedish environmental taxes (Eldh 2003).
- Andersen et al. (2004) uses the EcoSense model (developed in the ExternE project) to calculate the welfare economics damage costs from emissions to air of particles ($PM_{2.5}$), NO_x and SO_2 in Denmark. The costs are presented for four different types of locations (rural areas and urban areas with different numbers of inhabitants).
- The Norwegian pollution agency, SFT (2005) presents estimates of environmental costs of emission to air of NO_x, SO_2 and nmVOC for Norway. The costs are calculated using both a damage cost approach (dose-response) and implicit valuation based on Norwegian commitments in international agreements (the Gothenburg protocol).
- Bjerrum & Dengsøe (2004) present estimates of the damage costs of emissions of dioxin from waste incineration in Denmark.
- Enviros & Eftec (2004), assessing external costs and benefits from waste management options for the UK, including both landfilling and incineration. This project was undertaken on behalf of DEFRA, UK.

A database for exploring the latest developments within the growing field of valuation techniques exists; The Environment Valuation Reference Inventory (EVRI) has been initiated by a number of national EPAs and environment ministries and EEPSEA[21]. This database, which is accessible online, is highly recommended and provides monetary estimates with the purpose of facilitating low-cost benefit transfers. The EVRI database will be extended with Nordic valuation studies in the beginning of 2007.

[20] Compared to the ExternE study which assumed a value of statistical life (VSL) of EUR 3.1 mill, the BeTa assumes a VSL of EUR 1 mill.

[21] Environment Canada, DEFRA, US EPA, Ministère de l'écologie et du développement durable, and the Economy and Environment Program for Southeast Asia (EEPSEA). See: www.evri.ca and Navrud and Vågenes (2000).

Table 5.4 Marginal external costs of emissions in rural and urban areas (BeTa), EUR/tonne, 2000-prices

RURAL	SO_2	NO_x	$PM_{2.5}$	VOCs
Austria	7 200	6 800	14 000	1 400
Belgium	7 900	4 700	22 000	3 000
Denmark	3 300	3 300	5 400	7 200
Finland	970	1 500	1 400	490
France	7 400	8 200	15 000	2 000
Germany	6 100	4 100	16 000	2 800
Greece	4 100	6 000	7 800	930
Ireland	2 600	2 800	4 100	1 300
Italy	5 000	7 100	12 000	2 800
Netherlands	7 000	4 000	18 000	2 400
Portugal	3 000	4 100	5 800	1 500
Spain	3 700	4 700	7 900	880
Sweden	1 700	2 600	1 700	680
UK	4 500	2 600	9 700	1 900
EU15 average	5 200	4 200	14 000	2 100
URBAN		**$PM_{2.5}$**		**SO_2**
City of 100 000 people		33 000		6 000
Population Factors		**$PM_{2.5}$**		**SO_2**
500 000 people		5		5
1 000 000 people		7.5		7.5
Several million people		15		15

Note: Data are based on a 1998 emission scenario. Urban results for NO_x and VOCs are taken to be the same as the rural effects, given that quantified impacts are linked to formation of secondary pollutants in the atmosphere (ozone, nitrate aerosols). Given that these take time to be generated in the atmosphere, local variation in population density has little effect on the results. Urban externalities for $PM_{2.5}$ and SO_2 for cities of different sizes are calculated by multiplying results for a city of 100,000 people by the factors shown below. Results scale linearly to 500,000 people, but not beyond. These results are independent of the country in which the city is located. Once results for the cities are calculated, nationally specific rural externalities should be added to account for impacts of long range transport of pollutants, Holland and Watkiss (2002). The dose-response effect for $PM_{2.5}$ related mortality was scaled down in BeTa, using the consolidated dose-response effect the figures arrived at for mortality is a factor 3 higher. New estimates are available in the context of the CAFE (Clean Air For Europe programme) and RAINS/GAINS simulations.

6. Discounting

One of the most debated issues of the cost-benefit analysis is related to discounting: should it be done and at what rate?

This chapter aims at answering these two questions by presenting the concept of discounting, the social discount rates applied in the Nordic countries, and by discussing whether a single discount rate or a time-declining discount rate should be applied. A recommendation is given in the last section.

6.1 The concept of discounting

Discounting is a process of assigning a lower weight to a benefit or cost in the future than to that benefit or cost now. The first reason for this is that for the individual there is a general wish to consume today rather than later because of uncertainty about life expectancy. This is the "pure" time preference ("utility discounting"). The second reason to discount is that economic growth is normally expected to result in better future opportunities for consumption than the present ones; hence consumption postponed to the future should be assigned a discount rate so as to allow for adequate comparison with present consumption ("consumption discounting"), (Perman et al. 1999, 37ff.). This is known as Ramsey's "optimal growth model", where the marginal time preference, δ, consists of two elements: the pure time preference, ρ, and the expected growth in future consumption, $\varepsilon \cdot g$[22]:

$$\delta = \rho + \varepsilon \cdot g \qquad (6.1)$$

The discount rate, i.e. the weight put on future benefits and costs, is estimated as:

$$w_t = \frac{1}{(1+\delta)^t} \qquad (6.2)$$

where δ is the *real* discount rate, and t the time where the cost or benefit occurs. This approach is often called "exponential discounting", (Pearce

[22] The expected growth in future consumption is the product of: the elasticity of marginal utility, ε, and the rate of change of consumption, e.g. Hanley and Spash (1993, 129). In Boardman et al. (2006, 246) ε is a constant (>0), and g is the long-term rate of growth in per capita consumption.

2006, 184). This was discussed by Jespersen (1994, 94f) who showed the consequences of different discount rates[23]. Thus, the higher the discount rate, the lower the "weight" on the future will be. This is illustrated in Table 6.1. From the table it can be seen that at a discount rate of 5%, EUR 100 in ten years will only be worth EUR 61 in today's prices. Table 6.1 also shows that if the discount rate is zero, there is no loss in value and the future weights are the same as today.

Table 6.1 The future value of one monetary unit at selected discount rates

Time horizon	7%	5%	3%	0%
10 years	0.51	0.61	0.74	1.0
30 years	0.13	0.23	0.41	1.0
100 years	0.001	0.007	0.05	1.0

6.2 Choice of social discount rate

In a perfect economy with complete information and no distorting taxes etc., the marginal time preference, δ, will equal the marginal rate of return on capital. The latter can be derived from the market rates on investments and borrowing. However, with taxation (e.g. corporate and income) and transaction costs, the marginal rate of return on capital will in effect be higher than the marginal time preference (Boardman et al. 2006, 244). The question then is which rate should be used as the social discount rate: the marginal time preference or the marginal rate of return on capital? Or to put it differently: should the discount rate be defined on the basis of market interest rates or consumption rates? In practise, it is often difficult to estimate the marginal time preference rate, hence the discount rate is often derived from the real, after-tax, observable rate in financial markets (e.g. bonds)[24].

In the Nordic countries the recommended discount rates vary between 3% and 6%. The discount rates are shown in Table 6.2[25].

---

23 Discounting raises the question of inter-generational equity. If the current generation has a strong preference for current consumption (i.e. a high discount rate), which implies a high use of resources, then fewer resources will be saved for future generations. On the other hand, if the discount rate is zero and the interest rates are positive, it would imply that the current generation would reduce their consumption to benefit future generations. At the extreme, however low the level of consumption of the current generation, further reductions could always be made to increase the welfare of future generations, (Pearce 2006, 185).

24 Boardman et al. (2006) estimate the social discount rate on the basis of e.g. (6.1). Using annualised per capita quarterly data on real consumption expenditures for 1947-2002, they estimate an average growth rate for the United States of 2.3 per annum. Sensitivity analysis is done at 2.0% and 2.5%. The constant, ε, is set to 1, with sensitivity analysis at 0.5 and 1.5.

25 Møller (2003, 14ff) presents a proposal for a new way of managing discounting which includes that projects are discounted with a discount rate of 5% (reflecting the marginal rate of return on capital) and then among those projects with a positive NPV, the project with the highest NPV calculated at a discount rate of 2% (reflecting consumption discounting) is chosen.

Table 6.2 Discount rates in the Nordic countries

Denmark[1]	Finland	Norway[2]	Sweden
6%	5%	4% / 6%	4%

Notes: 1) The Danish Ministry of Finance recommends a discount rate of 6%. The Danish EPA recommends a discount rate of 3% combined with a return of capital calculated as an alternative rate of return of 6%. 2) In Norway a discount rate of 4% is recommended (calculated as a risk free discount rate of 2% and a risk premium of 2% covering systematic risk, whereas the risk premium should be 4% for projects with a substantial systematic risk profile; but for projects less dependent on the economic business cycle, a discount rate of 4% can be applied.

Source: Møller (2003, 14), the Norwegian Ministry of Finance (2005).

Møller et al. (2000, 137ff) argues that when investing in one project, society will lose the possibility of investing that money in another project, given that funds for investment are scarce. As a result, the social discount rate should be supplemented with a factor that also represents the alternative return of capital (in effect the shadow price on capital). For this purpose, the return of capital is estimated as, (Møller 2001)[26]:

(6.3) $$f_K = \sum_{t=1}^{T} \frac{q}{(1+r)^t} + \left(\frac{1}{(1+r)^T}\right) = \frac{q}{r} \cdot \left(1 - \frac{1}{(1+r)^T}\right) + \left(\frac{1}{(1+r)^T}\right)$$

where r is the social discount rate based on marginal time preference; q is the alternative return of capital; and T is time horizon of the project[27].

6.3 Constant vs. declining interest rate for environmental effects

It has been questioned whether the assumption of a constant discount rate is reasonable. Do individuals really have the same marginal time preference at all times in the future or does it vary? There seems to be empirical evidence for a time-declining discount rate, (Pearce 2009, 186; HM Treasury 2003, 98) so the question is how such a rate is to be estimated? So far, no definite approach has been found or decided on. For some approaches, the key issue seems to be how individuals treat uncertainty about the future.

[26] See also Boardman et al. (2006, 253ff) for a discussion of the shadow price on capital.

[27] The f_K was used in Table 5.2: with a discount rate of 3%, an alternative rate of capital of 6% and a time horizon of 25 years for the incineration plant, the f_K is 1.5225 and the amortisation factor is 0.0574. Then, the 'unit price' in Table 5.2 is equal to 1.5225 * 0.0574 = 0.0874.

Table 6.3 Example of declining certainty-equivalent discount rate

	Weights in period t				
Interest rate scenarios	10	50	100	200	500
1%	0.91	0.61	0.37	0.14	0.01
2%	0.82	0.37	0.14	0.02	0.00
3%	0.74	0.23	0.05	0.00	0.00
4%	0.68	0.14	0.02	0.00	0.00
5%	0.61	0.09	0.01	0.00	0.00
6%	0.56	0.05	0.00	0.00	0.00
7%	0.51	0.03	0.00	0.00	0.00
8%	0.46	0.02	0.00	0.00	0.00
9%	0.42	0.01	0.00	0.00	0.00
10%	0.39	0.01	0.00	0.00	0.00
Certainty-equivalent weight	0.61	0.16	0.06	0.02	0.00
Certainty-equivalent discount rate	4.73%	2.54%	1.61%	1.16%	1.01%

Source: Pearce et al. (2006, 187)

One of these approaches takes a probability weighted average (i.e. an expected value) to the likely weights, (Pearce et al. 2006, 186ff). Table 6.3 shows ten scenarios with different interest rates, and five different time horizons of projects. For the ten-year project a "certainty-equivalent discount rate" of 0.61 is estimated as the average of each year's weight. Based on this weight, the discount rate is 4.73% using equation (6.1) in Section 6.1. This approach suggests a declining discount rate that goes towards 1% at very long time horizons.

Pearce et al. (2006, 191) do not provide a clear recommendation on the issue, but concludes that a time-declining discount rate[28] would overcome the problem of the "exponential discounting", but it may have other problems of time consistency. However, Hansen (2006) concludes that time inconsistency is a feature of discounting as such and not due to declining discount rates.

The Norwegian Ministry of Finance (2005, 39) recommends using the same discount rate for all costs and benefits, also environmental effects. The reasoning is based on the assumption that the relative prices for environmental benefits are likely to increase over time, both due to the fact that future generations will be more wealthy (and hence have an increased WTP for the environment) and that there might be an increasing shortage of environmental benefits in the future. This increase in the relative price for environmental goods should, according to the Norwegian Ministry, be handled through the shadow prices used for these goods and not through the discount rate. This recommendation is supported by Møller et al. (2000, 146).

Another approach is recommended in the United Kingdom, where the Treasury has introduced a time-declining interest rate for projects where the time horizon is more than 30 years, (HM Treasury 2003, 97ff). The

[28] Hyperbolic discounting

French Commissariat Général du Plan has also revised its discount rates so that it is 4% for the first 30 years, and thereafter there is a continuous decrease to a lower level of 2%, (Lebègue et al. 2005).

A time-declining discount rate is also supported by Boardman et al. (2006, 264ff). Different sets of discount rates for different time horizons are suggested depending on whether the project runs for more or less than 50 years; and how the social discount rate has been defined (on the basis of market-based interest rates or the marginal time preference).

6.4 Recommendation

The choice of discount rate is difficult as it may have a decisive influence on the outcome of the CBA. Nevertheless, discounting should be made, and this guideline recommends that the national choices of discount rates are applied in the CBA. Moreover, a sensitivity analysis should be made for the discount rate (e.g. +/- 2%).

When the same discount rate is used for all costs and benefits it is necessary also to account for the fact that the prices for environmental benefits are likely to increase over time. The reason is that the environment is physically limited and does not grow as the economy grows. Accounting for the increasing price of environmental benefits requires a dynamic method for valuation of environmental benefits and costs. None of the methods described in Section 5.2 is dynamic in itself. This means that the valuation methods need to be supplemented by a time-dependent technique, if the discount rate is applied to environmental costs and benefits.

For further discussion of the use of discounting, see Boardman et al. (2006), Møller (2003) and ECON (2001).

7. Evaluation

The evaluation is the crucial step in a CBA: should the policy or projected be accepted or rejected? In the evaluation the costs are subtracted from the benefits to estimate whether the policy or project is beneficial to society. As mentioned already, the CBA is rarely able to capture all the relevant information for the decision-maker. The presentation of the results, along with the limitations of the study, may therefore be very important for the final outcome of the decision to be made.

This chapter presents the net present value as the general decision criterion and discusses why other criteria should not be applied. Other considerations for evaluation of the results apart from the strictly economic aspects are also presented. Finally, it is discussed how to include risk in a CBA, and why it is important to test the robustness of the results through a sensitivity analysis.

7.1 Evaluation criteria

The CBA is a tool for selecting the projects or policies that are efficient in terms of their use of resources. The appropriate criterion for evaluation is the net present value criterion. Despite this, other criteria are sometimes applied.

7.1.1 Net Present Value (NPV)

The investment (I), costs (C_t) and benefits (B_t) and how they accrue over time (t) can be illustrated by using time lines, as shown in Figure 7.1.

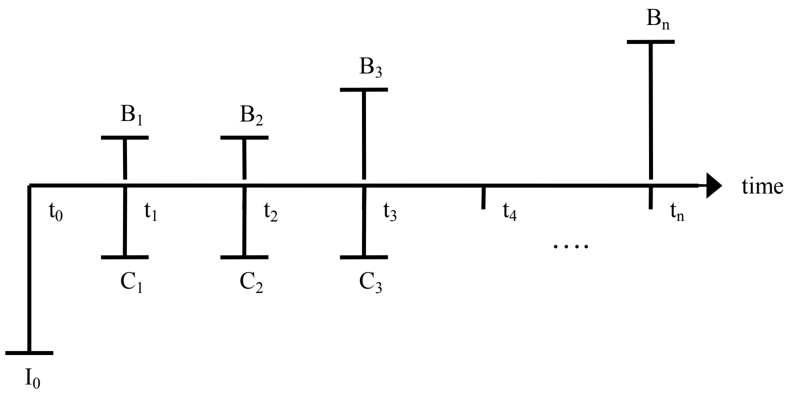

Figure 7.1 Illustration of the time line of investment, costs and benefits

The criterion for accepting a policy or project is that the sum of discounted benefits exceeds the sum of discounted costs. If the net present value (NPV) is positive, the policy or project is beneficial to society. The net present value is calculated as:

$$(7.1) \qquad NPV_0 = \sum_{t=0}^{N} \frac{B_t - C_t - I_t}{(1+r)^t}$$

where B_t is the benefit at time t, C_t is cost at time t, I_t is the investment at time t, N is the time horizon, and r is the social discount rate discussed in Chapter 6. As shown in equation (6.3), Møller et al. (2000, 151) suggest including the alternative rate of return on capital, whereby the NPV becomes[29]:

$$(7.2) \qquad NPV_0 = \sum_{t=0}^{N} \frac{B_t - C_t - f_K I_t}{(1+r)^t}$$

In principle, all projects and policies with a NPV > 0 should be implemented or undertaken. A positive NPV indicates that the rate or return is higher than the alternative rate of return. However, due to budget constraints or to the fact that some policies or projects are mutually exclusive not all policies and projects will be undertaken. Hence, it may be necessary to rank the projects.

According to Pearce et al. (2006, 67) the general rule is to rank projects by their NPVs. This is also true if the projects are mutually exclusive.

If the projects are independent, the Norwegian Ministry of Finance (2005, 19) and the Finnish Ministry of Transport and Communication[30] recommend[31] an approach where the projects are ranked according to their benefit-cost ratio, or rather the NPV per cost unit of the project[32]. This is calculated as:

$$(7.3) \qquad NPVP_0 = \frac{NPV_0}{\text{Present value of all costs of the project}}$$

[29] If the social discount rate is equal to the opportunity cost of capital (i.e. the alternative rate of return), the internal rate of return is a test of whether a project earns a social rate of return that exceeds what could be earned by investing economic resources elsewhere, (Pearce et al. 2006, 71).

[30] Personal communication, Markku Olikainen, October 2006.

[31] According to Pearce et al. (2006, 69) ranking of projects in terms of the benefit-cost ratio is for single-period rationing of scarce resource inputs only.

[32] This approach is similar to a ranking in terms of the benefit-cost ratio of projects.

To illustrate this approach, an example is shown in Table 7.1. Two projects, A and B, have initial investment costs of EUR 10 and 5, and annual benefits equal to EUR 2.9 and 1.5 respectively. The discount rate is 4%. When comparing the two projects, project A has the highest NPV, but project B has the highest NPV per payment ($NPVP_0$) on the project. This criterion however, is only valid when the projects are not mutually exclusive and when the value of increasing the budget constraint is constant in all periods.

Table 7.1 Example: how to rank projects according to NPV per payment

	Project A	Project B
Costs, t=0	-10	-5
Annual benefits	2.9	1.5
Time horizon, years	4	4
Discount rate	4%	4%
NPV	0.53	0.44
$NPVP_0$	0.053	0.089

Source: The Norwegian Ministry of Finance (2005, 19)

7.1.2 Other evaluation criteria

In addition to the NPV, other criteria are sometimes applied or referred to. Nevertheless, none of these criteria should be applied in a CBA. The criteria are:

- Benefit-cost ratio (B/C)
- Internal rate of return (IRR)
- Pay-back time

The benefit-cost ratio is the ratio of benefits over costs, and according to this criterion any project with a B/C ratio > 1 should be implemented, as the benefits per unit of cost are positive. However, the B/C ratio can confuse the choice when projects are of a different scale (e.g. differ significantly in costs) and if the benefits are sensitive to the treatment of willingness-to-pay (e.g. if WTP is negative, is it treated as a cost or negative benefit). The B/C ratio criterion may also produce a wrong outcome if projects are mutually exclusive. As a consequence the benefit-cost ratio time should not be used as an evaluation criterion, (Boardman et al. 2006, 16 and 33).

The internal rate of return is the discount rate that will yield a NPV = 0, and according to the IRR criterion, a project with an internal rate of return higher than the discount rate is beneficial. However, applying the internal rate of return encompasses some problems as it may not always result in the optimal solution. It is possible that the project with the highest internal rate of return does not have the highest NPV, which would lead to an erroneous choice. Also, more than one internal rate of return may exist, and then the internal rate of return criterion is misleading. As a consequence the internal rate of return should not be used as an evaluation criterion, (Boardman et al. 2006, 16; Danish Ministry of Finance 1999, 31f; Hanley and Spash 1993, 18; Pearce et al. 2006, 71f).

The pay-back time shows the amount of time before the net benefits have paid back the investment cost. One serious drawback of the pay-back time criterion is that it does not take into account the payments and benefits that occur after the investment has been repaid. Moreover, the pay-back time cannot be used to compare two or more projects. As a consequence the pay-back time should not be used as an evaluation criterion, (Danish Ministry of Finance 1999, 31f).

7.1.3 Projects with different time horizon

If the projects have different time horizons, they need to be made comparable. To do this, two approaches are recommended and they both provide the same result, (Boardman et al. 2006, 145f):

- Rolling over the shorter project: choose a time horizon equal to the lowest common denominator of the projects. If time horizon of project A is 10 years and 5 years for project B, then project B will be repeated after termination.
- Equivalent annual net benefit method: divide the net present value with the annuity factor that has the same time horizon and discount rate as the project.

In addition to these approaches, Møller et al. (2000, 33f) also suggest that projects could be compared by choosing an infinite time horizon.

7.1.4 The optimal time for starting projects

The starting time (year) of the baseline and alternative scenarios should be made explicit. The time for initiating the policy or project may influence the outcome if there is an uncertainty about the future benefits and cost, which may be replaced with certainty if the project is postponed for some time. Examples regarding the choice of starting year are presented in the Norwegian Ministry of Finance (2005, 38), Møller (2003) and the Danish Ministry of Finance (1999, 18).

7.2 Sensitivity analysis

When the CBA is conducted before the actual implementation of the policy or project, i.e. as an ex-ante study, there may be some uncertainty about what the actual quantities and prices will be. The purpose of a sensitivity analysis is to study how sensitive the result is to changes in key assumptions. For this reason, it is recommended always to conduct a sensitivity analysis of the key assumptions made in a CBA.

Two forms of sensitivity analysis are presented here: the partial analysis where only one variable is changed at the time; and the Monte Carlo analysis that allows an assessment of the consequences of simultaneous uncertainty about several variables and can take into account correlations of these variables.

7.2.1 Partial sensitivity analysis

The sensitivity analysis may be supplemented by a "worst case" and "best case" that show the worst or best possible values.

If the result varies significantly with a change in a key variable, then it is important to define this variable as precisely as possible.

Møller et al. (2000, 153) assume that the uncertainty for market-based goods is limited whereas the uncertainty for the environmental effects and monetisation usually will be considerable.

Below some typical key variables for the analysis of waste and resources are listed. It should be considered whether a sensitivity analysis is necessary for the following variables:

- discount rate
- time horizon of the policy or project
- quantity of waste to be managed or the resource considered in the analysis
- investment cost
- prices of recycled products
- prices of electricity and fuel
- monetised, key environmental effects
- householders' time spent on recycling activities and the valuation of this time
- change in demand of waste disposal, e.g. due to changes in regulation or public awareness.

The outcome of the sensitivity analysis should be presented together with the result of the CBA, so that it becomes visible for the decision-maker where the potential drawbacks of the policy or project are. Also, the decision-maker should be made aware of any assumptions that are uncertain and have a high impact on the result, (Danish Ministry of Finance 1999, 37).