North Denmark EU-office
Rendsburggade 6
DK 9000 Aalborg
www.eu-norddanmark.dk

The WaterCost project is funded by
the EU Interreg IIIB North Sea Programme

METHOD AND CASE STUDIES

WaterCost

ELEMENTS OF COST-EFFECTIVENESS ANALYSIS

FOREWORD

Groundwater protection measures and their cost effectiveness are at the heart of the EU policy protection framework, primarily through the WFD (Directive 2000/60/EC) and the new Groundwater "daughter" Directive 2006/118/EC. In this context, Member States have to implement measures necessary to prevent or limit the input of pollutants into groundwater and to prevent the deterioration of the status of all bodies of groundwater with the ultimate objective to reach a "good status" by 2015. Under this regime, (ground)water management is designed along the development of River Basin Management Plans (RBMP), which should integrate all identified pressures and impacts and identify appropriate programmes of measures to tackle them.

This implies a thorough assessment of risks and a design of appropriate responses which are not only based on an effective implementation of parent legislations from various sectors (e.g. agriculture, industry, nature conservation etc.) but also on an evaluation of cost-effectiveness and technical feasibility of identified measures.

The identification and implementation of protection measures involve risk-based integrated components, requiring multidisciplinary and multisectoral cooperation. The complexity of this management makes it necessary to proceed in a stepwise, iterative manner, ensuring an effective participation of water actors and a full integration of scientific knowledge. In this respect, the WaterCost project is providing new knowledge about cost effectiveness analysis which is based on case studies presented in the project booklet.

The way results are presented in the booklet, describing the case studies, the cost-effectiveness analysis concept and its applications, will be a precious source of information and inspiration for Member States while designing programmes of groundwater protection measures.

Philippe Quevauviller
European Commission, DG Environment

TABLE OF CONTENTS

Introduction

1

1.1 Project background

Sustainable water management is a key challenge for Europe and one of the drivers for the Water Framework Directive (WFD). The focus of the WaterCost project is closely linked to the Groundwater Directive (GWD) and the groundwater aspects of the WFD. According to the WFD the member states and local/regional authorities are to develop River Basin Management plans by 2008-09. Protection of drinking water resources must be integrated for each river basin – incl. protection of aquatic ecology, habitats and bathing water; an integrated approach to water management.

Cost-effectiveness of protective measures for groundwater is at the core of the River Basin management plans, the WFD and the GWD. The implementation of these directives will have a significant effect on competent authorities; whose focal point is that they need to reach the policy targets of the Groundwater Directive with the limited resources available.

The river basin plans shall include detailed accounts on how the objectives for the river basin (i.e. water quality) shall be reached and an economic analysis – in order to enable a rational discussion on cost-effectiveness of various protective measures.

Thus the essential question is: "How to reach high quality groundwater for the least amount of money?" Spatial plans need to ensure a fair balance between the value of the groundwater resources and the total costs of the plan and thereby aim at ensuring land use which will provide high quality groundwater as well as avoid extra costs of cleaning the groundwater.

It is therefore essential to analyse the cost-effectiveness of the various protective measures in order to plan for land use that contributes to the objectives of the WFD. The foundation for doing this has not yet been fully discussed or developed. The WaterCost project has aimed to provide a broad picture of the cost-effectiveness of groundwater protection measures.

The WaterCost project has aimed to provide a broad picture of the cost-effectiveness of groundwater protection measures

1.2 WaterCost objectives and activities

The intention with the WaterCost project has been to deliver a framework for Cost-effectiveness analysis (CEA); a framework to assess the cost effectiveness and include wider impacts for various measures for groundwater protection. Thereby, the WaterCost project aims to provide guidance on how to obtain an integrated and environmentally sustainable approach to groundwater protection and management.

This objective is clearly reflected in the 4 case studies that have been carried out as part of the WaterCost project. The British case study is setting out to look in more detail at the cost-effectiveness of different measures and the non-market benefits associated with them. The purpose with the Danish study is to cast light on the implications of wider impacts to cost-effective ranking of measures aimed at ground water protection. The German case study aims to test the CEA approach, as a way of finding the most cost-effective way of meeting this target for Water Protection Area (WPA) Thüsfelde. The purpose with the Dutch study is to justify the higher costs related to the Tusschenwater Project, which provides an integrated solution for an area, which solves several problems that have to be solved. It is studied if the higher project costs can be justified by taking side benefits into account.

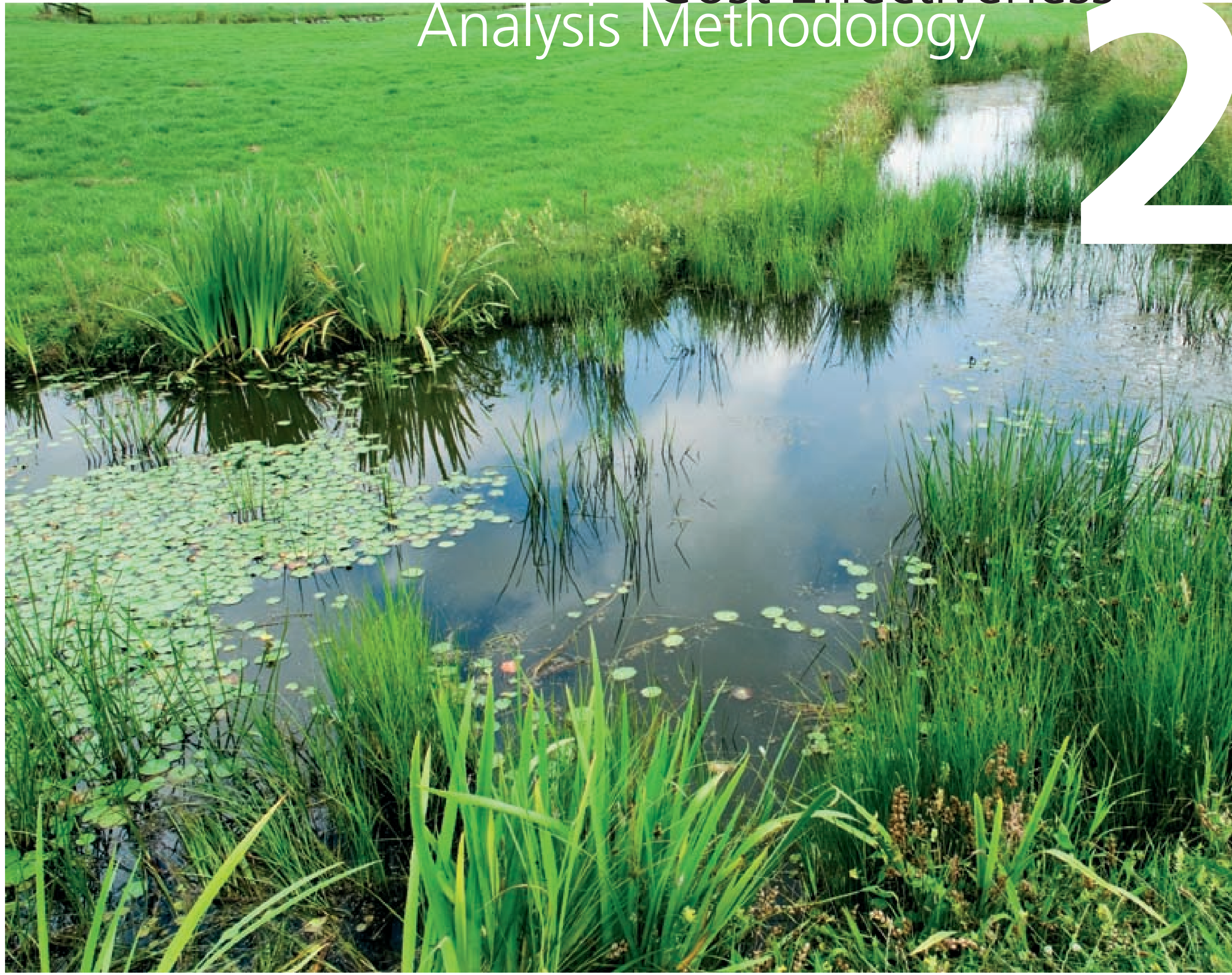
1.3 The booklet

The main content of this booklet is the presentations of the 4 case studies, which shows how CEA have been applied in partner regions, identified strength and weaknesses with the methods, etc.

The booklet also contains a description of the CEA concept and the different approaches associated with it, as well as a general approach to inclusion of wider impacts in CEA.

Cost Effectiveness Analysis Methodology

2



2.1 Introduction

The Water Framework Directive (WFD) requires that a River Basin Management Plan (RBMP) is elaborated before 2009 for each river catchment in the EU. Each RBMP is required to include a cost-effective action plan on how to meet the water quality objectives the cheapest possible way. For this reason, during the coming years every river basin authority throughout the EU will have to carry out a CEA. Methodologies for Water Framework Directive related CEA have, to some extent, been elaborated previously in EU projects 1) and at national scale 2). None of these have, however, targeted groundwater specifically.

The purpose with the WaterCost project, including the case studies carried out, is to test available methodologies to find out if they are sufficient to meet the special requirements for groundwater. Furthermore, it has been a specific purpose to find and test an approach to include wider impacts to the CEA, which meets the requirements of 1) resting on sound theory and 2) being reasonable to implement.

Cost Effectiveness Analysis is a concept to find the options or combination of options that can meet the target in the cheapest possible way

2.2 What is CEA?

Cost Effectiveness Analysis is a concept to compare different options to meet the same target, in order to find the options or combination of options that can meet the target in the cheapest possible way. From a methodological point of view, CEA is a relatively open concept, which can be applied on different levels of analysis, ranging from, a screening based on an analysis using financial prices to, a political decision support analysis based on welfare economic theory.

The Water Framework Directive is not decisive as to which level to carry out the CEA. The WATECO guideline recommend an economic approach (as opposed to a financial), but do not provide specific guidance. Later guidelines (Donne van Engelen et.al. (2007), Danish guidelines (prepared but not published), Entec (2006)) are in favour of a (welfare) economic approach to CEA. This approach is closely connected to the welfare theory, and is for that reason sometimes referred to as welfare economic CEA, and has the objective to find the best alternatives for the society as a whole. The exception to this rule is the German guideline, which recommends a multi-criteria approach. On the WaterCost project the partners agreed to use the Entec (2006) guideline to CEA, which is recommending an economic approach, and is pragmatic but open to different levels of methodological sophistication. In the following a "narrow" CEA refer to a CEA where wider impacts have not been taken into account, while an "extended" CEA refer to a CEA where also the wider impacts have been included.

1 E.g. the NOLIMP Interreg project (www.nolimp.org/)
2 Entec (2006), Interwies, M.A et.al. (2003), the Danish methodology (elaborated but not published)

2.3 The 6 steps in CEA[3]

The WaterCost project has evolved during four workshops involving the partners in the project (Copenhagen Energy, OOWV, Landwirtschaftskammer Niedersachsen, provincie Drenthe, Environment Agency, Kristianstads Kommun and Aalborg Municipality). At the outset a methodology for CEA of groundwater was elaborated and discussed. It was agreed to take point of departure in the Entec (2006) guideline document, which contains a stepwise approach on how to carry out a CEA based on sound economic principles.

Below, the 6 steps in the standard methodology are briefly described. For more elaborate guidance to each of the steps, please refer to the Entec (2006) methodology.

Step 1: Describe and Quantify Gaps

Identify the gap in environmental condition that needs to be closed in order to meet the specified WFD objective. This gap is the difference between the expected environmental condition in 2015 and the environmental standards necessary to achieve the specified objective. In short, this involves three sub-steps, namely defining the problem being considered i.e. identifying pressures (e.g. diffuse sources) and activities (e.g. agriculture); identifying the objective, e.g. "Good status in 2015"; and finally, identifying the gap to be closed, which is the difference between the environmental conditions that are predicted to occur in 2015 if no additional WFD measures are put in place and the environmental standard(s) specified for the parameter being considered.

Step 2: Identify Measures

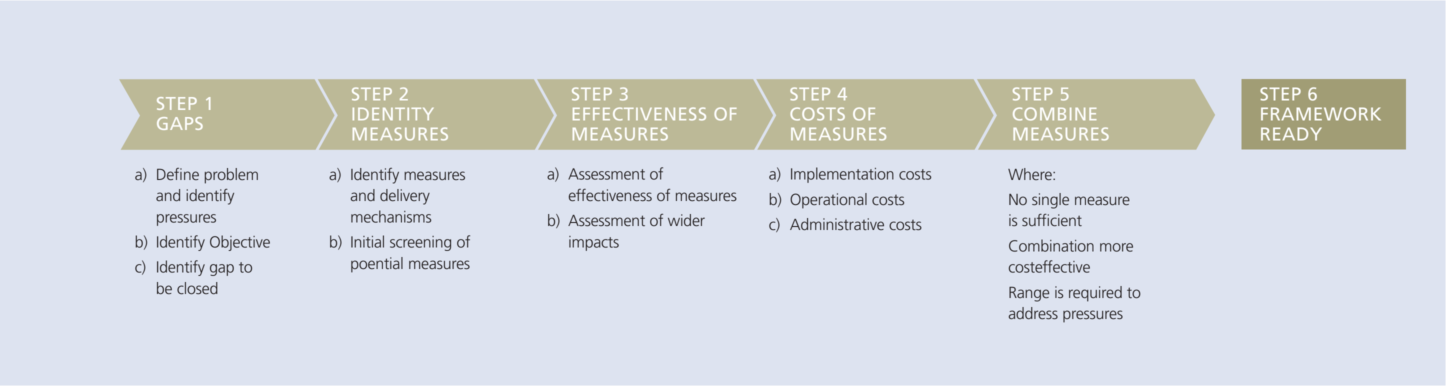
Having described and quantified the gaps between expected and target conditions, options for measures and delivery mechanisms to make environmental improvements can be identified. Activities in this context are identifying measures and delivery mechanisms (a measure is assumed to cover both the measure and the delivery mechanism identified for that measure) and initial screening of potential measures: Is the measure technically feasible? Is the measure clearly less cost-effective?

Step 3: Consider Effectiveness of Measures

The effectiveness of a measure relates to how successful that measure will be in closing the gap required to meet the specified WFD objective by the specified deadline. The consideration of effectiveness of measures involves an assessment of the effectiveness of measures in closing the gap and an assessment of the wider impacts of the measures. The former is the extent to which the measure will make environmental improvements to close the gap and the time it takes for the measure to become effective and deliver environmental improvement, combined with the probability that the measure will have the desired effect. The latter is the impacts on water resources, in terms of both the availability of water for the environment and the possible impacts on reducing the availability of water for abstraction, the impacts on water quality, aquatic habitats, and fish migration and fisheries.

The effectiveness of a measure relates to how successful that measure will be in closing the gap required to meet the specified WFD objective

3 As set up in Entec (2006)



The total economic costs should be considered only if it is not possible to make a confident judgement of the most cost-effective measure based on the direct financial costs

Step 4: Consider Costs of Measures

The costs of each measure are then considered. This must take into account implementation costs, ongoing operational costs and any administrative costs associated with the implementation. Additional positive and negative impacts not related to the targeted objective must also be considered. All costs are annualised.

Assessment of the economic costs of potential measures takes the point of departure in the direct financial costs of the measures (what it will cost the operator and the competent authority to implement the measures). In order to derive the direct economic costs of the measures, the financial costs are adjusted for various transfers such as tax payments. On top of this, the wider impacts of potential measures are assessed in order to arrive at the total economic costs, which combine the direct costs and the costs (or benefits) associated with wider environmental, social or economic impacts.

Following the main principle of assessing costs to the extent necessary to support a decision, the approach to the cost assessment should initially be to compare the direct financial costs. If there is a significant difference that allows the user to make a confident judgment, there is no need for any further assessment. Generally, the total economic costs should be considered only if it is not possible to make a confident judgement of the most cost-effective measure based on the direct financial costs.

Step 5: Combine Measures

Combinations of identified measures and mechanisms that may close the gap are then identified. The combined cost-effectiveness of these measures can then be included in the assessment. This is done by identifying combinations of measures, where no single measure has been identified as being sufficiently effective to fully close the identified gap by itself; where a combination of measures may be more cost-effective in closing the gap than any individual measure, even though individual measures have been identified that are able to fully close the gap; and finally where a range of measures is required to address pressures arising from the activities of different sectors.

Also, changes to effectiveness and costs of combined measures should be considered at this stage. This may be the case if the incremental effect of one measure included in the combination is affected by the implementation of another included measure, or if the combined implementation of measures results in cost savings as a result of streamlined operational costs.

Step 6: Identify Most Cost-Effective Measure/Combination of Measures

Depending on the research question an adequate framework is set up for identification of the most cost-effective combination of measures.

2.4 Assessment of costs in the 4 case studies

The four case studies show a large degree of difference in the methodological treatment of costs, which is in full accordance with step 4 above, in which it is stated that "the approach to the cost assessment should initially be to compare the direct financial costs. If there is a significant difference that allows the user to make a confident judgment, there is no need for any further assessment".

The British case study is based on generic costs, which have been found in the national inventory of measures [4], and hence are assessed in accordance with the general theory, as outlined in the Entec (2006) methodology. It is, however, not clear to what extent taxes have been taken into account, and as such whether a welfare economic market price approach has been applied. Since the additional conversion factors described above were only introduced during the project process (with inspiration from the current praxis in Denmark) these have not been included in the cost figures applied in the British study. The Dutch and the German case studies are based on cost assessments carried out specifically for the measures being considered. The overall principle behind the costing methodology applied in the two case studies follow the idea behind economic CEA, although some prices included are financial costs, which have not been adjusted to economic costs, taking taxes and conversion factors into account. The Danish case study is carried out in accordance with the welfare theory and the Entec (2006) guidelines, but is, however, not an attempt to carry out a full CEA for the WMA in question.

The approach to the cost assessment should initially be to compare the direct financial costs

2.5. Welfare Economc Extentions to the Entec framework

During the WaterCost project process it was suggested to extend the The Entec (2006) framework to assimilate closer to the welfare theory, in order to suggest an approach that is generally applicable and yet resting on sound economic principles. Specifically it is suggested to include three conversion factors. This relate specifically to assessment of costs and assessment of wider impacts (step 3 and 4 above).

[4] See Cuttle, S.P. et.al. (2006) and (2007)

2.5.1 Standard Conversion Factor (SCF)[5]

The point of departure for welfare economic analyses is that consumption is being used to measure welfare effects, i.e. more consumption means more welfare. Consumers optimise spending on goods in order to be in line with preferences. This optimisation is based on market prices including taxes. Therefore the welfare economic analysis should also be based on market prices including taxes.

The narrow CEA is based on values which are possible to value by market prices

Project related costs are, however, typically given in factor prices (i.e. prices without taxes). In this case, prices should be adjusted with the SCF in order to reflect the market price. As such the SCF is a short-cut from factor prices to market prices. The SCF is defined as the ratio between the Gross Domestic Product (value added including taxes) and the Gross Factor Income (value added without taxes). In Denmark, the SCF can be calculated to 1.17.

If the prices in question reflect internationally traded goods, a SCF for internationally goods should be applied, with the same reasoning as for the SCF, discussed above. The SCF for internationally traded goods is the ratio between the value of imported goods including taxes and the value of imported goods in factor prices. In Denmark, the SCF for internationally traded goods can be calculated to 1.25.

2.5.2 Deadweight loss [6]

Income taxes cause distortions to the economy, which makes total output smaller than it otherwise would have been. As such, taxes cause a "dead weight loss" to the economy. Since public funding of projects is financed from these distorting taxes, public funding therefore adds an extra cost (dead weight loss) to a project. Ministry of Finance estimates that in Denmark the dead weight loss amounts to 20%, i.e. the marginal costs of public funding (MCPF) is 1.2

2.5.3 An example of welfare economic cost categories for a typical measures

The table below gives an example on which costs should be included in the cost assessment of a typical measure, using the Entec (2006) methodology combined with the 3 conversion factors described above, i.e. taking a welfare economic approach to CEA.

Table 2-1 Example of cost categories included:

Afforestation

- Dead weight loss from the sales value of agricultural land
- Investment costs, afforestation
- Cost of lost agricultural production
- Earnings from afforestation
- Income from game hunting
- Administrative costs

---

[5] See Møller et. al. (2000) (look for "nettoafgiftsfaktor")
[6] Ibid.

In the example shown in the table above, the water authority buy a number of hectares in order to convert the land use from the current agricultural use to afforestation.

The first thing that happens is that the public authority buys the land. The acquisition costs as such are not considered, since this amount is only passing from one agent to another within the same system, resulting in no real economic effect. This is generally referred to as "an internal transfer. The only impact of the land acquisition being included in the analysis is the deadweight loss arising because the buying party is the public sector, and hence financing the acquisition through taxes or fees, which are distorting the economy, as was described in section 2.5.2 above.

Having acquired the land, the public authority invests in afforestation. Since trees are produced domestically, the investment cost (including labour costs) is multiplied with the standard conversion factor in order to arrive at welfare economic prices including taxes, and annualised using the discounting factor. Afforestation of the area has two direct effects: Agricultural production is lost and new income is generated from afforestation. These are found calculating the welfare economic land rent, which simply put is farm gate profit per hectare of the land use in question multiplied by the standard conversion factor. One indirect effect is recorded, namely game hunting, which is known to be better in an afforested area than in an agricultural area. The assessed increase in value of game hunting for the area is multiplied by the standard conversion factor. Finally, administrative costs are assessed and multiplied by the standard conversion factor.

2.6 Wider impacts

It is obvious, that goods like milk and a trip to the Cinema has a value. These goods can be bought at the market for a given price. The narrow CEA is based on values which are possible to value by market prices. However, it is not a condition that only goods that are sold on a market represent a value to consumers. As an example most people gain great pleasure when walking in the forest. The extended CEA includes also values which are not valued by market prices, like the example with a walk in the forest.

In order to carry out an extended CEA, results from environmental valuation of the relevant wider impacts are needed. Environmental valuations are generally costly to carry out, and furthermore it is difficult to generate reliable results. In this section approaches to environmental valuations are briefly introduced, after which a pragmatic approach to include wider impacts in the CEA is suggested.

Extended CEA includes also values which are not valued by market prices

2.6.1 Economic value

The total economic value expresses the total value of a good such as a walk in the forest. The total economic value consists of so-called use and a non-use values.

A characteristic of the use values is that people attain pleasure when using the good. The non-use value refers to the value people derive from goods independent of any use. Both the use and the non-use value can be subdivided into smaller groups. The use values include:

Table 2-2
The different wider impacts.

	Direct use value	Indirect use value	Option value	Non use value
Recreational opportunities	x		x	x
Biodiversity	x		x	x
CO2 storage		x		
Filtration and decomposition of polluting subatances		x		
Erosion control		x		
Robustness towards climate change		x		

- The direct use-value: The benefit people experience from direct use of environmental goods either personally or as a production factor.

- Indirect use value: The benefit people indirectly experience i.e. filtration and decomposition of polluting substances and CO2-storage of forests, etc.

- Option value: The value individuals experience from having opportunity to use existing environmental goods, e.g. recreational areas.

The non-use value includes:

- Existence value: the benefit people receive from knowing that a particular environmental resource or endangered species exist.

- Bequest value: the value of preserving a good for the use of future generations

2.6.2 Description of impacts

Apart from the direct effects like increased hunting value afforestation and permanent grass contributes with a number of wider impacts. The most important are listed in the table 2-2.

As can be seen from the above table, the impacts are categorised according to the kind of value they belong to.

2.6.3 Valuation of wider impacts

Several techniques have been developed to measure the value of goods that are not traded in a market. The techniques are divided in two categories, depending on whether the approaches use consumer preferences or not while measuring the values. These are called preference based and non-preference based approaches, respectively.

The non-preference based methods use the costs to attain a given environmental target as a starting point. Thus the value of CO2 can be estimated by looking at the alternative reduction costs i.e. the allowance prices.

The preference based methods can be subdivided into stated/hypothetical and revealed methods. In the stated methods, a hypothetical market is created, at which respondents are asked about how much they are willing to pay for the good in question i.e. how much are you willing to pay to get access to specific recreational areas (in the Danish case: Øster Ådal)? In this case, the valuation is based on interviews.

The revealed preference approach uses the housing market as a base for the valuation. If it is possible to identify a positive relationship between the house prices and the environmental good it is assumed that the price premium reflects the value of the environmental good.

There is an ongoing discussion about reliability and usefulness of environmental valuation. However, research (and experience) has increased the validity of the benefit estimates as advanced valuation methods have been developed.

2.6.4 Approach taken to inclusion of wider impacts [7]

As will have become clear after having read the above sections, valuation studies of wider impacts of nature protection are generally very complex and for that reason time consuming and expensive to carry out.

Since the main objective with applying a CEA is to find a solid ranking of various alternatives to meet the same objective, focus is on whether the relative difference between the Cost/Effectiveness ratios for the alternatives is right – and not so much on whether the absolute values are correct. This aspect of the CEA concept is applied to develop a simple approach for inclusion of wider impacts to the analysis.

For practical purposes the so-called "benefit transfer" approach has been developed. This is a relatively simple approach, where results from studies carried out in one location is transferred and applied to other locations. During the transfer process values are adjusted to differences between the two areas. As such, benefit transfer allows for putting money values to nature in a relatively simple manner. It is, however, a requirement for benefit transfer that relevant studies have been carried out in advance. A modified approach to benefit transfer was suggested at the WaterCost project, and has been used in all of the four case studies.

The modified approach[8] to benefit transfer was first used in Denmark, and has been applied in a study for the Danish EPA. The approach takes point of departure in a narrow CEA. The narrow analysis ranks different measures with respect to their relative cost-effectiveness. The question is whether the wider impact can be expected to change the ranking of the measures. To provide an answer to this question, first the kind of wider

The main objective with applying a CEA is to find a solid ranking of various alternatives to meet the same objective

---

[7] For a general, however technical introduction to environmental evaluation and benefit transfer, see for example Freeman (2003)

[8] See Miljøstyrelsen (2006), methodology section

impacts associated with the different measures are identified. When this is done, research on relevant studies of the value of the wider impacts are carried out. These studies give an indication of the relative sizes of the wider impacts. If these relative sizes point to the same ranking as was done by the narrow CEA, no further action is taken. If the wider impacts suggest that the ranking might be changed, the actual relative size of the value of the wider impacts needs to be considered. Based on the studies included, an argument is produced in favour of either maintaining or changing the ranking.

The 4 case studies all take wider impacts into consideration, using aspects of the modified benefit transfer approach ranging from Multi-criteria analysis based on expert judgement (German and Dutch case studies), over research based semi-quantification (the Danish case study) till development of a stringent mathematical framework for inclusion of wider impacts (the British study).

The application of the modified approach to benefit transfer in the case studies explore a wide range of possible levels for analysis, while they all take wider impacts into consideration.

Case studies

3

Figure 3-1 Groundwater protection areas

On basis of the decided CEA methodology this chapter will provide the reader with insight into the CEA approach in the partner regions and the applicability of the selected measures in the regions. This will illustrate similarities and differences between the regions.

The case studies will also provide assessment on the cost-effectiveness of the chosen measures. The presentation of these case studies will lead to recommendations / suggestions for an approach to CEA. This will be presented in the final chapter.

3.1 Denmark, Aalborg

3.1.1 Introduction

The purpose with this study is to cast light on the implications of wider impacts to cost-effective ranking of measures aimed at ground water protection. This is done by preparing a "narrow" Cost-Effectiveness Analysis (CEA) for the nitrate and pesticide reducing measures afforestation and permanent grass in an area in the vicinity of Aalborg Municipality in northern Denmark.

Today, groundwater protection is done through preparation and implementation of action plans. These plans are complete "packages" to be fully implemented before the groundwater in question is being protected. A package typically consists of measures that address pollution with nitrogen, pesticides and from point sources, respectively. Within a groundwater protection package it will be possible to rank specific initiatives to the extent that alternative measures exist. In that connection it will be of particular interest to look at the wider impacts of the measures. However, typical initiatives that address pesticides and point sources will not or only to a limited extent have wider impacts. Therefore, wider impacts of groundwater protection are primarily linked to measures that are directed towards nitrogen, i.e. area-based initiatives in a Danish context.

The general purpose with the analysis carried out is to provide input for applying the CEA methodology on groundwater related measures. The specific purpose is to illustrate the wider impacts of groundwater protection, – as mentioned above, primarily wider impacts of area-based initiatives.

Analyses are carried out for three selected area-based measures from two existing Groundwater Protection Plans – one for an area close to the town of Drastrup[9] and one for an area in Aalborg South East. This is done in order to estimate the narrow and wider economic impacts of these measures. The narrow analysis provides a socio-economic measure with only the direct costs and benefits associated with a given measure. A broader analysis includes wider impacts i.e. the welfare benefits from improved nature, such as recreational opportunities, more biodiversity etc.

3.1.2 Approach

During the WaterCost project it was decided to test the usage of a CEA Guidance document and spreadsheet tool developed for DEFRA by Entec Ltd.[10] to deal with Water Framework Directive (WFD) related CEA on a national level. The spreadsheet tool, guideline and methodology have been investigated, and the overall methodology was found useful in a Danish context. The analysis is therefore carried out in accordance with this, applying the six-step approach recommended in the above mentioned publication.

The objective with groundwater protection is to secure or achieve a given groundwater quality in the upper groundwater. In order to make this operational in term of the analysis, it has been agreed to focus on concentrations in the soil water seeping from the lower part of the root zone. Pollutants in question are nitrate and pesticides. The target values are: 50 mg/l NO3- and no pesticide in soil water leaving the root zone.[11]

Wider impacts of initiatives for groundwater protection

Based on the delimitation accounted for in the introduction to this document a brief study will be conducted of the wider impacts of initiatives for groundwater protection. The purpose of this is to give an input to the choice of initiatives, not only based on the direct costs, but also so as to take the wider, welfare-economic effects into consideration.

A narrow CEA includes all costs and benefits that can be valued by market prices

The study will include:

- Identification of main area-based initiatives;
- Identification and description of direct costs and revenues;
- Identification and description of wider impacts;
- Literature study of analyses of the identified wider impacts implemented in a Danish context;
- Methodology description of how to include the results in planning;
- Reporting

Methodology

This study applies narrow and an extended CEA. A narrow CEA includes all costs and benefits that can be valued by market prices, and relate the net costs to the effect of the groundwater protection measure (GWPM) – which is measured by the unit Kg N removed/ha or Kg Pesticide removed/ha. In this study, three GWPM have been applied, and the output of the analysis is a cost effective

---

[9] See Aalborg Kommune (2001)

[10] Entec (2006)

[11] This approach was agreed upon by the project partners at the Oldenburg meeting 20. December 2006.

ranking of these measures. Wider impacts[12] are identified and mentioned but not included in the narrow CEA. Please see section 2.5 above for an introduction to welfare economic CEA, which have been applied in this study.

CEA – both narrow and extended, deals with the different choices possible, and in that sense works better as a tool when different measures provide different degrees of fulfilment

An extended CEA puts a monetary value on the wider impacts and include them in the analysis. Thus the wider impacts are treated in the same way as direct or indirect effects, i.e. the cost of lost agricultural production. Output is – as in the narrow CEA – a ranking of the measures based on cost effectiveness. As such, In order to carry out an extended CEA, environmental valuation of the relevant wider impacts is needed, which is generally costly to carry out, and furthermore it is difficult to generate reliable results. However, the result of a narrow CEA might be a ranking which do not reflect the true welfare of the society, something the broader perspective of the extended CEA tries to include. Please see section 2.6 above for a more elaborate introduction of inclusion of wider impacts to the CEA framework.

The difference between a Cost Benefit Analysis (CBA) and an extended CEA is small. The difference between the two methods is whether a monetary value is put to the effect, which should be reduced or not. To get the full picture of the socio-economic effect of a given measure, a CBA is a more obvious choice. On the other hand, CEA – both narrow and extended, deals with the different choices possible, and in that sense works better as a tool when different measures provide different degrees of fulfilment of an environmental objective at different net costs. In particular in terms of inclusion of wider impacts, the CEA leaves space for methodological freedom, as focus is solely on the ranking of the measures, and not – as in the CBA – on the absolute values.

3.1.3 Area and measures

Introduction

Initially two case areas were identified; the Aalborg South East catchment – with ongoing implementation of a groundwater protection plan adopted in 2005, and the Drastrup catchment – with a nearly implemented groundwater protection plan. The Aalborg Southeast case provides up to date data, while the Drastrup case provides both present and historical data, as well as experiences and data on wider impacts. The overall idea is to show the two different political and legislative situations before and after the new groundwater protection act (GPA) adopted in 1998. Before, it was possible to buy land for multiple purposes – recreation, nature development and groundwater protection. Now it's only possible to spend water consumer money on the least extensive measure necessary to meet the target. Effectively this means no further than cultivation declaration, while buying land – even on voluntary basis – is considered to be more severe. However, in order to create three comparable measures, Drastrup was chosen as the main case and knowledge from South East was brought in to qualify the measure settings. Drastrup groundwater catchments, is situated southwest of Aalborg, partially including the two small townships; Frejlev and Drastrup. During the last two decades the area has been transformed from regular farm land and a number of quarries, (extraction of raw materials) to a more diverse and extensified landscape.

---

[12] Generally measures have three kinds of effects: Direct, indirect and external. A direct effect is happening as an immediate consequence of the measure (e.g. planting of trees in an afforestation project). Indirect effects can be connected with the measure, but do not happen as a direct consequence (e.g. transport costs related to the planting of trees). External effects due to wider impacts of the measure (e.g. improved recreational value), but they are not traded on a market, and hence have no market price attached to them. If they should be included in the economic analysis, they should therefore be valuated, using techniques from environmental economics.

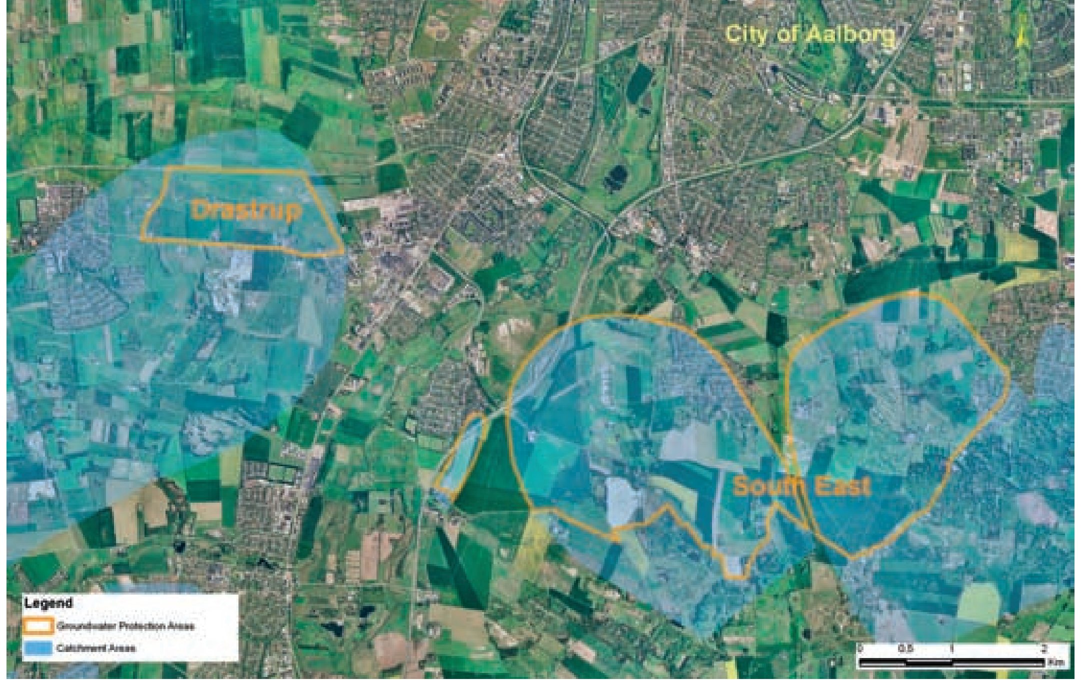


Figure 3-2
Location of projects

The plan included aspects of recreation, as well as nature protection and groundwater protection

The Drastrup Project was initiated before the new GPA, which includes compulsive groundwater protection plans, was put in place. The project was based on a holistic plan for activities in the area – the first of the kind in Denmark – and was initiated due to the need to find clean groundwater for abstraction. The plan included aspects of recreation, as well as nature protection and groundwater protection. The project was carried through because of the many different interests coming together, the legislative situation and political willingness within the municipality. The project was entirely based on voluntary agreements and land swapping, which was a rather unique approach, and an approach that no longer is possible, due to the interpretation of the GPA. Even though most people agree that the project has provided a number of wider social, environmental and nature benefits including clean water.

With the new GPA in 1998 the possibility to carry through land use change as part of ground water protection was formalised. As a starting point the land use change shall be based on voluntary agreements with full economical compensation for affected landowners. If voluntary agreements can not be achieved, the local authorities can force expropriation towards future land use to the necessary extend – still fully compensated and with tax exemption.

Compensations are financed via consumer's payment. Because of that, money can solely be spend on groundwater protection and not on recreation, nature or any other non groundwater related issues.

All three measures are designed to meet the same target, and will effectively result in some sort of land use change

Because of these facts the most commonly used measure is cultivation declaration to achieve the needed land use change. At present there are ongoing negotiations with landowners in the South East catchments.

For the purpose of this study, three area based measures for the Drastrup area were defined. The three measures for the Drastrup area are:

- Measure 1: Afforestation by buying land
- Measure 2: Permanent Grass by buying land
- Measure 3: Permanent Grass by declarations

In measure 1 and 2 it is assumed that the municipality buys the agricultural land from the farmers in order to implement the measures, while in measure 3 it is assumed that the farmers accept a cultivation declaration which state that the agricultural land should be converted to permanent grassland. Measures 1 and 2 are pretty much measures used before the GPA, and measure 3 what is possible within the framework (limitations!?) of the GPA. It should also be noted that all three measures dealt with in this case, has been put to use as part of an EU-life project, and as part of implementation of the groundwater protection plan.

Effectiveness of measures

As mentioned in the introduction, focus is on two different land use change measures and a legislative tool. All three measures are already tested on a 1:1 scale and have provided us with basic experiences, as they were used in relation to the Drastrup area.

All three measures are designed to meet the same target, and will effectively result in some sort of land use change. The two first measures, afforestation and permanent grassland, were initiated by simply buying the land, by which diffuse pollution from pesticide spraying or manuring was stopped immediately. The last measure is negotiated with the farmer and aims at meeting the target, which effectively means reducing and not stopping pollution.

On this basis measure 1 and 2 does not just meet the target, they have a built in security buffer and will – given the same preconditions, not be as sensitive to external variations e.g. arial deposition, flooding of meadows etc.

Afforestation without pesticides does, however, demand a serious relay – deep ploughing, which results in a temporary leaching of nutrients from the root zone. Alternatively a more sensitive but laborious and expensive cultivation can be chosen to avoid this. When forest is established it is a stable ecosystem that will accumulate nutrients, and harvesting methods can be designed to remove nutrients.

	Low estimate (kg N/ha)	High estimate (kg N/ha)
Afforestation	46	68
Permanent grass	20	34
Cultivation declaration	20	34

Table 3-1
Impact on nitrate from measures

Permanent grassland is just as efficient as afforestation, but is dependent on yearly management either grassing or hay harvest to continuously remove nutrients and meet the management duties for permanent grassland under the nature protection act – the same act that after 7 years forbids relay, sowing, tilling, spraying or other soil and plant disturbances, and thereby protects it even further.

Cultivation declarations are designed to meet the target and the ownership of the land remains with the farmer. It is still possible to do some extensive farming, mainly grassing and hay harvesting, and a professional farmer will normally use the area to its maximum to meet the different farming demands and regulations. This gives a more uncertain protection with no buffer as described earlier. It is also a situation that calls for, at least, annual control to ensure the restrictions and regulations within the declaration are met.

The table 3.1 shows the estimated impact of the three measures on nitrate, providing a low and a high estimate for each measure.

The table shows that afforestation has the largest N reduction potential, while permanent grass and cultivation declarations have potentials in the same range. All 3 measures provide complete removal of pesticide emissions.

The table shows that afforestation has the largest N reduction potential

Table 3-2
The different cost and income elements in the 3 measures

Afforestation	Cultivation declaration	Permanent grass
Dead weight loss from the sales value of agricultural land	Cost of cultivation declarations	Dead weight loss from the sales value of agricultural land
Cost of lost agricultural production	Cost of lost agricultural production	Cost of lost agricultural production
Income from game hunting	Income from game hunting	Income from game hunting
Earnings from afforestation	Earnings from permanent grass	Earnings from permanent grass
Administrative costs	Administrative costs	Administrative costs

Costs and income from implementing the measures

The table 3-2 & 3-3 summarises the various cost and income elements, which are included in the narrow CEA.

The cost estimates are based on best available data. All estimates are 2006 figures and express the welfare economic costs. This means that all prices are adjusted with the three conversion factors used in Danish welfare economic CBA and CEA: The Standard Conversion Factor, the Standard Conversion Factor for internationally traded goods, and the deadweight loss from public financing. These are explained in more detail in the methodology section (Section 2.5).

Selling of agricultural land

In measure 1 and 2 it is assumed that the municipality buys the agricultural land from the farmers.

This analysis looks at the welfare economic consequences of the 3 measures. A welfare economic analysis does not include the sales value of the agricultural land or the compensation costs. These costs are as such not a social cost, but a transfer between the municipality and the farmer. However the financing result in a welfare loss (dead weight loss) which is included. The dead weight loss can be estimated as 20 % of the public expenditures[13].

Since the buying of agricultural land is financed through taxes, the deadweight loss of 20 % should be added, amounting to DKK 35.100 (EUR 4.710).

---

[13] The public expenditures are corrected for the net tax factor

DKK (EUR)	Measure 1	Measure 2	Measure 3
Sale value of agricultural land per hectare	175.500 (23.551)	175.500 (23.551)	0
Dead weight loss per hectare	35.100 (4.710)	35.100 (4.710)	0

Table 3-3
Welfare economic sale value of agricultural land

Cultivation declarations

In measure 3, it is assumed that the farmers accept a cultivation declaration which state that the agricultural land should be converted to permanent grassland. In return the farmers receive an up-front compensation. From past experience it is estimated that the farmers on average receive DKK 163.800 (EUR 21.981) per hectare in compensation[14]. This amount is higher than farmers would receive if the land was sold. However, the farmer is obliged to cover any costs due to the permanent costs of having permanent grass lands in the future.
Since the compensation paid to the farmers is a money transfer that takes place between agents within the nation, which provides the physical limitation to the analysis, this have no impact on the aggregated welfare. For this reason, the only effect to be included in the analysis from the compensation payments is the dead weight loss stemming from price distortions of (income-) taxes, which finances the compensation payments. As for the land purchase, this loss is estimated to be 20 % of the payment.

Lost agricultural production

Nutrients and agricultural land are both inputs to the land-based production process, and as such provides for production of agricultural products. Changing the use from agriculture to other uses will have effects in form of lost income from the production, which consequently will disappear.

---

[14] This amount expresses the welfare economic costs of cultivation declarations.

Table 3-4
General estimates of economic rent from agriculture in DKK per Hectare (DKK, 2006-level)

DKK (EUR)	Crop production	Crop and livestock production
Denmark	4.672 (627)	5.812 (780)
Eastern Denmark, all soil types	4.562 (612)	5.576 (748)
Western Denmark, all soil types	4.705 (631)	6.139 (824)
Denmark, loamy soil	4.744 (637)	6.017 807)
Denmark, other soil types	4.335 (582)	5.649 (758)

Costs of eliminating agricultural activities or forestry production are measured in terms of changes in economic land rent. In areas with considerable livestock production it can be necessary also to include losses from reduced livestock production due to policy measures requiring certain number of area units per animal equivalent.

For a specific area, the economic land rent is the remuneration of the production factor land measured as the residual after all other costs have been subtracted from the gross output from the area in question. Hence, when eliminating agricultural activities the economic land rent is the difference between the value of the crop (market value) and the total production costs. The production costs encompass capital investments, labour, fertilizers, pesticides, seeds, maintenance, etc. For this reason the economic land rent depends both on the soil type and of these parameters.

The National Environmental Reseach Institute (NERI) in Denmark made in 2005 a study[15] with the purpose of developing guidelines for assessing the costs (loss of economic rent) of changing agricultural activities to non-productive land-uses. One of the outcomes of the study was general estimates of the economic land rent. In this study the costs of eliminating the agricultural activities are based on these general figures, which are shown in the table 3-4.

The first column represents the economic land rent from crop production. This estimate should be used when the extensive land-use have no consequences for the livestock production. This is the case when only a small part of the land available is changed to extensive use or when the number of livestock per hectare is low. Column 2 represents the economic land rent from crop and livestock production. This estimate should be used if the extensive land-use influences the livestock production.

---

[15] See Schou et.al (2005)

The study by NERI shows that the differences between the economic rents in Eastern and Western Denmark and between different soil types are limited. Thus, the gain by doing a detailed listing based on soil type and region is limited. The present study uses the general estimates of the economic land rent as a starting point, i.e. a loss at DKK 6.139 (EUR 824) per year per hectare of agricultural land given up.

An important aspect of the calculation of the economic land rent is the effect of the reform of the Common Agricultural Policy (CAP). The general economic land rents above are based on historical data which means that the EU agricultural subsidies are included. The strength of this approach is that the used data reflects reality. However, decoupling of the support from the volume of production will have an effect on the size of economic land rent and therefore the costs of the measures.

Income from game hunting licenses

A result of stopping agricultural production is that the hunting possibilities will be increased. Generally the income from game hunting licenses is higher in forests and from permanent grassland than from licenses on agricultural land. Also the income from licenses is higher in forests than on permanent grasslands.

Furthermore, game hunting on private land is more valuable than game hunting on public land. This is because of the assumption that hunting will affect the recreational experience negatively. Hunting on public land are therefore more restricted than on private land, i.e. less hunting days are permitted. A consequence of this is that the game hunting licenses are less expensive in public forests.

As an average for the value of increased hunting conditions an estimate of DKK 230 (EUR 31) is used for forests[16]. The increase will be less significant for permanent grass land. Several studies have assumed that the value is about half the value of game shooting licenses in forests. Thus a value of DKK 140 (EUR 19) will be used on private grass land and DKK 120 (EUR 16) on public land.

The income from game hunting licenses is higher in forests and from permanent grassland than from licenses on agricultural land

---

[16] Miljøstyrelsen (2005)

DKK (EUR)	**Economic rent per hectare** (2006-DKK)
Oak	-1.803 (-242)
Beech	-1.821 (-244)
Ash	-918 (-123)
Birch	300 (40)
Common spruce	288 (39)
Pinus sylvestris	-648 (-87)
Thuja	-1.559 (-209)
Abies	11.089 (1.488)
Weighted average	**-526 (-71)**

Table 3-5
Economic rent
for afforestation

Earnings from afforestation

In the first measure it is assumed that 75 % of the area will be planted with forest. The remaining 25 % of the area is open grasslands, public facilities, etc. It is assumed that the area will be planted using different species with 66 % oak and beech, 9 % ash and birch and the remaining 25 % is conifer wood. It is also assumed that the forestry is continuous which means that either regeneration or new trees are planted after logging.

It is possible to estimate the cost of afforestation by the economic rent. A study[17] estimates the income from tree production. They find that the economic rent is positive for common spruce and abies. The weighted average economic rent is DKK -526 (EUR -71), cf. the 3-5 table.

In welfare economic analysis EU subsidies should also be taken into account. However the new Rural District Programme has just changed the possibilities of receiving afforestation subsidies. Today it is only possible for private owners of agricultural land to receive subsidies. It is no longer possible for public authorities – in this case the Municipality of Aalborg – to receive subsidies.[18]

As an estimate for nature conservation a rough estimate of DKK 180 (EUR 24) per hectare is used. This estimate is among others used in cost-benefit analysis of Vollerup forest (Møller et al. 2000).

[17] See Damgaard et al. (2001)
[18] Danish Forest and Nature Agency (2007).

Earnings from permanent grass

The economic rent from permanent grass (no use of fertilizers and pesticides) arises because of sheep and steer production. It is assumed that the animals graze during summer and spend winter in deep bedding stables. Apart from the income from meat sales the economic rent takes costs of winter fodder, labour, veterinary costs, transport and fencing into account.

NERI has made some general estimates[19] of the economic rent from permanent grass. The economic rent depends on the alternative use. If the land does not have alternative usage – which means that the alternative value is zero – then the costs of summer coarse fodder is zero. If on the other hand the land has an alternative value e.g. because it is possible to have an income from crop production then the value of summer coarse fodder should be taken into account. In measure 2 it is assumed that the municipality buys the land and in measure 3 it is assumed that the farmers receive compensation in return of converting the agricultural land to permanent grass land, due to the declarations. In both cases normal agricultural production is no longer possible and the land has no alternative value. Hence, a welfare economic rent of DKK -911 (EUR -122) is used. The negative economic rent indicates that it is costly to have permanent grass areas.

[19] See NERI (2005); Hasler et.al. (2004); and Schou (2003)

DKK (EUR)	**Welfare economic rent**
No alternative value	- 911 (-122)
The land has an alternative value	- 5.519 (-741)

Table 3-6
Welfare economic
rent per hectare
(2006-kr.).

Table 3-7
Welfare economic
administrative costs

DKK (EUR)	Measure 1: Afforestation – Land purchase	Measure 2: Permanent grass – Land purchase	Measure 3: Cultivation Declaration	Unit
1.Brokerage	35.100 (4.710)	35.100 (4.710)	30.000 (4.026)	Initial expense per land owner
2.Negotiation costs		-	50.000 (6.710)	Initial expense per land owner
Subtotal	*35.100* *(4.710)*	*35.100* *(4.710)*	*80.000* *(10.735)*	*Initial expense per land owner*
3. Management	585 (79)	585 (79)	-	Annual expense per hectare
4. Inspection	6 (0.8)	410 (55)	878 (118)	Annual expense per hectare
5. Monitoring	6 (0.8)	12 (1.6)	59 (8)	Annual expense per hectare
Subtotal	*597* *(80)*	*1007* *(135)*	*937* *(126)*	*Annual expense per hectare*

Administrative costs
Table 3-7 show the estimated administrative costs caused by implementing each of the three measures.

The table shows a breakdown of the administrative costs into initial and annual expenses. For each of the measures initial payments are brokerage and negotiation costs (the latter only being relevant for measure 3). Annual expenses are subdivided into management of the measure and inspection and monitoring of the area. For each measure the shown sub-totals are included in the analysis as welfare economic costs, which in turn are corrected for dead-weight loss.

Table 3-8
Annual welfare
economic costs
and revenues
in measure 1.

DKK (EUR)	Measure 1: Afforestation
Number of hectares	650 where 75 % is afforestated
Dead weight loss from the sales value of agricultural land (20 % of the sales value)	-684.450 (-91.848)
Nature conservation not afforested areas (NPV of DKK 180 (EUR 24) for 162,5 hectares)	-34.223 (-4.592)
Cost of lost agricultural production (NPV of DKK 6139 (EUR 824) per year for the 650 hectares)	-3.990.350 (-535.474)
Income from game shooting (NPV of DKK 230 (EUR 31) per year for 650 hectares)	152.100 (20.411)
Earnings from afforestation (NPV of DKK -526 (EUR -71) for 487,5 hectares)	-256.425 (-34.410)
Administrative costs including dead weight loss (Initial expense at DKK 35.100 (EUR 4.710) and NPV of annual expense at DKK 597 (EUR 80), plus deadweight loss)	-528.606 (-70.935)
Total	*-5.341.954 (-686.232)*
Reduced N emission (low estimate) kg N/hectare	39,5
Reduced N emission (high estimate) kg N/hectare	90,9
Pesticides (kg/hectare)	1,45
Cost per tonnes of reduced N (low estimate)	*208 (28)*
Cost per tonnes of reduced N (high estimate)	*91 (12)*
Cost per tonne of reduced pesticides	*5.668 (752)*

3.1.4 Narrow CEA

It is possible to calculate the costs of the 3 measures by the difference in economic land rent before the measures are implemented and after the measures are implemented. The costs should be corrected for increased hunting value, administrative costs, etc.

The 3 tables 3-8 to 3-10 summarises the costs presented in chapter 3.

In all tree measures the only income arises from game shooting. All other costs are negative. In the first measure the expected nitrogen reduction is corrected for that only 75 % of the area is forest.

Table 3-9
Annual welfare economic costs and revenues in measure 2.

DKK (EUR)	Measure 2: Permanent grass
Dead weight loss from the sales value of agricultural land (20 % of the sales value)	-684.450 (-91.848)
Cost of lost agricultural production (NPV of DKK 6139 (EUR 824) per year for the 650 hectares)	-3.990.350 (-535.474)
Income from game shooting (NPV of DKK 120 (EUR 16) per year for 650 hectares)	76.050 (10.205)
Earnings from permanent grass (NPV of DKK -911 (EUR -122) per year for 650 hectares)	-592.150 (-79.462)
Administrative costs including dead weight loss (Initial expense at DKK 35.100 (EUR 4.710) and NPV of annual expense at DKK 1.007 (EUR 135), plus deadweight loss)	-848.929 (-113.920)
Total	*-6.039.829 (-810.498)*
Reduced N emission (low estimate) kg N/hectare	20
Reduced N emission (high estimate) kg N/hectare	34
Pesticides (kg/hectare)	1,45
Cost per tonnes of reduced N (low estimate)	*465 (62)*
Cost per tonnes of reduced N (high estimate)	*273 (37)*
Cost per tonne of reduced pesticides	*6.408 (860)*

Figure 3-3
Ranking of the 3 measures based on DKK per reduced N.

The results depend on the expected nitrogen reduction. The higher reduction the lower the cost per kilo reduced nitrogen. The cost estimates are significant lower for measure 1 than from the two other measures. This is the case both the expected nitrogen reduction is low and when it is high. In the first measure the cost per kilo reduced nitrogen is between DKK 91 and 208 (EUR 12 and 28). In the second and third measure the costs are DKK 273 to 465 and DKK 272 to 462 respectively (EUR 37 to 62 respectively for both measures). This means that if the measures are ranked in accordance to nitrogen reduction afforestation will be preferred, while public owned permanent grass land and cultivation declarations are equal.

Table 3-10
Annual welfare economic costs and revenues in measure 3.

DKK (EUR)	Measure 3: Permanent grass (cultivation declarations)
Dead weight loss from cultivation declarations (20 % of compensation payments)	-638.820 (-85.725)
Cost of lost agricultural production (NPV of DKK 6139 (EUR 824) per year for the 650 hectares)	-3.990.350 (-535.474)
Income from game shooting (NPV of DKK 140 (EUR 19) per year for 650 hectares)	91.260 (12.246)
Earnings from permanent grass (NPV of DKK -911 (EUR -122) per year for 650 hectares)	-592.150 (-79.462)
Administrative costs including dead weight loss (Initial expense at DKK 80.000 (EUR 10.735) and NPV of annual expense at DKK 937 (EUR 126), plus deadweight loss)	-874.080 (-117.295)
Total	*-6.004.140 (-805.709)*
Reduced N emission (low estimate) kg N/hectare	20
Reduced N emission (high estimate) kg N/hectare	34
Pesticides (kg/hectare)	1,45
Cost per tonnes of reduced N (low estimate)	*462 (62)*
Cost per tonnes of reduced N (high estimate)	*272 (37)*
Cost per tonne of reduced pesticides	*6.370 (855)*

Since the 3 measures provide the same number of tonnes pesticide removal (complete removal of pesticides), the most cost-effective measure is simply the cheapest. Ranking based on pesticides is therefore in this case the same as for nitrate, which is shown in the figure.

It should be emphasized that this is a welfare economic analysis. The result may not be the same as the result of a financial analysis.

Figure 3-4
Ranking of the three measures based on DKK per tonne of pesticide removal

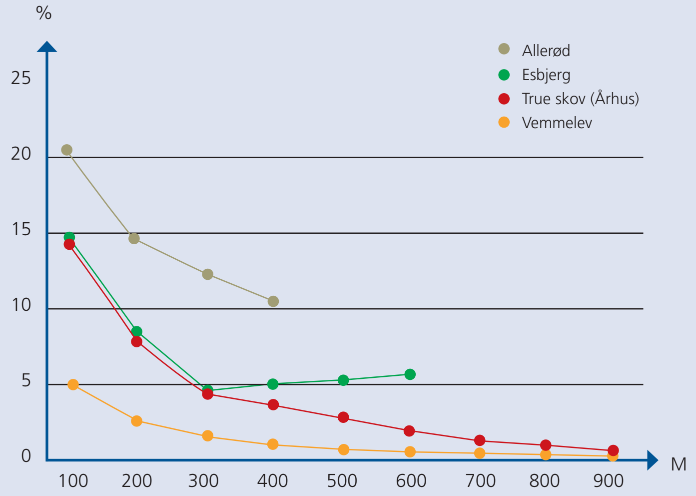


Figure 3-5
Different studies estimates of how the house prices are influenced by proximity to a forest

3.1.5 Wider impacts

Please see section 2.6 above for an elaborate introduction to notions and concepts relating to environmental valuation, benefit transfer, and the approach applied to include wider impacts in the CEA framework.

Relevant studies and quantification

In this section relevant results of Danish valuation studies will be presented. There are no Danish studies that examine the value of increased biodiversity, erosion control or diffuse pollution as a consequence of afforestation or permanent grass. If these values are included in an economic analysis they are often only included qualitatively.

Increased recreational opportunities from afforestation

The recreational value arises because it is easier to get access to the forest than before where the land was under agricultural production. For example there are better conditions for walking and biking trips, primitive overnight stays, watching of flora and fauna, collection of fungus and berries, picnic, dog walking, horseback riding etc. An essential parameter for the size of the recreational value is the size of population including the location of the forest in relation to developed areas. The exact value depends on the actual use and the expected number of visitors, cf. below. The value also depends on the situation of substitutes. If the afforestation area is close to similar areas then the marginal value of the new forest will be very small.

It is possible to estimate the recreational value of afforestation by the hedonic price method. There exist several Danish studies which have estimated the recreational value by using this method. All studies show that the house price is affected positively by proximity to a forest. The figure below shows how large a percentage of the house prices that is attributable to the forest. From the figure it can be seen that up to 20 % of the house price of houses that lie within 100 meters from the forest edge can be attributable to the forest.[20]

---

20 Damgaard et.al. (2001); Hasler et.al. (2002)

This means that the recreational value of the forest is highly dependent of the house prices and the number of houses. Hence a forest in urban areas with high house prices has a higher value than a forest in the countryside.

One study has estimated the recreational value of afforestation in Drastrup. The study finds that the house prices are positively affected already in the planning phase. The study concludes that the recreational value of Drastrup is DKK 146.000 (EUR 19.592) per hectare.21 This figure will be used in the present analysis.

However the hedonic method does not capture the recreational value from persons that does not live close to the forest. The contingent valuation method has been used to estimate the total value of forest visits to DKK 164 (EUR 22) or DKK 5 (EUR 0.7) per visit.

One study has investigated the influence of the tree species composition. They found that the recreational value is affected positively when a coniferous forest is converted to a beech forest or to a forest with different tree species. Besides this the recreational value will be affected positively with selected cutting and if the forest contains a few big dead trees.

Increased recreational values from permanent grasslands

The recreational value depends on public access

The recreational value depends on public access. If there is no public access as in the case with cultivation declarations the recreational value is zero. When the public have access, the recreational value of permanent grass lands are positive. However there exist no Danish valuation studies directly on permanent grasslands. It is therefore guesstimated that the value is positive, but not larger than for forests.

Environmental consequences – CO2 storage

The values of environmental consequences are linked to the protection of groundwater and the reduction of ammonia and greenhouse gasses. The scope of this study is to evaluate different cost effective measures to protect drinking water by reducing the emission of nitrogen and pesticides. Therefore the value of drinking water is not included. There is widespread uncertainty of how the different environmental consequences should be valued. One way is to value the reduction by estimating of how much it cost to reduce 1 kilo CO2 by alternative means.

Broadleaved wood bind in average 8 tonnes of CO2 per year per hectare over a period of 90 years and coniferous wood bind in average 14 tonnes of CO2 per year per hectare over a period of 50 years22. After this period the net CO2 binding is zero. These figures can be taken to represent a lower bound. According to the Intergovernmental Panel on Climate Change the CO2 binding is 25 % higher. As a starting point for

---

21 Hasler et al. (2002).
22 According to Dubgaard et al. (2003) and Møller et al. (2000)

the value of CO2 the Danish Environmental protection Agency recommends DKK 180 (EUR 24) per tonne. This value reflects the expected future allowance price.

During the research process it has not been possible to find sources on CO2 binding of permanent grass. It is however assumed that permanent grass is binding CO2, but to a lower level than forests.

Option and non-use values from afforestation

The option and the non-use value comprise the value of that it will be possible for current generations to use the forest in the future and also that it is possible for future generations to use the forest. These values can be valued by using the hypothetical methods. However there are no Danish studies that have estimated this and the existing international studies mainly focus on unique nature locations or threatened animal species and not ordinary forest areas. Furthermore, the question is how large this value is for a single afforestation area. It is likely that there is already another forest close by and as a consequence a minor change in the total forest area will only have a minimal influence on the size of the option and non-use value. Because of this the option and non-use value are often only described qualitatively in valuation studies of afforestation.

The approach suggested to include wider impacts to the CEA in this case gave a consistent and clear answer

A cost-benefit analysis for Drastrup transfers the estimated existence value from a nature conservation project in Sommerset Levels plus Moors. If the estimates from Sommerset Levels plus Moors are corrected for differences in number of households and size of project area they find that the present existence value for Drastrup is about DKK 45 million (EUR 6 million). However the characteristics for Sommerset Levels plus Moors and Drastrup are only partly similar. It is only the main biological features which are comparable.

Option and non-use values from permanent grasslands

There are no Danish studies examining the option value or non-use value from permanent grass. However, there are several studies that look at the existence value of extensive farming. All studies are made for well known and unique localities and cannot be regarded to be representative for permanent grass on ordinary agricultural land.

	Measure 1	Measure 2	Measure 3
Narrow cost-effectiveness	Good	Medium	Poor
Recreational value	+++	+	+
Biodiversity	+	+	+
CO2 binding	++	+	+
Option and non use value	-	-	-

Note: +++ is relatively high value and – means no significant value.

Table 3-11
Narrow CEA (N)
and wider impacts

Relating results of studies to the narrow CEA

The table 3-11 summarises the results from the narrow CEA and the analysis of wider impacts (extended CEA).

Table shows that in this case the results of the analysis of the wider impacts points to the same direction as the result of the narrow analysis, namely that measure 1 should be preferred, while scenarios 2 and 3 have wider impacts at the same level. For this reason, using the approach described at the outset of this chapter, no further analysis of wider impacts is needed.

3.1.6 Conclusions

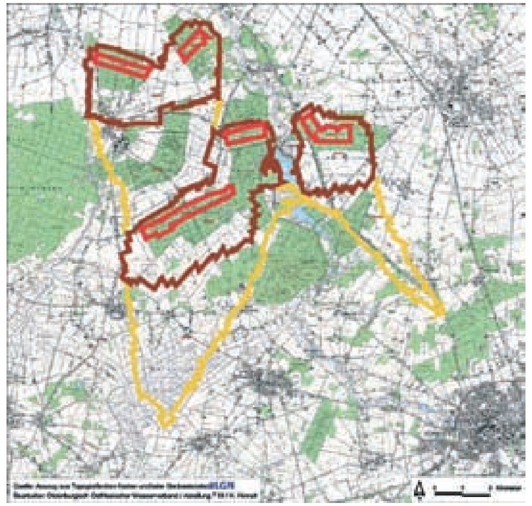
We set out this study with the purpose to cast light on the implications of wider impacts to cost-effective ranking of measures aimed at ground water protection.

In order to do so, a "narrow" CEA for the nitrate and pesticide reducing measures afforestation and permanent grass were carried out, using the general welfare economic methodology as suggested in Entec (2006), combined with inclusion of two conversion factors, which have become common to use in a Danish context (the net tax factor and the dead-weight loss).

By using this approach to the narrow CEA, it was found that afforestation was the most cost-effective measure to reduce nitrate as well as pesticides.

The question to be answered by including wider impacts to the analysis was whether these would challenge the ranking as suggested by the narrow analysis. Based on a literature study of relevant valuation studies carried out in a Danish context, a modified benefit transfer was carried out. This analysis showed that afforestation also could be expected to be the measure with the largest wider impact and in turn the largest broad economic benefits.

As such, the approach suggested to include wider impacts to the CEA in this case gave a consistent and clear answer.



3.2 Germany – Elements of Cost-effectiveness Analysis

3.2.1 Background and purpose

The pilot plot Thülsfelde in the northwest of Lower Saxony has the status as WPA since 1996. Voluntary Agreements were introduced to farmers more than 10 years ago. Therefore lots of data are available regarding nitrate concentration below different crops, amount of percolation water and nitrate loading. The baseline scenario was given by 1628 t Nitrate loss as diffuse pollution for the WPA Thülsfelde. The average nitrate concentration was determined as 83 mg/l below the root zone. The aim was to reduce the nitrate concentration to 50 mg/l in the leaching water.

The purpose with the analysis carried out in this case study was to test the CEA approach as a way of finding the most cost-effective way of meeting this target for WPA Thülsfelde. In order to do this, different scenarios were investigated, including varying land use and measures.

The soils are mostly podzolic soils. The annual rainfall is about 750 to 800 mm. The groundwater recharge varies between 200 mm (forest) 250 mm (grassland) and 350 mm (arable land). The N-emission rate is quite high due to intensive animal husbandry in the region of Cloppenburg. At least 30 to 35 kg N/ ha can be stated as input by rainfall.

The different percentage of land use according to crop and crops with Voluntary Agreement of the total area of app. 8000 ha of the Water Protection Area Thülsfelde is shown below. 41% is arable land, 38% is used as forest and 7% remains for grassland.

Financing is available due to the water abstraction charge paid by the water consumer to compensate farmers for income losses and other expenses. In Lower Saxony on app. 13 % of the area water protection areas do have a protection decree. Therefore restrictions are made for farming practice and additionally a catalogue of measures is available to sign Voluntary Agreements between farmers and water supplier. Some of the land use options shown in table 3-12 are used as measures for the CEA and are described below.

Table 3-12
Current land use

Landuse in the water protection area of Thülsfelde	ha
Forest	2840
Afforestation	300
Fallow*	89
Grassland	244
Extensive grassland*	49
Organic farming on arable land *	197
Maize without Voluntary Agreements	764
Maize with reduced row spacing and less fertilizer*	168
Maize with 100 kg N/ha*	1
Cereals without Voluntary Agreements	950
Cereals with intercropping*	818
Potatoes	19
Gras on arable land*	52
Rape	14
Other agricultural land use	48
Other	1500
Sum	**8053**

* Voluntary Agreement

3.2.2 Description of available measures

The table 3-13 summarises the measures available in relation to the WPA Thülsfelde. The measures described are offered as Voluntary Agreements (except afforestation) between farmers and water supplier OOWV in the Water Protection area of Thülsfelde in Lower Saxony[23]. Details on costs and effectiveness of the respective measures are elaborated in below sections.

---

[23] For afforestation the official figures of the agri-environmental payments have been used. www.ml.niedersachsen.de/master

Table 3-13 Available measures

	Suitability	Acceptability	Calculation basis
Afforestation	Afforestation is suitable for a variety of locations and soils. If broadleaf trees are used, the nitrate loss is much lower than under old coniferous stands (app. 11 mg NO3/l to 55 mg NO3/l)	In Germany farmers mostly do not like to sell farming land, which will be afforested afterwards. According to law a way back to farmland is not allowed, because the percentage of forest is still too low	724 € per year and hectare
Active fallow plot greening	Very efficient	Is getting less, because the demand for set asides will be abolished in 2008 and because of requirements of energy farmers and because of higher prices for cereals	300 € per year and hectare
Permanent grassland	It depends on whether or not the farmers can use the grass for stock feeding or whether he can sell the grass to other farmers. Measure only makes sense if lasting longer than for 5 years contract time	Directly linked to suitability. Contracts can be signed for 5 years only	250 € per year and hectare
Extensive grassland	Not suitable for regions with intensive dairy farming and little possibilities to lease more grassland	Good in regions with extensive grassland Time horizon: Only for one year but also for more than a year	350 € per year and hectare
Promotion of Organic Farming	Comparing the organic land use with normal land use under similar conditions (crops/ soils/ rainfall) the nitrate loss is definitely less under organic plots	It depends on the decision of the farmers. Only few farmers will decide to follow the EU rules of organic farming or the stricter rules of the national organic farmers associations. The decision depends further on the nearness to the market, the prices for the product he can sell and the incentives from EU and national government. It also depends whether the stables can be modified easily to fulfill the requirements	300 € per year and hectare
Maize with reduced row spacing an N-input below normal farmers practice	Very good results in N loss reduction in comparison to 180 kg N input	Very good and increasing acceptance among the farmers	50 € per hectare and year
Maize with limited fertilizer (100 kg N/ha)	Maize shows much acceptable residual mineralized nitrogen in autumn when N- application is reduced to 100 kg. The reduction in yield is approximately 10% less. According to field- investigations nitrate concentration will not exceed 50 mg/l in percolating water, having podzolic soils and 300 mm groundwater recharge as given in WPA Thülsfelde	No experience yet, but assumed that farmers are not very interested in	200 € per hectare and year for the difference in yield and additional money for further risks
Cereals with intercropping	For all locations and with very good results regarding minimizing N reduction	Very good. Time horizon: one year duration contracts	170 € per hectare and year
Integrated fertilizer and manure application	Efficient to make a prognosis regarding effectiveness of groundwater protection measures	Good acceptance	12.50 €
Temporal restrictions for manure spreading	The effect to spray slurry during times, when the following crop needs the fertilizer is obviously very good, because losses due to erosion and diffuse pollution are avoided	Moderate to good	25 € per hectare and year
Towed umbilical hose/ slit injection	All locations, on grassland only trailing shoe or slit injection	Growing acceptance	30 € per hectare and year

3.2.3 Assessment of costs of measures

Cost figures for individual measures can be established in different ways. For example in the German case, cost figures rest on a unified view taking both capital and recurrent cost estimates together. It has to be considered, though, that full financial cost estimates for measures are difficult to establish, since the landowners and farmers tend to follow a variety of enterprises and capital costs mostly present over-heads whose share in financing individual measures varies and are not easily tractable, even for the business owners. In addition, such capital over-heads in most cases have to be borne by farmers and landowners anyway. Due to the specificity of investments large proportions of such costs may also be seen by landowners and farmers as sunk costs, which are no longer relevant for economic decisions.

The cost calculations in the German case have been based on full cost indicators

Overall full costs including recurrent and capital costs tend to be rather well reflected in the cost estimates for example for organic farming, since such measures affect whole enterprises. Still, in private investment decisions capital costs are commonly discounted while recurrent costs such as operational and maintenance costs may neither be discounted nor included. Overall, prospected capital costs of new measures have to be born by some entity and they should therefore be included in financial as well as economic cost-effectiveness analyses.

The cost calculations in the German case have been based on full cost indicators from the start on and no distinction between capital and recurrent costs has been made, since full financial costs are needed for cost-effectiveness analysis anyway. Recurrent costs, such as gross margin figures, are rather used for farm level management decisions and not for such policy decision-making problems and mid terms strategic decisions the project focuses on. However, farmers who have to adopt environmental measures may base their decisions on a comparison of per hectare gross margins of measures with those of their farming enterprises. Commonly operating and maintenance costs are considered to constitute recurrent costs. To this also opportunity costs, e.g. of farm income foregone may be added. Opportunity costs are, however, also relevant to capital costs as capital may be invested with greater returns elsewhere. A clear distinction between capital and recurrent costs may therefore not be possible in practice. However, conceptually recurrent costs are the variable cost components like casual labour inputs, seeds, fuel, etc. added to capital expenses like wears, ditches, buildings, machinery, etc., which are fixed assets at least in the medium term.

Administrative costs are not being considered in mainstream economic approaches. However, they are considered to be of major importance in certain economic branches, like contract theory and transaction cost economics, which are not as well established among economists as neoclassical microeconomic approaches. Consequently administrative costs have not been considered as cost components of measures in the German case. This is also due to the fact that these are difficult to establish and depend on success and transparency of the administrative processes, which may well be highly uncertain in relation to individual water protection measures. Furthermore such costs can be distributed among the parties involved, like public administration or private farmers in varying proportions. This is all depending on the legal framework, administrative procedures, monitoring and compliance requirements, responsibilities of the parties, contract design, economies of scale of administrative transactions etc.

Hence administrative costs are difficult to be readily allocated to individual measures. However, further and much more extensive research could have had established some indicative figures on administrative costs relating to the case study. Such research is not common in Germany, which is exacerbated by the fact that administrations do not have to make their cost structures sufficiently transparent. In the German case study, however, administrative costs have been considered as side benefits of organic farming, which is suggested to entail lower transaction costs, especially for public administrations (Beckmann et al. 2003, see case study). Also in the wider European context research on transaction costs is scarce. Important studies on transaction costs relating to farming and the environment, especially to environmental measures, however, include Vatn (2002) and Falconer and Saunders (2002), Rorstad et al (2007) as well as publications of the ITAES project under the 6th Framework Programme (http://merlin.lusignan.inra.fr/ITAES/website/Publicdeliverables). Still, such transaction costs are often difficult to express with sufficient precision in monetary terms and consequently most of these studies rest on comparative qualitative assessments.

Also relating to transaction costs are implementation costs, though they may not be readily conceptualised. If they consider only expenses of public and private administrations they would fall under administrative costs. They may on the contrary also be seen to encompass all costs incurred by all parties affected starting after the formulation of measures and ending at successful goal attainment, e.g. of water quality targets. The goals, however, would have to be maintained and it may well be argued that the relating costs would actually also have to be part of administrative costs of running water protection measures for achieving objectives over the whole time-span considered.

Administrative costs as part of transaction costs can be seen as important for the actual costs of measures.

In summary, administrative costs as part of transaction costs can be seen as important for the actual costs of measures. Procedures to elicit and allocate such cost with sufficient precision have not been established as yet, which may be due to the dominance of mainstream economic approaches in policy evaluation.

Taking the above into consideration, the costs of the considered measures were largely calculated on the basis of past experience in the region. In the Thülsfelde groundwater protection area, a number of groundwater protection measures have been carried out in the last two decades, often in the form of cooperative schemes. In these schemes, farmers contractually commit to apply certain measures that limit or reduce nutrient input, over and above the standards of good agricultural practice. Where such measures impose additional costs on farmers – be it in the form of investment costs or foregone income from set-aside land or extensive farming – farmers are compensated for these losses. The amount of compensation paid is centrally determined, based on assumptions about the average costs of implementing such measures. Cost estimations cover the costs of new investments (including depreciation) as well as the opportunity costs (foregone income). Participation in these schemes is voluntary, so that farmers can judge themselves whether the offered compensation will be sufficient to cover their costs. Thus, the calculations for the different measures were carried out as follows:

- For active fallow plot greening, conversion of arable land into permanent grassland or into extensive grassland, as well as for maize farming with reduced row spacing and cereals with intercropping, actual compensation payments on a per-ha basis were used.
- For organic farming, a certain area owned by the local water supplier is leased to farmers at a reduced rate. The cost of this measure was calculated as the difference between the reduced leasing rate and the standard lease rate in the area.
- For temporal restrictions on fertiliser, manure or slurry spreading, and for the set of technical measures (towed umbilical hose, trailing shoe or slit injection), the actual payments per ha were applied. For integrated fertiliser and manure application, where compensation is paid per farm rather than per ha, the per-farm payment of 250 Euro was divided by the approximate average farm size of 20 ha.
- For afforestation, the officical payments of the afforestation program in Lower Saxony were used. They consist of an initial financing of purchase of the seedlings and their planting and the yearly payments for the agri-environmental schemes. The initial planting is estimated to costs 7000 €/ha. With this one-off costs discounted at 3.5% over a 25-year-period, this results in an annuity of 424 Euro. To this, the costs of the annual premiums of 300 Euro per ha, as paid by the agri-environmental schemes in Lower Saxony, were added. Hence the total annuity of the measure afforestation sums up to 724 €/a.

Participation is voluntary, farmers can judge themselves whether the offered compensation will be sufficient to cover their costs

Land use in the water protection area of Thülsfelde	ha	Potential load kg N	Groundwater recharge mm	Nitrate concentration mg/l	Percolating water m3	Nitrate load kg
Forest	2840	25	200	55	5,680,000	314,530
Afforestation	300	5	200	11	600,000	6,645
Fallow*	89	19	300	28	267,000	7,491
Grassland	244	31	250	55	610,000	33,509
Extensive grassland*	49	19	300	28	147,000	4,124
Organic farming on arable land *	197	51	300	75	591,000	44,508
Maize without Voluntary Agreements	764	117	300	173	2,292,000	395,989
Maize with reduced row spacing and less fertilizer*	168	80	300	118	504,000	59,539
Maize with 100 kg N/ha*	1	33	300	49	3,000	146
Cereals without Voluntary Agreements	950	100	300	148	2,850,000	420,850
Cereals with intercropping*	818	43	250	76	2,045,000	155,821
Potatoes	19	79	300	117	57,000	6,649
Gras on arable land*	52	103	300	152	156,000	23,727
Rape	14	101	300	149	42,000	6,264
Other agricultural land use	48	71	200	157	96,000	15,097
Other	1500	20	250	35	3,750,000	132,900
Sum	**8053**				**19,690,000**	**1,627,790**

* Voluntary Agreement

Table 3-14
Load calculation

3.2.4 Assessment of effectiveness of measures

According to the data available and the experiences gained within ten years of cooperation with farmers, the actual load of the WPA Thülsfelde can be calculated as shown in the table 3-14.

In Thülsfelde we have an actual load of 45 kg N/ha. The average nitrate concentration is 83 mg/l and the aim is to have only 50 mg/l. By varying the ha-amount of the different land use options with measures available different scenarios are calculated to close the gap of the loading so that an average of 50 mg/l will be achieved (at least theoretically).

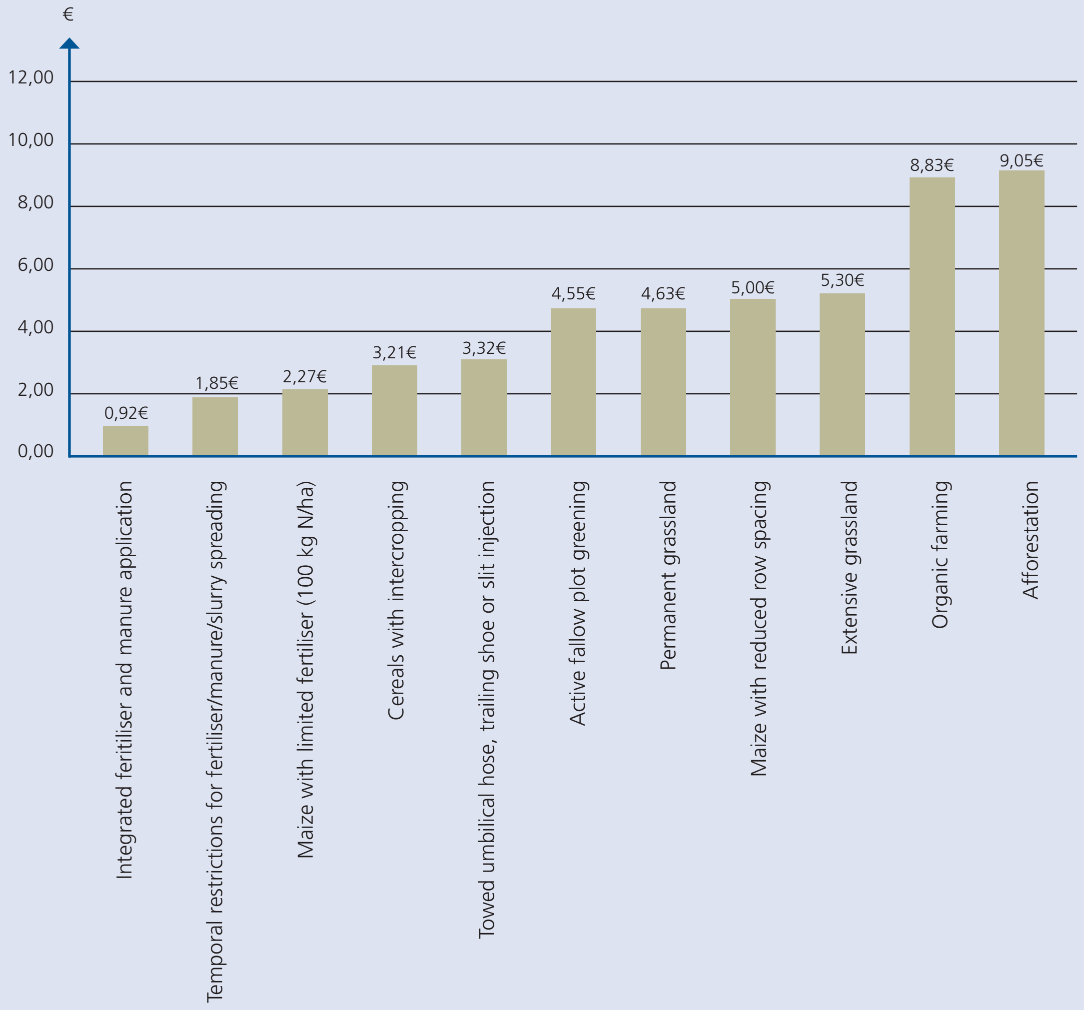


Figure 3-6
Cost-effectiveness
of groundwater
protection measures
(Euro/kgNred.)

Cost-effectiveness ratios for individual measures

Based on the above information, two types of cost-effectiveness ratios were calculated:

- Costs per percentage point of closing the calculated gap (for the calculated load reduction of 656 tons NO^3), and
- Costs per kg of avoided N load.

The two ratios basically convey the same information, and differ only in scale: one percentage point of closing the gap corresponds to 6,560 kg NO^3, or 1,480 kg N. Correspondingly, the cost per percentage of closing the gap is always 1,480 times the cost per kg N load removed.

The cost-effectiveness ratios allow a first comparison of the available measures. The picture that emerges is that measures aimed at improving agricultural practices (improved nutrient management and application) are clearly more cost-effective than measures involving land use changes. With the exception of reduced row spacing for maize, all measures aimed at agricultural practice rank higher than those involving land use changes.

It is clear that this is only a first, rough comparison of the available measures. In practice, there are a number of other factors that influence the desirability and applicability of a measure. This includes:

- The total available capacity for such measures: some measures may be relatively cheap, but their potential scope of application may be limited, hence their overall effect will remain small. This issue is addressed in the following chapter, which discusses possible combinations of the available measures.
- Certainty / irreversibility of the effect: for the different measures, it will be more or less certain whether they will be adopted, whether they will be implemented correctly, whether they will achieve the desired effect, and whether the measure can be reversed. For measures related to changed agricultural practices, such uncertainties tend to be higher than for land use changes.

The cost-effectiveness ratios allow a first comparison of the available measures

- Side-benefits of measures: the figure above ranks the available measures only in terms of their main effect, which is to reduce the N load to groundwater. In addition, some measures create additional benefits, both related to water quality and to the wider environment. These side-benefits are discussed in the following.

	Afforestation	Active fallow plot greening	Permanent grassland	Organic farming	Maize with reduced row spacing	Maize with limited fertiliser	Cereals with intercropping	Integrated fertiliser application	Temporal restrictions	Towed umbilical hose etc.
Afforestation	X	X	X	X	X	X	X	X	X	X
Active fallow plot greening		X	X	X	X	X	X	X	X	X
Permanent grassland			X	X	X	X	X	+	+	+
Organic farming				X	X	X	+	+	+	+
Maize with reduced row spacing					X	+	X	O	+	O
Maize with limited fertiliser						X	X	O	+	O
Cereals with intercropping							X	+	+	X
Integrated fertiliser application								X	+	+
Temporal restrictions									X	+
Towed umbilical hose etc.										X

+ = synergies from combined implementation

O = neutral

X = mutually exclusive

Figure 3-7
Possibilities and effects of combining the proposed measures

3.2.5 Combinations of measures

In the following step, the different available measures are combined (see figure 3-7). This is done on a spatial basis: for each measure, it is estimated on how many hectares the measure would be applicable in principle (maximum scope of application), and on how many hectares the measure might be implemented under realistic conditions. In doing so, possible combinations and mutual exclusiveness between measures are also considered: for example, improved fertiliser application through towed umbilical hose etc. can be combined with intercropping for cereals, while intercropping cannot be applied to agricultural land that has been converted to permanent grassland or to forest. The possible combinations are presented in the matrix above.

In calculating the combined effect of measures, two simplifying assumptions were made:

- Measures carried out in combination to one and the same ha of agricultural land have been treated as additive, that is, the effects of the two measures simply add up.
- For measures involving land use change, the effectiveness was calculated as the average of all arable land in the status quo situation. In principle, it makes a difference whether a conventional maize field or a cereal field with intercropping is converted to grassland. To keep the analysis manageable, such differences were blended out by using an average "representative hectare" as the basis for the comparison.

Measure ...	... carried out on an additional area of
Afforestation	20 ha
Active fallow plot greening	30 ha
Permanent grassland	40 ha
Extensive grassland	10 ha
Organic farming	50 ha
Maize with reduced row spacing	470 ha
Maize with limited fertiliser (100 kg N/ha)	180 ha
Cereals with intercropping	100 ha
Integrated fertiliser and manure application	757 ha (25% of all arable land)
Temporal restrictions for fertiliser/ manure/ slurry spreading	(applied throughout)
Towed umbilical hose, trailing shoe or slit injection	200 ha

Table 3-15
Combination of measures under the realistic scenario

Taking the above into consideration, three scenarios are set up with different combinations of measures: A realistic scenario, an optimistic scenario, and a technocratic scenario. The scenarios are described in greater detail below.

The realistic scenario

The realistic scenario describes a situation that could realistically be expected in the medium term, i.e. over a three- to five-year-period, if the current level of ambition is maintained and some additional measures are taken. It basically involves all measures that are available, and assesses on how many additional ha this measure could be applied. The additional available potential for each measure was estimated based on expert knowledge, considering which measures are already implemented in how many ha in the area, and what progress has been made with these measures in recent years. The realistic scenario could therefore be described as a business-as-usual scenario. In detail, the realistic scenario comprises the measures shown in the 3-15 table.

This combination of measures achieves a nitrate load reduction of 203,856 kg, which corresponds to 31.1% of the estimated gap – that is, only a third of the gap is closed. The total cost of the scenario adds up to 143,977 Euro, which means that the average cost per percentage point of the gap closed is 4,631Euro. The average cost per kg of N avoided is 3.13 Euro. The cost-effectiveness curve for this scenario is presented in the following graph. (figure 3-8).

The realistic scenario could be expected in the medium term, i.e. over a three- to five-year-period

Figure 3-8
Reduction of N emissions and costs per kg N saved under the realistic scenario

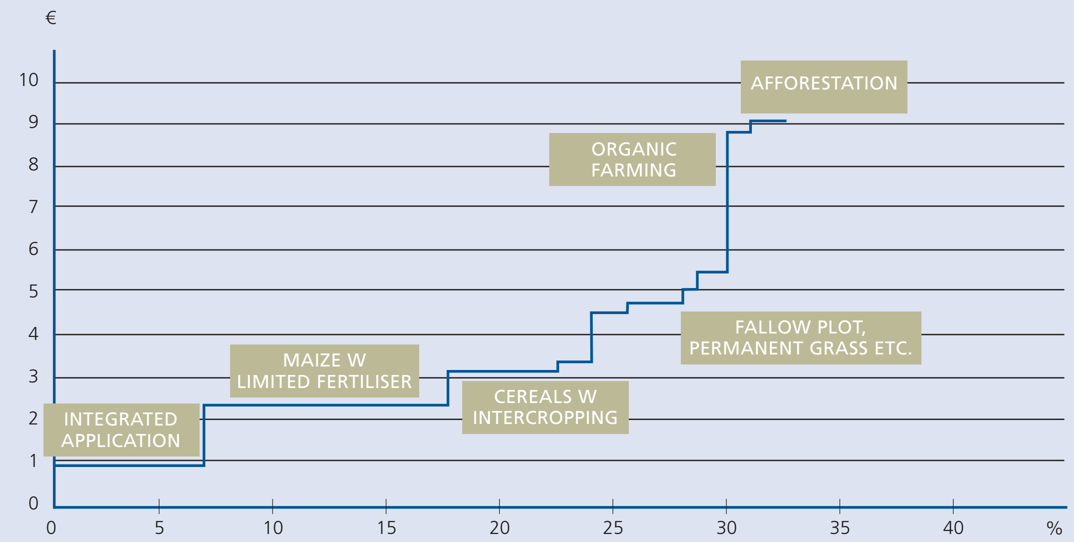


Table 3-16
Combination of measures under the optimistic scenario

Measure ...	**... carried out on an additional area of**
Afforestation	60 ha
Active fallow plot greening	90 ha
Permanent grassland	120 ha
Extensive grassland	30 ha
Organic farming	150 ha
Maize with reduced row spacing	364 ha
Maize with limited fertiliser (100 kg N/ha)	400 ha
Cereals with intercropping	250 ha
Integrated fertiliser and manure application	757 ha (25% of all arable land)
Temporal restrictions for fertiliser/ manure/ slurry spreading	(applied throughout)
Towed umbilical hose, trailing shoe or slit injection	200 ha

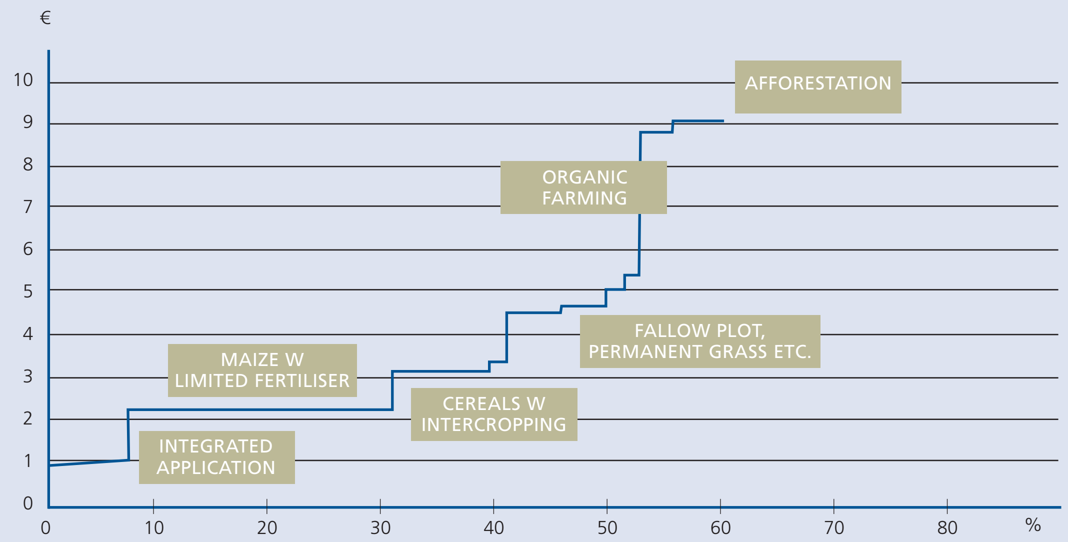


Figure 3-9
Reduction of N emissions and costs per kg N saved under the optimistic scenario

The optimistic scenario

The optimistic scenario describes a higher level of ambition than the realistic scenario, which could still be realistically achieved in the medium term (over a three- to five-year-period), but which would require significant additional efforts and resources compared to the status quo. Like the realistic scenario, it involves all available measures, and assesses on how many additional ha these measures could be applied if very optimistic assumptions are made about the uptake of measures. Again, the additional available potential for each measure was estimated based on expert knowledge. The optimistic scenario could therefore be described as a reference scenario of what could be achieved if the necessary commitment is given.
In detail, the optimistic scenario comprises the following measures:

Under the optimistic scenario, the area converted to grassland, forests, organic farming and greened fallow plots is three times that of the realistic scenario. For maize, under the optimistic scenario, measures are applied on all of the 764 ha currently used for maize (see tabel 3-16).

The optimistic combination of measures achieves a nitrate load reduction of 391,891 kg, which corresponds to 59.8% of the estimated gap – that is, almost two thirds of the gap are closed. The total cost of the scenario adds up to 312,137 Euro, which means that the average cost per percentage point of the gap closed is 5,222 Euro. The average cost per kg of N avoided is 3.53 Euro. Compared to the realistic scenario, the optimistic scenario is more than twice as expensive in terms of the total cost, and about 13% more expensive in terms of the costs per unit. The cost-effectiveness curve for this scenario is presented in figure 3-9.

Table 3-17
Combination of measures under the technocratic scenario

Measure ...	... carried out on an additional area of
Active fallow plot greening	841 ha
Maize with limited fertiliser (100 kg N/ha)	764 ha (all maize)
Cereals with intercropping	250 ha
Integrated fertiliser and manure application	757 ha (25% of all arable land)
Temporal restrictions for fertiliser/ manure/ slurry spreading	(applied throughout)
Towed umbilical hose, trailing shoe or slit injection	200 ha

The third scenario: what would be the result if the selection of measures was done purely based on cost-effectiveness

The technocratic scenario

Finally, the third scenario basically describes the central planner's optimum: what would be the result if the selection of measures was done purely based on cost-effectiveness, ignoring all real-life constraints and limitations. In contrast to the previous two scenarios, full achievement of the reduction target was treated as a binding constraint in this scenario. Starting with the cheapest measure available and then moving to the next-best measures, each measure was applied to the maximum area where it could potentially be applied. This exercise was repeated until 100% target achievement would be reached.

In this way, the more expensive measures – such as afforestation or organic farming – do not enter into the scenario. The most cost-effective measure – limiting fertiliser applications on maize to 100 kg – is applied for the entire area where maize is grown today, i.e. 764 ha. The next, more expensive measure, intercropping, is applied to 250 ha of the cereals, which is considered as the maximum area where this measure is applicable. For the remainder, arable land is converted to greened fallow plots on 841 ha – which is almost 9 times more than what was considered feasible in the optimistic scenario. The end result is the technocratic optimum: a result that cannot be achieved in practice, but which can serve as a yardstick for real-life solutions.

In detail, the technocratic scenario comprises the following measures:

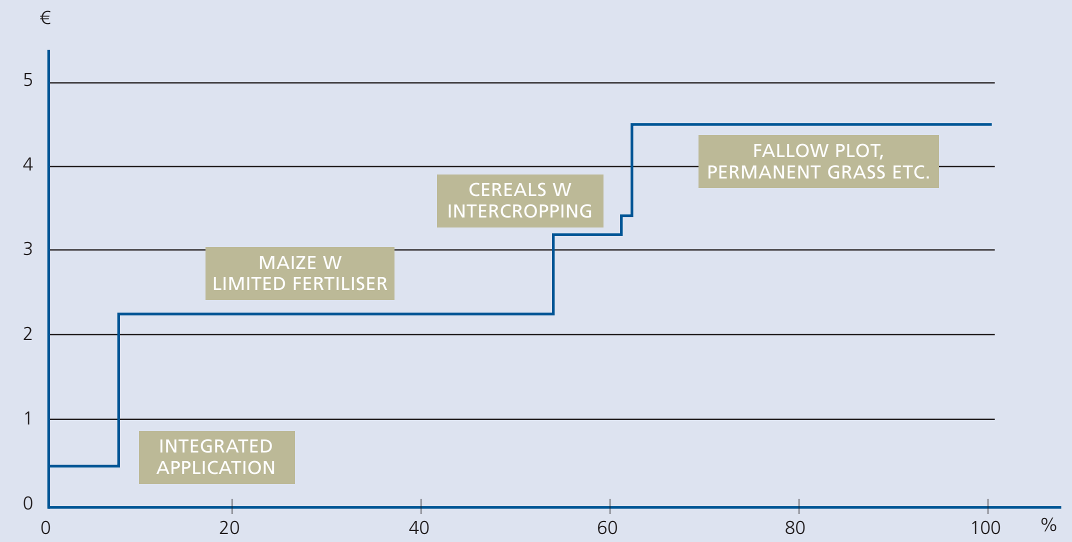


Figure 3-10
Reduction of N emissions and costs per kg N saved under the technocratic scenario

The technocratic combination of measures achieves a nitrate load reduction of 655,873 kg, which closes the estimated gap entirely – which was, of course, the binding constraint. The total cost of the scenario adds up to 463,097 Euro, which means that the average cost per percentage point of the gap closed is 4,630 Euro. The average cost per kg of N avoided is 3.13 Euro. Compared to the realistic scenario, the technocratic scenario is more than three times as expensive in terms of the total cost, but about 3% cheaper in terms of the costs per unit. The cost-effectiveness curve for this scenario is presented in figure 3-10.

Comparison of the three scenarios

The figure 3-11 gives an overview of the total costs and the effectiveness of the three combinations of measures. The average cost-effectiveness of each scenario is given by the gradient of the line from the origin to each of the red triangles: the flatter this line, the more cost-effective the scenario on average.

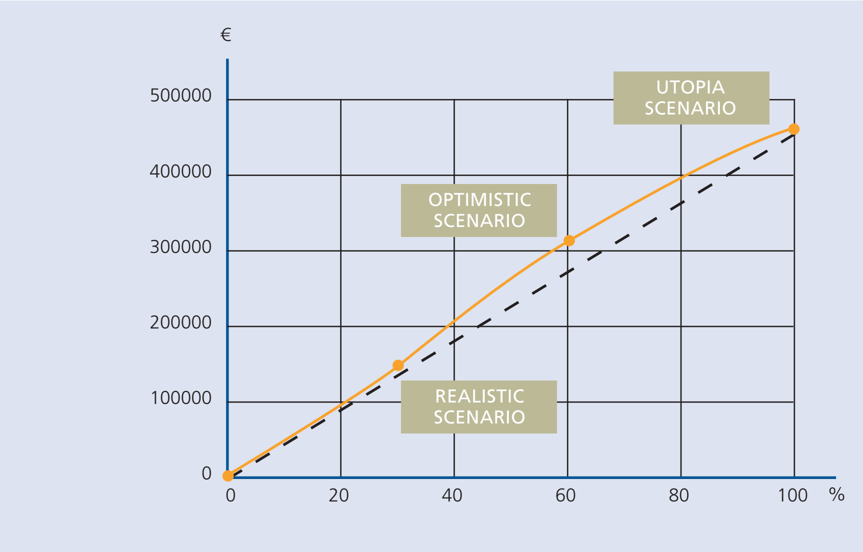


Figure 3-11
Total costs and the effectiveness of the three combinations of measures

Two conclusions can be drawn from the comparison of the different scenarios:

- First, despite the differences in the assumptions made, the cost-effectiveness ratios of all three scenarios are reasonably similar: in the graph above, the red triangles indicating the three scenarios are almost on the same line.
- Secondly, a caveat applies to the interpretation of the graph above. Contrary to what the dashed line suggests, it will not be possible to move from the "optimistic" to the "technocratic" scenario, since the latter will not be a feasible scenario under real-life conditions. In reality, the costs of further N reductions are more likely to increase substantially as the level of ambition increases beyond the "optimistic" scenario. This would suggest that the costs of reaching the 100% objective are more likely to be somewhere between 600,000 and 800,000 Euro.

3.2.6 Extended Cost-effectiveness analysis: Side benefits

Taking the point of departure in the approach to side benefits elaborated in the methodology section, the following seven dimensions of such potential side benefits were investigated in the case study:

- Water quantity
- Water quality in terms of pesticides
- Surface water quality
- Biodiversity
- Landscape
  (including amenity value and tourism)
- Climate change
- Soil quality

In the following, the ten measures to reduce nutrient load to groundwater were assessed in terms of their potential side benefits along the seven dimensions described above. The assessment was first done as a qualitative quick check, based on the expert opinions of the analysts. In this assessment, impacts were rated on a seven-step scale, from negative high, negative medium, negative low, neutral, positive low and positive medium to positive high.

	Afforestation	Active fallow plot greening	Permanent grassland	Extensive grassland	Organic farming	Maize with reduced row spacing	Maize with limited fertiliser (100 kg N/ha)	Cereals with intercropping	Integrated fertiliser and manure application	Temporal restrictions for fertiliser spreading	Towed umbilical hose, trailing shoe, slit injection
Water quantity	÷	x	÷	÷	x			÷			
Water quality (pesticides)	✓	✓	x	x	✓						
surface water	+	✓	+	+	+	x	+	+	+	+	+
Biodiversity	+	+	x	x	+			x			
Landscape & tourism	✓		+	+	x			x			+
Climate change	+				+						
Soil quality					+					+	

÷ = Negative low
= None
x = Positive low
+ = Positive medium
✓ = Positive high

Table 3-18
Potential co-benefits of the ten measures

It was found that all of the considered measures had beneficial side impacts on surface water, although mainly to a medium degree. A few measures, including afforestation, permanent and extensive grassland and cereals with inter-cropping had negative impacts on water quantity; otherwise, no negative side-effects were anticipated.

In general, both the extent and the type of side benefits differed for the different measures. One pattern that emerges quite clearly is that by and large, side benefits matter most for those measures that involve a land use change – such as afforestation, fallow plot greening, organic farming and permanent or extensive grassland. For the measures that relate to agricultural management practices, side benefits are mainly related to surface water. Other than that, there are few and isolated side benefits, e.g. through technical improvements in the application of manure fertiliser (towed umbilical hose etc.), which also reduce odour-related effects, and thereby benefit touristic uses in the area.

As for the narrow CEA, with the considerations on side benefits it has to be kept in mind, though, that measures may be located at the same area and thus they may partly exclude each other. For example maize with limited fertiliser excludes the option of cereals with inter-cropping at the same area. However, there may also be synergies or added values due to a combination of measures at a single plot, such as integrated fertiliser and manure application combined with temporal restrictions on manure or fertiliser spreading or cereals with inter-cropping.

The crucial question is whether the inclusion of side-benefits will lead to a changed ranking of the measures

The role of side-benefits in a Cost-Effectiveness Analysis

In the cost-effectiveness analysis carried out for the Thülsfelde area, the identified side benefits were only described in qualitative terms, describing impacts as by their general direction (positive or negative) and the strength of effect (low, medium and high). However, they were not quantified in the first instance.

Since the cost-effectiveness analysis, as it was applied in the Thülsfelde case, is a tool that is used to provide a ranking of different alternatives, it is sufficient if the analysis can indicate whether one option is preferable to another; it is not necessary to also specify how much better one option is in comparison to another. Following this line of reasoning, the crucial question is whether the inclusion of side-benefits will lead to a changed ranking of the measures. In other words: are the monetised benefits large enough to offset the difference in costs that was observed between the different measures.

This implies that, for the decision at hand, it is not necessary to know the absolute amount (monetary value) of the benefits – instead, it is sufficient to establish whether the benefits are likely to be sufficiently large to affect the ranking (i.e. whether they could outweigh the cost difference to the next-best measure).

As an example, it was found that active fallow plot greening reduces N loads at a cost of 4.55 € / kg N. By comparison, planting cereals with intercropping could reduce N loads at a cost of 3.21 € / kg N. Thus, the former measure is about 40% more expensive than the latter. At the same time, active fallow plot greening has several side-benefits: it has a positive impact on water quantity (whereas intercropping increases water consumption), it has highly positive impacts in terms of reduced pesticide use and impacts on surface water quality (whereas intercropping has only a medium positive impact on surface water), and it has a medium positive effect on biodiversity (whereas intercropping only has a small positive effect). By contrast, the only side-benefit where intercropping scores higher than active fallow plot greening is the effect on landscape, aesthetic value and thereby on tourism – with a low positive effect for intercropping, and no effect for active fallow plot greening.

The key question then becomes how the two types of information – the qualitative information on the side-benefits and the monetary information on cost-effectiveness – can be combined in any meaningful way. For the example above, one option would be to attach weights to the different impact categories (from highly negative = -3 to highly positive = +3), and to sum up these weights. This would produce the following result:

	Afforestation	Active fallow plot greening	Permanent grassland	Extensive grassland	Organic farming	Maize with reduced row spacing	Maize with limited fertiliser (100 kg N/ha)	Cereals with intercropping	Integrated fertiliser and manure application	Temporal restrictions for fertiliser spreading	Towed umbilical hose, trailing shoe, slit injection
Water quantity	÷1	1	÷1	÷1	1	0	0	÷1	0	0	0
Water quality (pesticides)	3	3	1	1	3	0	0	0	0	0	0
Surface water	2	3	2	2	1	2	2	2	2	2	2
Biodiversity	2	2	1	1	2	0	0	1	0	0	0
Landscape & tourism	3	0	2	2	1	0	0	1	0	0	2
Climate change	2	0	0	0	2	0	0	0	0	0	0
Soil quality	0	0	0	0	2	0	0	0	0	2	0
SUM	**11**	**9**	**5**	**5**	**12**	**1**	**2**	**3**	**2**	**4**	**5**

Table 3-19
Aggregated scores of potential side benefits of measures

While this is a quick and simple way of combining the information, there are two strong assumptions involved:

- One assumption concerns the choice of weights (i.e. positive low equal to +1 or negative high equal to -3). These have been determined arbitrarily; it would also be possible to set positive high equal to +5 (rather than +3). The absolute value of the weights is not as critical in this context. However, the relative value of the weights in comparison to each other – whether "positive high" has 2, 3 or 5 times the weight of "positive low" – will certainly influence on the result.
- The second assumption concerns the summation of the different categories. In the simple example above, all categories were weighed equally. However, in a more realistic setting, it is likely that some types of impacts will be considered more important than others – e.g., in a touristic area, impacts on landscape and tourism will receive greater attention, whereas in an area with abundant water supply, impacts

on water quantity will be less important. In the summation, this can be achieved by attaching correction factors to the different categories, which reflect the societal relevance of the issue.

Benefit-based approaches attempt to derive an estimate of the actual benefits created by an intervention.

Thus, both assumptions can be contested. One possibility to arrive at a more justifiable aggregation procedure would be through public participation, e.g. asking representative stakeholders to rank the different impact categories according to their perceived salience, or to assign weights to the different impact categories.

Returning to the previous example, and using the simple aggregation scheme above, we would now find that the active fallow plot greening receives 9 "benefit units", and therefore three times as much as intercropping for cereals. However, the question remains whether this difference in side-benefits is sufficient to compensate for the 40%-difference in cost-effectiveness.

One elegant but time and data-intensive way of dealing with this problem is to measure the side-benefits in money terms. The obvious drawback of this approach is that the "goods" and "services" in question – such as clean air, less pesticides in groundwater, a beautiful rural landscape – are atypical, since they are not traded on a market. As a consequence, there is no market price attached to them, which could be used to measure their value. Nonetheless, they clearly contribute to human wellbeing and quality of life in a number of ways. It is therefore plausible that they should have some value, in the sense that users would be prepared to sacrifice some other goods in exchange for better environmental quality.

To quantify this value in monetary terms, a number of economic valuation methods have been developed and used over the last decades. These methods can be broadly subdivided as follows:

- Cost-based and benefit-based approaches. The former look at the cost of repairing, avoiding or compensating for damage, and use this information as a proxy for the benefit achieved – thus assuming that the benefits are at least as high as the costs. This is a strong assumption, and while cost-based approaches have the advantage of being relatively quick and inexpensive, their information value is limited. By contrast, the benefit-based approaches attempt to derive an estimate of the actual benefits created by an intervention.

- Within the benefit-based approaches, market-based and non-market-based methods can be distinguished. Market-based methods measure benefits through actual market transactions, e.g. looking at the value of water as a production factor in agriculture and industry, or through the market price of fish caught from a river. Unfortunately, it is in the nature of environmental improvements that many of its effects are not reflected in market transactions. To elicit these effects, non-market-based methods are needed.

- Within the non-market-based methods, there are revealed-preference and stated-preference methods. Revealed-preference methods (such as hedonic pricing or the travel cost approach) infer the influence of environmental factors on observed market transactions – e.g. studying how water quality improvements affect house prices nearby a river. By contrast, stated-preference approaches (such as contingent valuation or choice experiments) elicit how much individuals would be willing to pay for an improvement in water quality.[24]

3.2.7 Side benefits of organic farming and afforestation – Examples

Based on the qualitative appraisal, afforestation and organic farming emerged as the measures with the largest array of side benefits. Side benefits take the form of reduced pesticide levels in water, as forests and organic farmland receive little or no pesticide applications. Other side benefits of both measures relate to improvement of surface water and biodiversity as well as to mitigation of climate change. Surface water benefits from reduced pesticide and nutrient loading, in particular of phosphorus.

Monetary estimates of the side-benefits of organic farming

Organic Farming is generally environmentally beneficial compared to conventional farming practices. A major UK assessment of organic farming (Shepherd et al 2003) concludes that organic farming

- on average creates positive benefits to wildlife conservation;

- generally improves soil quality, due to increased organic matter content, soil structure enhancement and increased earthworm numbers and microbial activity, but also due to less soluble nutrients, pesticide and veterinary medicine residues;

- commonly involves decreased nitrogen leaching;

- rarely leads to pesticide pollution, in particular in the case of herbicides;

- tends to have lower ammonia emissions per area;

- has lower carbon emissions from fossil fuel per area;

- has generally a greater per area energy efficiency;

- commonly operate with lower nutrient surpluses; and

- tend to produce less controlled waste, due to lower input use.

---

[24] These different valuation methods and their application in the context of the EC Water Framework Directive is the subject of the ongoing European research project AquaMoney. See www.aquamoney.org for more information.

Organic farming is useful if broad environmental concerns are to be addressed

These considerations are in line with Stolze et al (2000). The latter conclude that organic farming is useful if broad environmental concerns are to be addressed, since it leads to improvements in most environmental indicators. In particular, organic farming can significantly contribute to biodiversity, since organic farms are less intensive and often introduce a more mixed land use (Soil Association 2000). Organic farming practices are both beneficial to flora and fauna in terms of habitat and food abundance (Latacz-Lohmann and Renwick 2002). It has been estimated that 57 per cent more species can be found on organic farms (Soil Association 2000). Several rare and declining wild arable species are found only on organic land and total floral species numbers can be up to six times higher (Reiter 2004). In addition the number of endangered herbs is significantly higher (Reiter 2004). A major reason for this is the use of herbicides and higher fertiliser application rates on conventional land, but also the crop rotations of organic farms have a positive impact on biodiversity (Reiter 2004). Extensive management of grassland in terms of fewer cuts, lower stocking densities and fertiliser application contributes to a more diverse flora. In particular declining species are being supported by organically managed grassland (Reiter 2004).

It is estimated that there are 25 per cent more birds at the field edge, 44 per cent more in-field in autumn and winter and 2.2 times as many breeding skylarks and higher skylark breeding rates at organic fields (Soil Association 2000). The latter review also finds indications for greater abundance and diversity of arthropods that comprise bird food, non-pest butterflies and spiders. Further, organic farming improves breeding and food conditions for avifauna. This is due to greater insect numbers, greater shares of fallow land (Oppermann 2004) and greater diversity of crops, including spring crops, which are beneficial for the declining species of skylarks (Reiter 2004). Long-term experiments have shown that organically managed soils have greater microbial activity and increased microbial biomass (Reiter 2004). They also tend to have greater humus content and greater biodiversity and biological activity. Hence, medium side benefits accrue due to improved soil quality (Dabbert and Häring 2003).

Effects on climate change derive from lower emissions of climate impacting gases like N2O and methane, but also due to capture of CO2 in wood and soil. Organic farming is suggested to emit 40 to 60 per cent less CO2 per area compared to conventional agriculture (Dabbert and Häring 2003). In addition, organic farming requires lower energy inputs and thus carbon emissions per area than conventional agriculture.

The findings of the economic evaluation of the DEFRA Organic Farming Scheme (Defra 2003) reflect those above cited benefits of organic farming, however it finds that these benefits are difficult to express in monetary terms.

In economic terms, side benefits of organic farming are reflected to some extent in the premium product prices paid by consumers of organic food. Commonly organic products sell at prices at least 30 per cent higher than average conventional farm products.

External costs of…	**…organic farming**		**…conventional farming**	
	per farm	per ha	per farm	per ha
Intensive farming	771	44	1212	69
Medium intensity	719	41	1087	62
Extensive farming	355	20	389	22

Table 3-20
External costs of organic and conventional dairy farms in Austria

Benefits of organic farming also include the avoided external costs that are associated with conventional farming

However, benefits of organic farming also include the avoided external costs that are associated with conventional farming. In this context an Austrian study compares the external costs of organic dairy farms with conventional farms (Krachtovil et al 2003). In terms of greenhouse gas emissions (CO2, N20 and methane) the organic farms caused externalities per farm of 771 Euro per farm or 44 Euro/ha (intensive organic farming), 719 Euro per farm or 41 Euro/ha for medium intensity of organic farming, and 355 Euro per farm or 20 Euro/ha for extensive organic farming. By comparison, externalities at conventional farms amounted to 1212 Euro per farm or 69 Euro/ha for intensive farming, 1087 Euro per farm or 62 Euro/ha for medium intensity farming, and 389 Euro per farm or 22 Euro/ha for extensive farming. The overall range of the difference between the external cost of conventional and organic farming would then be 2 to 25 Euro/ha (Table 3-20). Overall, the study concludes that a switch from conventional to organic farming would reduce such external costs by 27 per cent across farm types.

Another study focuses on social costs of organic farming in comparison to conventional farming associated with green house gases, nitrate leaching and pesticide residues (O'Riordan and Cobb, 2001). Such costs were estimated to range from 15 to 22 Euro per hectare for organic farms and between 37 to 59 Euro per hectare for conventional systems (O'Riordan and Cobb, 2001). Thus the difference of external costs between conventional and organic farming (i.e. the benefit of organic farming) ranges from 15 to 44 Euro per hectare.

Pretty et al (2000) provide figures on monetary costs of UK agriculture. A number of the cost components would be reduced by organic farming and thus can be considered as relevant indications of side-benefits of organic farming:

- annual externalities of pesticide use are estimated at 49 Euro per hectare of land receiving pesticides (80% of costs arise from arable land);
- costs of organic matter and carbon dioxide losses from soils are estimated on average at 7.70 to 18.40 Euro per ha and annum (arable and grassland accrue each for 50% of costs);
- costs of emissions of harmful gases including methane, ammonia, nitrous oxide and carbon dioxide range on average around 146 Euro per hectare. Of this, nitrous oxide is making up the major share with 97 Euro per hectare. Arable and grassland accrue each for 50% of costs, but 80% of nitrous oxide arise from arable land; and
- Costs of biodiversity and wildlife losses are estimated on average at 1.30 to 4.60 Euro per hectare and year for the UK (arable and grassland accrue each for 50% of costs).

In addition Pretty et al (2000) cite studies by the Fleischer and Waibel, from which they conclude that the external costs of pesticides and gaseous emissions from arable and grassland in Germany range from 59 to 85 Euro per hectare (Fleischer and Waibel 1998 and Waibel and Fleischer 1998 cited in Pretty et al 2000).

Referring to benefits of improved surface water quality, phosphorus emission is a key factor next to other factors like nitrate. An indication of the benefits of reduced phosphorus emissions is given through the willingness of a population to pay for cleaner water due to lower phosphorus inflows. Bateman et al. (2006) for example found that households in the UK region of East Anglia were willing to pay 110 Euro on average for a policy that would reduce P emissions and thus prevent eutrophication and increase surface water quality. For the entire region the annual benefit amounts to more than 250 million Euro. Comparable studies focusing on reduced nutrient loads to surface water found willingness to pay of 47-79 Euro per adult in the Stockholm region (Söderqvist and Scharin 2000), 350-630 Euro per person for all of Sweden, and 46-90 per person in Poland (Turner et al 1999). Unfortunately, these numbers are not directly transferable to the question at hand, as they do not calculate benefit per hectare, neither do they specify the share of benefits attributable to changes in agricultural practice. Thus, while these figures indicate that there are substantial benefits from reduced eutrophication, and while reduced P emissions are among the side-benefits of organic farming, it is not possible to link the two based on these studies.

In an alternative approach Rüffer (2006) identified ranking orders by the public in terms of support of certain environmental attributes related to farmland. Taking "normal" arable land as a baseline, respondents from the Nordheim region preferred woodland features most. Herbal rich grassland ranked second, followed by herbal structures and finally herbal rich arable land. The wooden features could be framed as side-benefits of afforestation, while the latter three are commonly features of organic farming as opposed to conventional farming. All these benefits can be considered as amenity and landscape benefits of extensive land uses over the baseline of conventional agriculture.

Table 3-21
Monetary estimates
of external costs of
conventional farming

Type of external cost	External costs (Euro/ha*year)	Reference
Gaseous emissions in total	146	Pretty et al 2000*
Soil carbon	8-18	Pretty et al 2000
Nitrous oxide emissions (gaseous)	97	Pretty et al 2000
Pesticides	49	Pretty et al 2000
Pesticides and gaseous emissions	59-85	Pretty et al 2000
Biodiversity	1-5	Pretty et al 2000

* All figures from Pretty et al (2000) only indicate the external cost of conventional agriculture.
These figures will be lower for organic farming, which provides a measure of the benefits of organic farming.

The table 3-21 provides an overview of the different benefits estimated in the various studies cited.

Based on the available evidence, it is not possible to derive an average monetary value of the total benefit of switching from conventional to organic farming. For the UK, Pretty et al. Put forward an average value of 218 Euro per ha per year for the external costs of agriculture. How much of this could be avoided by switching to organic farming is not known. For gaseous emissions in Austria, Krachtovil et al. Found that the external costs of organic farming are about 27% lower than for conventional farming. For pesticides, the difference between conventional and organic farming could be somewhat larger still.

Yet, there are only a limited number of valuable attributes of organic farming listed here in monetary terms. There may be also benefits related to improved surface water and benefits due to improved landscape and amenity values. Suitable clear-cut monetary values of such effects of organic farming have not been fund in the literature.

For the question at hand, whether the inclusion of side-benefits would change the ranking of measures, the situation is as follows: for organic farming, an annual cost of 300 Euro was estimated, and a cost-effectiveness of 8.83 Euro per kg N reduced. To make organic farming as cost-effective as the next-best measure (extensive grassland cultivation, with a cost-effectiveness of 5.30 Euro per kg N reduction), the net costs of organic farming (costs minus benefits) would lie at 180 Euro per ha. In other words, the monetary benefits would need to be at least 120 Euro. Given the benefit ranges presented above, it is not unlikely, but far from certain that the side-benefits of organic farming could indeed reach this magnitude. An inclusion of amenity and landscape values as well as reduced phosphorus emissions improving surface water may, however, lead to an additional increase of the overall side-benefits of organic farming. Further also transaction costs may be included in the calculations. These have been suggested lower for organic farming measures (Beckmann et al. 2003).

Monetary estimates of the side-benefits of afforestation

Afforestation may bring substantial side benefits

Afforestation may bring substantial side benefits, because woodland has often high landscape and aesthetic value, which is also beneficial to tourism. Direct side-benefits of forests include timber and deer, but also water and air purification, while indirect benefits include noise mitigation and improved location for housing as well as general amenity and landscape value. Forests also function as carbon sinks. The economic value of forests as carbon sinks in Germany has been estimated to range between 48.8 to 449 million Euro (Thoroe et al 2005). Since the forested area of Germany is about 11,075,800 hectare (Bundeswaldinventur 2007), this corresponds to 4.40 to 40.50 Euro per forested hectare. For a drinking water protection project in Northern Jutland, annual carbon offsets due to afforestation have been valued at 105 Euro per hectare (Loubier 2003).

Recent studies also try to estimate the monetary benefits of forests with improved biodiversity. The relating mean willingness to pay of residents range between 6.60 and 13.28 Euro per household for forests in the Lüneburger Heide and between 6.23 and 6.64 Euro for forests in the Sollingen area depending on elicitation procedure (Meyerhoff et al 2006).

Focussing strictly on afforestation of agricultural land, a study from Belgium estimates yearly per hectare benefits at 28.50 Euro from timber, 15 Euro from hunting, 25 Euro from carbon fixation. These benefits are dwarfed by the recreational benefits, which amount to 1,440 Euro per ha on average, and the non-use benefits of 3,680 Euro (Moons 2002). The latter includes also general amenity and housing location. A Spanish study estimated the total willingness to pay for an afforestation project in Catalonia at 1,613 Euro per hectare (Riera and Mavsar 2005). Another study by Bateman et al (1996) estimated the willingness to pay for 40.5 hectares of woodland established on farmland in the UK. In the vicinity local to the woodland, households were willing to pay 14.68 Euro per year for a tax (total 65,675 Euro per year), while visitors were willing to pay 18.16 per adult for visits based on fees (total 208,700 Euro per year). Hence the recreational value of the woodland equals 1,622 Euro (tax) or 5,153 Euro (fee) per hectare.

Several studies have looked at the recreational value of forests in more detail. While such values are site-specific, they are generally positive and often substantial. Thus, the willingness to pay of visitors of forests in Hamburg and the Palatine have been estimated to range around 51 Euro per year or 4.10 Euro per visit (Elsasser 1996). Hence, the aggregated recreational value of forest for Hamburg ranged between 51 to 77 million Euro. In the Palatine forest visitors were willing to pay 15.30 Euro for their entire stays. A comparable study estimated that the value of the total forest areas in Switzerland in terms of travels and stays in forested areas ranges around 6.3 billion Euro per year and at 648 Euro per person and year (Ott and Baur 2005). Similarly it has been estimated that the recreational value of the forests in Germany ranges at around 3 billion Euro (Neidlein and Walser 2005). Considering the total forested area of Germany (Bundeswaldinventur 2007), this would correspond to an average value of 271 Euro per hectare. In a similar fashion Dubgaard (1998) identifies the willingness to pay of Danish individuals for recreation in Danish forests. The mean willingness to pay is 17.2 Euro per individual and 178 Euro per hectare of forest.

Type of benefit	Value of benefit (Euro/ha*year)	Reference
Carbon offset	41	Thoroe et al 2005 (adapted figure)
Carbon offset	105	Loubier 2003
Carbon offset	25	Moons 2002
Timber	29	Moons 2002
Hunting	14	Moons 2002
Recreation	1,440	Moons 2002
Recreation	271	Neidling and Walser 2005 (adapted figure)
Recreation	1,622 - 5,153	Bateman et al (1996)
Recreation	178	Dubgaard 1998
Non-use value	3,680	Moons 2002

Table 3-22
Monetary estimates
of potential benefits
of afforestation

Forested catchments may render drinking water treatment unnecessary. Consequently a Swiss study estimated that costs of 0.14 Euro per m3 treated drinking water can be saved through forest cover (Küchli and Meylan 2002). In a German case the value of forested land for water protection has been calculated at 35 Euro per hectare for Baden-Württemberg (Neidling and Walser 2005).

Table 3-22 presents an overview of the different benefit estimates for afforestation:

As becomes evident from this overview, there is a considerable variability between the different benefit estimates. Recreation and non-use benefits tend to dwarf the benefits from extractive uses and ecosystem services. Those studies that have ventured to assess the total benefits of afforestation across a range of benefit categories – Riera and Mavsar 2005 for Catalonia and Moons 2002 for Belgium – have found quite substantial benefit figures of 1,613 and 5,188 Euro per hectare per year, respectively.

Returning to the original question whether inclusion of the side-benefits of afforestation could change the relative attractiveness of measures and their cost-effectiveness ranking, the situation is as follows: the costs of afforestation were calculated at 950 Euro per ha. The corresponding cost-effectiveness was 11,88 Euro per kg N reduced. In order to be as cost-effective as the next-best measure – organic farming – the net costs of afforestation would need to lie at 706 Euro. In other words, the side-benefits would need to account for at least 244 Euro. Given the range of benefit estimates presented above, this would seem plausible. However, this calculation holds only if the side-benefits of organic farming are not included – otherwise, the side-benefits of afforestation would need to be larger still.

Measures targeting agricultural practices are much more cost effective than measures which rely on land-use change

In comparison to the next-best measure, extensive grassland cultivation, the side-benefits of afforestation would need to amount to at least 526 Euro to draw even in terms of cost-effectiveness. Given the benefit estimates presented above, this is likely, but not certain. The recreation benefits estimated by Dubgaard 1998 for Denmark, by Neidling and Walter 2005 for Germany, by Moons 2002 for Belgium and by Bateman et al 1996 for the UK ranged from 178 to 5,153 Euro per hectare. While the Belgian and UK estimates (of 1,440 - 5,153 Euro) would easily recommend afforestation as a highly efficient measure, the Danish figure of 178 Euro or the German estimate of 271 Euro would not change the choice. However, it should be noted that the 271 Euro-figure is a national average – it is quite plausible that recreation values would be much higher in areas that are frequented by tourists, as is the case in the Thülsfelde study area.

Finally, it is striking that some of the monetary benefits estimates (the recreation benefits calculated by Bateman et al. 1996 and the aggregated figures by Moons 2002 and by Riera and Mavsar 2005) in fact exceed the costs of the measure. This is an interesting, but not implausible finding: it means that carrying out the measure creates a net benefit to society, even without considering the effect on nitrate loads. Recreational and other benefits alone therefore justify carrying out the measure, irrespective of its effect on groundwater protection. The measure should therefore be carried out in any case.

3.2.8 Conclusions

Overall, the Thülsfelde case study provides a solid and illustrative case of how cost-effectiveness analysis can be applied to groundwater protection measures. The study benefits from fairly solid data, and above all from considerable experience with measures that were applied in the past. Based on these, there was rich expert knowledge on what measures work and which do not work.

From the pure cost-effectiveness considerations presented above, it can be concluded that measures targeting agricultural practices are much more cost effective than measures which rely on land-use change. This would suggests that many of the measures that are currently applied in Thülsfelde are in fact inefficient, and that more could be achieved by reallocating resources e.g. from afforestation to improved agricultural management.

Interestingly, the relative performance of the different measures would be judged somewhat differently if the side-benefits of different measures are included in the consideration. These side-benefits are particularly relevant for the more expensive measures involving land use change, such as afforestation, organic farming or conversion to grassland. If such side-benefits are included, the costs of the measures may be compensated in parts and thus the ranking of the measures in terms of cost-effectiveness may also change. This could be particularly relevant for afforestation, where some studies have found evidence for large side benefits, especially in terms of recreation and amenity. These benefits might even outweigh the costs of the measure. For organic farming the picture is less clear, as existing studies tend to describe the external costs of agriculture more generally, but pay less attention to the differences between conventional and organic farming.

Side-benefits are particularly relevant for the more expensive measures involving land use change, such as afforestation. These benefits might even outweigh the costs of the measure

One important implication of this concerns the distribution of costs, effects and side-benefits. By definition, side-benefits do not accrue to the parties that are implementing and financing the measures – in the case of recreation and amenity, it is the local public and the regional tourists who benefit. Thus, if the measures are carried out and financed by the water supplier, they provide an unintended and uncompensated benefit for the local population (that benefits in terms of increased quality of life), the tourists visiting the area (who benefit from increased amenity) and, indirectly, the local tourist industry such as restaurants, hotels and campsites (that benefit from increased visitor numbers).

Yet, also using cost-effective measures, the ambitious groundwater protection targets identified in the study will not be achieved easily. A realistic scenario, which more or less extrapolates current practices into the future, would only result in a one-third reduction of nitrate inflow. Even under the more ambitious optimistic scenario, only 60 per cent of the target is achieved. In consequence, full target achievement might well end up at a yearly cost of 600,000 to 800,000 Euro, in addition to the costs of measures that are currently implemented.

The UK handbook used for the case study was generally helpful as a source of inspiration and to structure the analysis. However, the task that proved most challenging and required most knowledge was the calculation of the actual costs and the effects of measures, as well as the estimation of side benefits. For both of these, the handbook and spreadsheet tool provide very little support – the spreadsheet tool in fact only requires the user to enter the cost and effectiveness data, but offers no support in getting to this data.

Also, the spreadsheet tool does not seem to be fully mature and road-tested yet. It is often unclear what information is required, and in which format the information should be entered. In several instances, the spreadsheet will not accept ranges for costs and effectiveness. Thus, a more flexible format would be advisable, since the spreadsheet will rarely be used for actual calculations, but mostly for data entry and organisation of data.

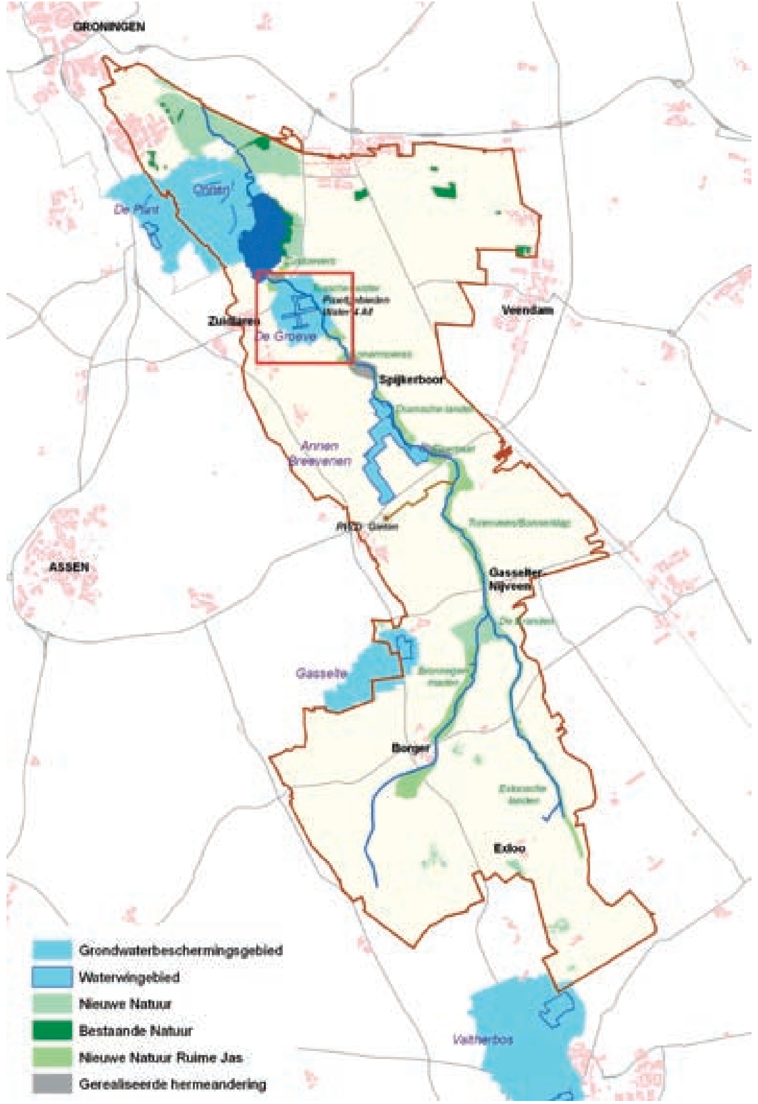


Figure 3-12
Location of the
Tusschenwater Project
in the Hunze Valley

3.3 The Netherlands, Drenthe

3.3.1 Background

Tusschenwater is the northernmost and deepest part of the Hunze Valley (South of the Zuidlaarder Lake that is located between the small villages of De Groeve and Zuidlaren – see figure 3-12) and one of the locations where the Drinking Water Company of the Province of Groningen (WBGr) extracts groundwater for drinking water production. The WBGr and the Drenthe Landscape Foundation (a nature NGO) already own most of the land in this area, which was previously used for agriculture.

The plan is to buy all the land, remove existing embankments and dikes, and construct new dikes further away, allowing for periodical inundation of an area of 480 ha in total (1st phase: 250 ha). This means that the area will be turned (back) into a wetland habitat with an expansive flooded area during winter and high groundwater levels throughout the year. Plans include the removal of a road, building a new bridge, and making a new inflow site to the lake with a new harbour for yachts so that the village of Zuidlaren will be nearer the water and have a new waterfront.

Creation of the partly inundated wetlands will increase the infiltration rate of surface water towards the exploited aquifer. This means that with the same impact on the shallow groundwater levels, the abstraction rate can increase by about 10%.

Since the Tusschenwater area will be turned into a natural reserve (wetland habitat), the groundwater production area will be well protected from the influence of human activities.

Other goals of the project are:

- Catchment of silt, to prevent sedimentation of the Zuidlaarder Lake
- Catchment of nutrients to improve the surface water quality of the Zuidlaarder Lake
- Water storage to prevent flooding downstream (Groningen City)
- Nature development
- Improving the landscape/scenery
- Economic upgrading of the area (tourism).

The whole area has already been designated as a nature area, a water production area and a water storage area in the latest physical plans of the Province and the municipality of Tynaarlo.

The integrated solution for the area (i.e. the Tusschenwater Project) solves several problems. These problems have to be solved anyway. However if each "problem owner" would solve the problem individually, the results might not be as effective as with the integrated solution. Besides, the integrated solution holds several side benefits that the total of individual solutions does not. The purpose of the CEA carried out in this case study is to justify the high costs related to the Tusschenwater Project with both the increased effectiveness (on the problems in the area) of the integrated solution and the several side benefits.

3.3.2 Baseline and identifying the gap

At present the entire area is in agricultural use, in combination with groundwater abstraction. The surface level of the area (polder) is below the surface water levels in the Hunze and Zuidlaarder Lake. All the water in the Hunze Valley passes through the Tusschenwater area, before it flows into the Zuidlaarder Lake and on towards (and through) the City of Groningen. Finally this water flows through the Province of Groningen before it is discharged into the Wadden Sea.

In the present situation the landscape is not attractive and the ecological value is negligible. The water quality of the Zuidlaarder Lake is not good (both the nutrient levels and the turbidity are too high). Because of the relatively high sediment (silt) transport of the Hunze, the Zuidlaarder Lake has to be dredged regularly to accommodate recreational boating. It is clear that the area needs an uplift to make it economically attractive again.

The integrated solution holds several side benefits that the total of individual solutions does not

3.3.3 Costs, income, and effectiveness of measures

The total costs for realizing the Tusschenwater Project are assessed at about €23,5M, including the costs of dredging the entire Zuidlaarder Lake (costs €8M – already spent) to enable a good starting position.

Table 3-23
Investments
Tusschenwater Project

	activity	Costs
1	Value loss of the land by change from agriculture to nature (80% of €3,800,000)	€ 3,040,000
2	reshape the project area	€ 8,900,000
3	modifications well field	€ 2,000,000
4	initial dredging Zuidlaarder Lake	€ 8,000,000
5	accessibility production wells	€ 100,000
6	observation wells groundwater quality (€11.000 per well)	€ 100,000
7	research costs and legal costs	€ 700,000
	Total	**€22,840,000**

The positive effects of the Tusschenwater Project on tourism has been assessed at almost €7 million a year (Regenboog Advies, "Study of the wider impacts of the Tusschenwater Project", 2007). Although this assessment of the economical effects on tourism is uncertain, it indicates clearly that the investment have the potential for a good socio-economic return.

In the following, the pros and cons and the costs and benefits of the Tusschenwater Project will be analysed in both a narrow and broad CEA.

3.3.4 Narrow CEA (cost analysis of integrated approach)

In the narrow CEA a comparison is made only between the hard technical solutions that can easily be expressed in monetary terms.

In the case of Drenthe, the costs related to the realization of the Tusschenwater Project (investments and the structural costs) have been compared with the costs that can be omitted (also investments and structural costs) due to realization of the project. The costs related to the Tusschenwater Project are presented in the 5 tables below. Of these, table 3-23 shows the investment costs for the project.

Value loss of the land

The Tusschenwater Project is projected in an area where the major form of land use is agriculture. If the agricultural land is converted into nature, the value of the land will drop by about 80%. This loss in value has to be covered.

Reshaping the area

The Tusschenwater Project will turn into a wetland habitat with a major function as floodplain (water storage). To accommodate this new function, a new course for the river Hunze through the project area has to be created. Besides natural seeming variations in surface elevation will have to be made, to provide a habitat for the various types of wetland vegetation that belongs there. To have a high potency to hold nutrients and silt, for instance reed fields will have to be planted. Reshaping of the area includes also the erection of technical water management constructions, like dykes, sluices, weirs, etc.

Table 3-24
Recurrent costs
Tusschenwater Project

	activity	Costs
1	monitoring groundwater quantity	€ 25.000
2	monitoring groundwater quality	€ 50.000
3	monitoring surface water quality	€ 25.000
4	monitoring ecology	€ 10.000
5	maintenance nature reserve area	€ 50.000
6	maintenance infrastructure	€ 25.000
	Total	**€ 185.000**

Modifications well field

Since a major part of the project area has a present function as well field for the public drinking water supply, measures are necessary to make sure that the drinking water sources will not be affected. For example, the head construction of each well will have to be elevated, to prevent the inflow of surface water.

Initial dredging of the Zuidlaarder Lake

At present, the water quality of the Zuidlaarder Lake is not good, due to the high concentration of phosphates absorbed to the silt. This causes explosions in algae (also blue algae) growth during summertime. To create the right starting point for any improvement (ecologically, tourism) this silt has to be removed.

Accessibility of the production well

In times of high discharge of the river Hunze, the entire project area will be flooded. To ensure a continuous production of drinking water from the well field, the well need to be accessible at all times. Therefore an access road needs to be constructed, which connects the individual wells.

Observation wells groundwater quality

The entire project area will be turned into a partially flooded wetland habitat. Due to the abstraction of groundwater by the production wells, part of the surface water (about 10%) will infiltrate towards the exploited aquifer. To monitor the possible effects of this infiltration on the quality of the deeper groundwater, a network of observation wells needs to be installed.

Research and legal costs

Before the project can be realized, a lot of research has to be carried out. After the results of these studies are available, a legal process will follow to get all the necessary licenses.

Total investments including land acquisition amounts to € 23.6 million. In the below table the recurrent costs are displayed.

Recurrent costs are € 185.000 annually. These costs are roughly assessed, based on experiences in similar areas in the province.

	Activity	Costs
1	selling of removed toplayer	€ 100.000
2	prevention of Environmental Impact Study (mer) abstraction licence	€ 100.000
3	measures by Groningen-City to prevent flooding	€ 500.000
4	Profit increase drinking water production scale vs developing new pumping station	€ 3.000.000
	Total	**€ 3.700.000**

Table 3-25
Profits/Omitted
investments

	Activity	Costs
1	increase of employment in the recreation sector	€ 250.000
2	reduced costs groundwater protection	€ 0
3	reduced costs compensating loss in productivity agricultural land	€ 15.000
4	less groundwater taxes	€ 185.500
5	reduction in dredging Zuidlaarder Lake	€ 400.000
6	reduction maintenance costs surface water system	€ 5.000
7	reduction in costs to pump up water from the IJssel Lake	€ 10.000
8	profit selling ”nature crops” (reed, willow twigs, etc.)	€ 25.000
9	increase value real estate (real estate tax)	€ 15.750
	Total	**€ 906.250**

Table 3-26
Structural profit/omitted
recurrent costs

Carrying out the project will prevent investments and generate income elsewhere. However, it is very difficult to get exact figures for these financial entries. Nevertheless, based on experience, people from the responsible Waterboard and from a consultancy agency provided the very conservative figures as used in table 3-25.

As it appears, the project when realized will generate income and avoided investments in the € size of 3.7 million.

The structural impacts of the project are shown in table 3-26. The values displayed are subject to some uncertainty due to difficulties getting exact values to indirect effects.

Recreation sector
The effects on the recreation sector have been assessed separately by Regenboog Advies (“Study of the wider impacts of the Tusschenwater Project”, 2007).

Productivity agricultural land
The abstraction of groundwater causes production losses for agriculture. In the Netherlands, the farmers have to be compensated for these losses by the drinking water company. Because agriculture is being removed from the area, The drinking water company doesn’t have these costs anymore.

Groundwater taxes
In the Netherlands a tax of 18.55 eurocents is raised on each cubic meter of abstracted groundwater. If 10% of the abstracted groundwater will be formed by infiltrated surface water, this amount will be free from groundwater taxes. This is a direct result from the realization of the Tusschenwater Project. This will lead to a reduction in costs for the drinking water company.

Reduced dredging Zuidlaarder Lake
Since the flow velocity of the surface water in the wetlands of the project area will slow down, a lot of silt will be trapped in the area. As a result, the Zuidlaarder Lake will not have to be dredged every 20 years anymore. This way, on a annual basis about €400,000 will be saved.

Maintenance costs surface water systems
When the Tusschenwater project will be realized, the total length of dykes in the area will be reduced. As a result the maintenance costs for the dyke will be reduced.

Pumping up IJssel Lake water
Because a large amount of surface water can be stored in the project area, the Waterboard will have to pump up less water (estimated at about 200,000m3 per year à €0.05/m3) from the IJssel Lake towards the Province of Groningen, to prevent it from water shortage in summertime.

Nature crops
The profit of selling nature crops has been assessed by Regenboog Advies (“Study of the wider impacts of the Tusschenwater Project”, 2007).

Real estate
The effects on real estate values has been assessed by Regenboog Advies (“Study of the wider impacts of the Tusschenwater Project”, 2007).

The structural impact of the project can be assessed to an amount of € 906.250 per year.

If the Tusschenwater Project should not be implemented another infrastructure needs to be realized, or the existing infrastructure must be upgraded. Incidental and recurrent profits can also occur as a result of the project’s realization.

From table 3-23 and 3-25 it is clear that a gap of about €20 million exists between the investments and the profits/omitted investments. It should be noted that the initial dredging of the Zuidlaarder Lake (€8 million) is an activity that has to be carried out anyway; independent of the Tusschenwater Project. Without this initial dredging it will not be possible to achieve good water quality in the Zuidlaarder Lake, and this is necessary for both the ecological balance and the recreational use of the lake. By excluding these costs in the balance, the gap would be reduced to €12 million.

There seems to be a simple payback time of about 15 years

Table 3-26 and 3-24 show that the recurrent profits/omitted recurrent costs exceed the recurrent costs of the Tusschenwater Project by a factor of almost 5.

From a top-of-the-head calculation this means that there seems to be a simple payback time for the project of about 15 years. It can therefore be concluded that all the wider impacts of the project, such as a more attractive landscape, greater biodiversity and so forth, are financed by the project itself over a relatively short period.
Profits related to the wider impacts are not always easily substantiated or easy to assess. So, it is desirable to gain more insight into the positive effects of the project. A good way to visualize these positive effects is to carry out an extended CEA.

	Integrated solution	**technical facet solutions**		
Benefits	Tusschenwater	Regular dredging Zuidlaarder Lake 1)	Drain water treatment	Measures to protect Groningen-City from flooding
Groundwater quantity	+			
Surface water quantity	+			
Surface water quality (pesticides)	+		x	
Surface water quality (nutrients)	✓	+	+	
Biodiversity	✓	+	+	
Landscape & tourism	✓	+	x	÷
Flood protection	✓			✓

1) Dredging should take place once every 20 years

÷ = Negative low
= None
x = Positive low
\+ = Positive medium
✓ = Positive high

Table 3-27
Assessment of
benefits per solution

3.3.5 Extended CEA

In the WaterCost-project in Drenthe, the focus is on the Tusschenwater Project, which is an integrated solution aiming to combine:

- Drinking water production (protection and increase of the abstraction rate)
- Improvement of the surface water quality of the Zuidlaarder Lake (reduction of nutrients, sediment)
- Improvement of the economic situation in the area (water recreation on the Zuidlaarder Lake and the Hunze River, recreation focusing on nature and landscape)
- Flood protection of both the City and Province of Groningen

The responsible authorities have already agreed upon an integrated approach to the problems in the area. The Tusschenwater Project shows that it is possible to solve all the above-mentioned issues by cooperating. All the relevant stakeholders have enthusiastically announced their support for the project. The one remaining hurdle is the question of finances. Therefore the WaterCost-project Drenthe focuses on the visualisation of the cost effectiveness of an integrated solution. To do this a comparison has been made between the integrated solution and a number of solutions focussing on individual technical aspects Each solution can have numerous benefits and side-benefits. However, the benefits and side-benefits are hard to quantify. Nevertheless, it is possible to assess the relative effectiveness of each solution compared to the other possible solutions. So all the technical solutions are compared individually to each other and then to the Tusschenwater Project on all relevant and possible benefits.

The assessment of the benefits of the various solutions have been qualitatively derived, as described in the following sections.

Groundwater quantity
One of the results of the Water4All-project has shown that the infiltration capacity of the Tusschenwater Area is much larger than previously assumed. This means that if the area is partially flooded in the future to create wetlands in the floodplain some of the water will infiltrate into the aquifer used by the Drinking Water Company of the Province of Groningen (WBGr) to abstract groundwater for drinking water production. As a result the abstraction rate can increase without increasing the effects of the water abstraction on the wider area. This increase has been estimated to be about 10% (depending on the inundation depth) of the present abstraction rate (10Mm3/yr). Therefore this effect has been assessed as "positive medium".

None of the technical facet solutions have a similar effect on the amount of groundwater that can be abstracted.

Surface water quantity
In the Tusschenwater Project, the area will be turned into extended wetlands/floodplains with much more surface water than at present. Water can be stored in these wetlands during periods of high rainfall and retained for periods of drought. The extra surface water has been assessed as "positive medium". None of the technical facet solutions have a similar effect on the amount of surface water.

Surface water quality (pesticides)

The Hunze water flowing through the Tusschenwater wetlands, will slow down because of the increased surface area. As a result silt will settle in the wetlands and absorb some of the pesticides. Because the topsoil of the wetlands is peaty (with a high amount of organic material) this will also absorb a share of the pesticides and result in improved water quality in the Zuidlaarder Lake.

The extended CEA shows that the integrated solution scores much better than any of the technical facet solutions

The only technical facet solution that has a possible effect on pesticides in surface water is individual drainage water treatment. The area necessary for individual treatment of drainage water is much less than the Tusschenwater wetlands, resulting in much less effectiveness. Therefore the effect of the Tusschenwater Project has been assessed as “positive medium”, whereas the technical facet solution is “positive low”.

Surface water quality (nutrients)

The Hunze water flowing through the Tusschenwater wetlands, will slow down because of the increased surface area of the wetland. Reed and other water plants growing in the flooded area will absorb the nutrients in the surface water (phosphate, nitrate). This will result in a great improvement of water quality in the Zuidlaarder Lake.

In the present situation (i.e. disregarding the Tusschenwater Project), silt settles at the bottom of the Zuidlaarder Lake and absorbs phosphate. However, fish will bring the silt and phosphate back into suspension. Also, diffusion will cause an exchange of phosphate between the silt and the surface water. So if the Hunze water flowing in has low phosphate levels the levels in the lake will increase because of this diffusion. Regular dredging of the Zuidlaarder Lake will remove settled silt from the bottom. Simultaneously the absorbed phosphate will also be removed and temporarily improve the surface water quality.

Individual treatment of drainage water will remove phosphates and nitrates. Potentially this could be as effective as the Tusschenwater wetlands. However it will be practically impossible to treat all the drainage water.

Therefore the effect of the Tusschenwater Project has been assessed as “positive high” and the two technical facet solutions have both been assessed as “positive medium”.

Biodiversity

Realisation of the Tusschenwater Project will lead to a dramatic increase in the area of wetland habitat. The surface water quality of the Zuidlaarder Lake will also improve. One may expect this to lead to a relatively strong increase in biodiversity. Both regular dredging and drain water treatment will improve the surface water quality of the Zuidlaarder Lake. This too would lead to an increase in biodiversity. However, no additional wetland habitat would be created. Hence these two technical facet solutions have been assessed as “positive medium”, where the Tusschenwater Project has been assessed as “positive high”.

Landscape & tourism

Realisation of the Tusschenwater Project will lead to a new extended wetland habitat and a more attractive landscape than the present agricultural landscape. Opening up the area for the public will lead to an increase in eco-tourism and tourism in general in the area. As a result of the wetland habitat, the out-flowing surface water to the Zuidlaarder Lake will be of a much better quality. This will have a positive effect on water related tourism (e.g. swimming, sailing, canoeing, fishing, bird watching etc.). These effects have been researched independently by Regenboog Advies (“Study of the wider impacts of the Tusschenwater Project”, 2007). Due to all these positive effects the Tusschenwater Project has been assessed as “positive high”.

Regular dredging of the Zuidlaarder Lake will lead to an improved surface water quality and better sailing conditions. This will have a positive effect on water related tourism (e.g. swimming, sailing, canoeing, fishing, bird watching, etc.. As a result this technical facet solution has been assessed as “positive medium”.

Individual drain water treatment facilities will have a technical impact on the landscape, making it less attractive. On the other hand, the surface water quality of the Zuidlaarder Lake will improve. Since the land is mainly used for agriculture the impact on the landscape will not be too great, while the impact on the surface water quality is positive. Together these effects have been assessed as “positive low”.

Technical solutions to protect Groningen City from flooding (e.g. higher dikes, pumping stations, wider canals to lead the water around the city), will have a negative impact on the landscape and will form physical barriers for traffic and fauna. Since the technical constructions will be in an urban environment, the effect on the landscape has been assessed as ”negative low”.

Flooding protection

Realizing the Tusschenwater Project will lead to an extra storage capacity for surface water of 1.6 million m3 and after realisation of all other related plans a total of 3 million m3. As a result the effect of this project on flood protection has been assessed as “positive high”. Both regular dredging of the Zuidlaarder Lake and drain water treatment have no effects on flood control. Technical measures to protect Groningen City from flooding can be “tailor made”. Therefore this technical facet solution has also been assessed as “positive high”.

Assessment of narrow costs and extended benefits of the various solutions

The costs per solution (both the integrated solution and the respective technical facet solutions) can be easily assessed, more or less. However it would not be fair to make a comparison of the solutions entirely based on costs. Each solution offers several benefits and side-benefits. Not all benefits can easily be expressed in financial terms. In this sense, it is difficult to define the influence of the solution on the landscape and tourism. Therefore this aspect has been researched separately for the Drenthe project by Regenboog Advies (“Study of the wider impacts of the Tusschenwater Project”, 2007).

While endeavouring to make a fair comparison between the respective solutions, the WaterCost-project decided to apply an extensive CEA. In this extensive CEA scores are given to each solution per benefit (see below table). The sum of these scores leads to a ranking of the solutions. In the Tusschenwater Project it is clear that the integrated approach scores far better than the technical facet solutions.

	Integrated solution	**technical facet solutions**		
Benefits	Tusschenwater	Regular dredging Zuidlaarder Lake 1)	Drain water treatment	Measures to protect Groningen-City from flooding
Groundwater quantity	2	0	0	0
Surface water quantity	2	0	0	0
Surface water quality (pesticides)	2	0	1	0
Surface water quality (nutrients)	3	2	2	0
Biodiversity	3	2	2	0
Landscape & tourism	3	2	1	÷1
Flood protection	3	0	0	3
SUM	**18**	**6**	**6**	**2**

1) Dredging should take place once every 20 years

Table 3-28
Assessment of
benefits per solution

	Integrated solution	**technical facet solutions**		
	Tusschenwater	Regular dredging Zuidlaarder Lake 1)	Drain water treatment	Measures to protect Groningen-City from flooding
Incidental costs in M€	**€ 23.6**	**€ 8**	**€ 4**	**€ 0.5**

1) Dredging should take place once every 20 years

Table 3-29
Costs related
to the solutions

Based on the narrow CEA it was already concluded that the integrated solution seems to have a simple payback time of about 15 years. The extended CEA shows that the integrated solution scores much better (i.e.18, see table 3-28) than any of the technical facet solutions. Therefore, the integrated solution, with all the wider impacts, such as a more attractive landscape, greater biodiversity and so forth, is by far to be preferred above the technical facet solutions. Even more so, because the technical facet solutions have (if it would already be possible to assess it) no or a much longer payback time.

3.3.6 Conclusions

The Tusschenwater Project is an integrated project that serves many stakeholders. During the research period of the project, there have been many meetings with all stakeholders and special attention was paid to the local citizens. These meetings have not always been easy as it was hard to persuade each stakeholder of the project's advantages. Many stakeholders showed a single-minded interest whereby certain obstacles were magnified out of all proportion.

Looking back on this process, many of these discussions would have been much easier, had the present CEA been carried out at that time.Now it has become clear that with a thorough CEA, it is possible to visualize all the advantages of an integrated project. Opposition from local citizens is less intense if they can see beyond the problems and be shown all the advantages realizing the project holds for them. Willingness in this respect contributes to open communication and fruitful discussions, thereby avoiding sometimes long and tedious legal procedures. Effects and costs are transparent for all stakeholders.

3.3.7 Recommendations and discussion

In the Netherlands the national authorities stimulate an integrated approach when solving issues. However, legislation is such that integrated solutions must be presented in different ways to the different Departments responsible for approving and providing the financial resources for projects. This can be extremely frustrating for the initiators of such projects. By including a thorough CEA in one application report the necessity of writing several application reports to obtain subsidies could be avoided.

3.4 The UK, Slea Catchment

3.4.1 Background

The UK case study was based in the same area examined in the previous Water4all project, namely the Slea catchment in Lincolnshire, eastern England. Arable farming is the dominant land use in the catchment and boreholes into the underlying Lincolnshire Limestone aquifer are an important source of public and private water supply. The River Slea itself is fed predominantly by major springs issuing from the groundwater resource.

With a thorough CEA, it is possible to visualize all the advantages of an integrated project.

During the 1990s the western part of the Slea catchment (where the limestone outcrop is exposed) was one of the sites included in the Nitrate Sensitive Areas (NSA) scheme. Under this scheme voluntary, compensated farm management measures (such as conversion of arable land to grassland) were supported under five-year agreements to reduce the levels of nitrate reaching soils and groundwaters. Uptake of the scheme covered 93% of the eligible agricultural area in the Sleaford NSA by 1998 and it was subsequently found to have had a measurable beneficial impact on reducing nitrate leaching to the soil zone (Silgram et al., 2005). Subsequently, however, the NSA scheme was phased out and has been replaced by Nitrate Vulnerable Zones (NVZs) where there are a number of compulsory, uncompensated measures, including restrictions on the extent and timing of fertiliser applications. Nitrate levels in surface and groundwaters are, nevertheless, still such that the catchment has been classified as at high risk of not achieving Water Framework Directive (WFD, 2000/60/EC) objectives by 2015 (Environment Agency, 2004).

In the Water4all project a process of stakeholder engagement was used to develop a series of land-use scenarios for the Slea catchment and the implications of these changes were modelled in terms of projected nitrate levels in soil and groundwater. This analysis found that nitrate concentrations could be reduced by up to 30% by 2015 if some 40% of arable land was converted to wood and/or grassland in the Slea catchment, or if protection zones were targeted in the vicinity of the main springs and boreholes in the area. Economic analysis suggested that the costs of preventing nitrate pollution through changing land use (around £30 per person in the water supply zone) were nearly four times higher than the costs of treating water to reduce the pollution to within regulatory limits (£8 per person). However, it was recognised that this difference would narrow if, as anticipated, treatment costs increase and other potential benefits of land-use change such as more opportunities for recreation and greater biodiversity were taken into account. Further details of this research are given in Lovett et al. (2006) and Hiscock et al. (2007).

Implementation of the WFD in the UK is now starting to focus on the identification of sets of measures to achieve ‘good status’ objectives (Department for Environment, Food and Rural Affairs (Defra), 2007a). When the opportunity arose to undertake a pilot CEA application as part of the WaterCost project it was decided by the UK partner (the Environment Agency) to contribute to ongoing national work through extending the existing Water4all analysis by looking in more detail at the cost-effectiveness of different measures and the non-market benefits associated with them. This study was carried out by researchers from the School of Environmental Sciences, University of East Anglia, Norwich and Economics for the Environment Consultancy Ltd, London. The main steps in the study followed the CEA guidance prepared by Entec (2006), with monetary costs and benefits being analysed in UK pounds (£). In the presentation that follows, however, some key results are also converted to Euros (€).

3.4.2 Baseline and identifying the gap

The study area was defined as an area of 10,800 ha west of the town of Sleaford (see Figure 3-13). This area represents 27, two km grid squares (the highest resolution for which Agricultural Census data are available) that overlaps a modelled steady-state recharge zone for the key springs and water supply boreholes west of Sleaford. A key advantage of using this area was that land-use information could be related to groundwater monitoring data and modelled future nitrate levels through to 2030.

Baseline (2003) land-use areas and livestock numbers for the study area were compiled in an Excel spreadsheet (see Table 3-30). An export coefficient methodology (e.g. Johnes, 1996) was then followed to estimate the total amount of nitrogen (N) entering the soil zone. This involved multiplying the land-use areas and livestock numbers by nitrogen loadings (derivation of these is discussed in Lovett et al., 2006). The initial total was 375,519 kg N per annum. To estimate baseline conditions a number of other assumptions were made:

- Current NVZ requirements apply to all land
- Existing setaside will remain in the future
- 5% of arable land has 6 m wide grass buffer strips
- 25% of spring crops have a preceding cover crop
- 10% of relevant crops (e.g. winter cereals on lighter soils) are established using minimal cultivation

These assumptions reduced the baseline to 364,206 kg N per annum, which when scaled to account for annual effective precipitation translates to 105.8 mg/l NO3 in the soil zone.

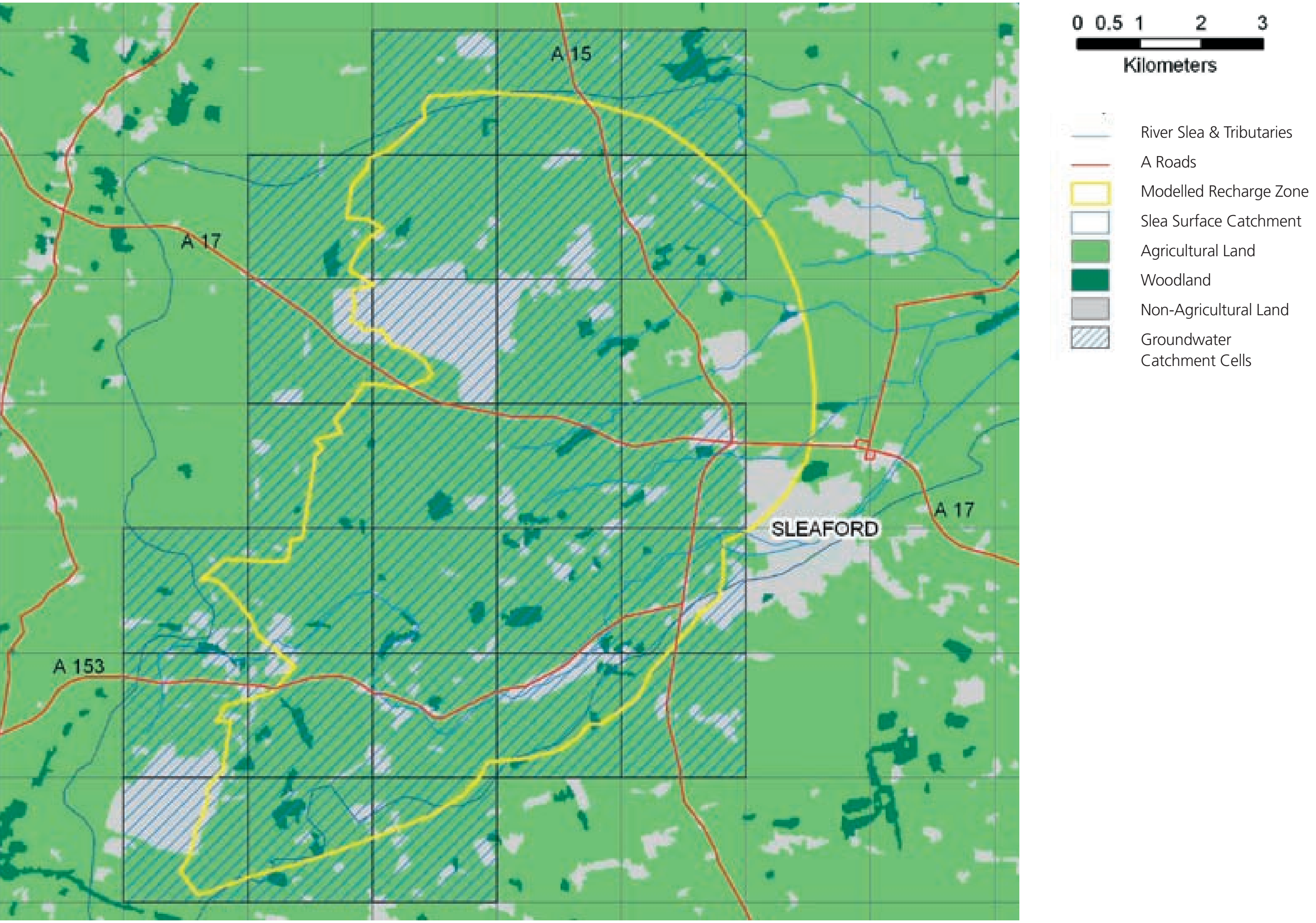


Figure 3-13
The Slea study area.

Land-Use Category	Area (ha)	Area (%)	Loading (N kg/ha)	Total (N kg/a)
Wheat	2328	21.55	44.23	102961
Winter Barley	768	7.11	27.87	21398
Spring Barley	1104	10.23	34.94	38582
Other Cereals	16	0.15	35.64	586
Potatoes	206	1.91	61.13	12581
Sugar Beet	988	9.15	15.26	15079
Field Beans	160	1.48	50.00	8008
Peas	267	2.47	80.00	21350
Oilseed Rape	382	3.54	77.10	29469
Other Arable	107	0.99	16.26	1744
Horticulture	177	1.64	40.78	7204
Permanent Grass	1110	10.28	11.73	13024
Temporary Grass	170	1.58	18.17	3092
Rough Grazing	31	0.28	5.00	153
Bare Fallow	11	0.10	5.00	53
Setaside	1088	10.08	25.00	27206
Unfertilised Grass	342	3.17	5.00	1711
Woodland	264	2.44	8.00	2111
Non-Agricultural	1280	11.85	5.00	6402
Livestock Numbers			**N kg/head**	
Cattle	1290		11.93	15392
Sheep	7236		1.72	12424
Pigs	3999		1.692	6767
Poultry	583682		0.0275	16051
Humans	8693		1.4	12170
			Total	**375519**

Table 3-30
Baseline land use and livestock information for the study area.

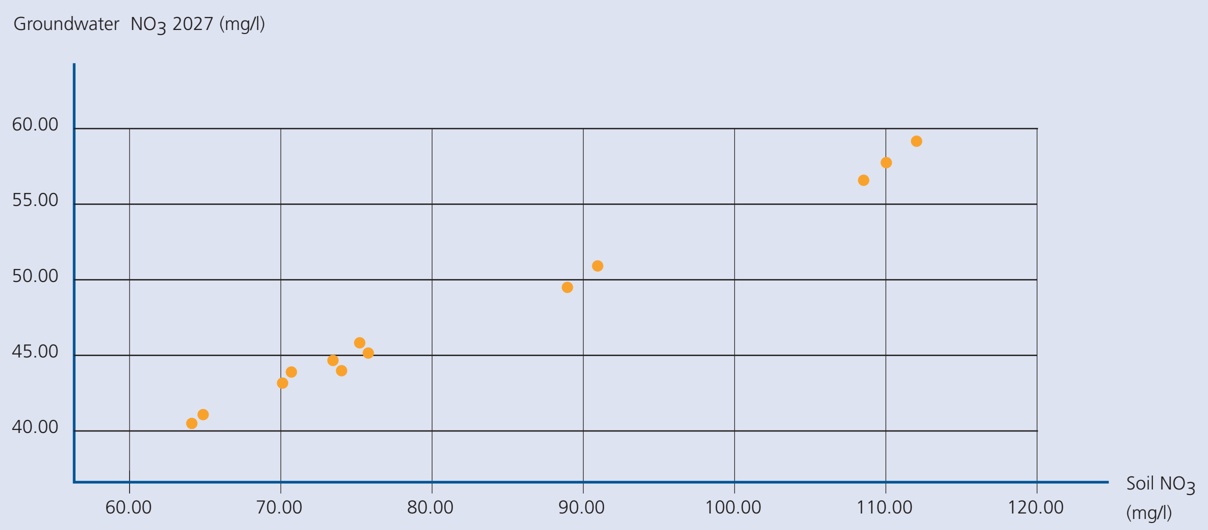


Figure 3-14 Relationship between predicted soil and groundwater nitrate levels in a range of land-use scenarios.

A groundwater nitrate target of 50 mg/l NO3 was defined to reflect the quality standard set in the recent Groundwater Daughter Directive (2006/118/EC). Further UK work on groundwater classification also proposes a lower screening value of 37.5 mg/l NO3 (UKTAG, 2007). If there was no natural attenuation of nitrate (e.g. through bacterial reduction) between the soil zone and aquifer then a target of 50 mg/l NO3 would equate to a N input of 172,112 kg across the study area. This represents 47% of the baseline input of 364,206 kg N per annum. However, results from the groundwater modelling carried out during the Water4all project (see Lovett et al., 2006; Hiscock et al., 2007) suggested there was a significant reduction in nitrate levels between the soil zone and aquifer boreholes (as a result of natural attenuation and mixing with low-nitrate water drawn from the confined aquifer). Figure 3-14 uses a number of Water4all scenario results to plot the relationship between soil zone NO3 levels in the study area and predicted aquifer NO3 concentrations at the Clay Hill borehole (an important public water supply source west of Sleaford). The aquifer concentrations were estimated for 2027 (the end date for the third round of WFD implementation) since the longer-term upward trend observed in the Water4all modelling results and an aquifer flushing time of around 20 years meant that predictions for an earlier year (e.g. the initial WFD target of 2015) would be unlikely to represent an equilibrium situation.

A strong linear trend is apparent in the scenario results plotted in Figure 3-14. This made it possible to derive a regression equation relating the soil and groundwater values with a high degree of fit ($r^2 = 0.99$) and then take borehole concentrations of 50 and 37.5 mg/l NO3 in 2027 and work backwards to estimate the associated soil zone NO3 and kg N input. As a consequence, it was possible to assess the cost-effectiveness of several different packages of measures with respect to a range of soil and groundwater NO3 targets.

3.4.3 Description of measures and scenarios

The selection of measures was based on discussions with stakeholders in the study area, findings from the previous Water4all project and a recent inventory of measures to control diffuse water pollution from agriculture produced for the UK Defra (Cuttle et al., 2006; 2007). This inventory provides cost (investment and operational) and effectiveness information for 44 measures, but many of these were not relevant to the Slea catchment. Sometimes this was because they concerned types of agriculture (e.g. dairy farming) that were of limited importance in the study area and in other cases the measures were already being implemented (e.g. as part of the NVZ requirements). Where considered appropriate, information from the Defra inventory was used, but for some measures this had to be supplemented by other reference sources and advice from a number of experts (e.g. local agronomists, advisory organizations such as ADAS, the Forestry Commission etc).

Seven measures were examined in detail. These consisted of four measures primarily concerned with land management and three where arable land was converted to alternative uses. No measures specifically concerned with livestock management were considered as this type of farming accounts for a relatively small proportion of N input within the study area (see Table 3-30). Perennial energy crops (e.g. Miscanthus) were included as an alternative land use since there are current plans to build a biomass power plant in Sleaford (www.sleafordrep.co.uk).

Table 3-31 summarises the cost and effectiveness estimates used for each measure, along with any constraints on how they were applied. For the purpose of the CEA the seven measures were implemented to varying extents in a set of scenarios labelled as follows:

- Realistic – achievable under current policies and budget support
- Optimistic – assuming higher political priority and budget support
- Utopian – ignoring real-world constraints (loss of food production etc) the level of nitrate reduction that could be achieved through substantial land-use modifications

Table 3-32 lists estimated baseline levels of different measures in the study area and how they were assumed to change under the different scenarios that were evaluated. There were two variants of the Optimistic scenario, the first involving only four land management measures and the second also including some land-use changes. The areas of unfertilised grass and farm woodland in the Utopian scenario were defined to match those in the Protection Zone scenario from the Water4all study (Lovett et al., 2006; Hiscock et al, 2007) since this option had appeared one of the most promising to examine further.

The implementation of the CEA was carried out in a series of Excel spreadsheets. A separate spreadsheet was created for each scenario and the baseline land-use areas from Table 3-30 were adjusted to reflect the changes specified in Table 3-32. This adjustment process typically involved scaling down the baseline areas for different arable crops in proportion to their share of total arable land and then increasing the extents of unfertilised grass, woodland and energy crops as required.

Once the scenario land-use profiles had been adjusted the relevant crop or grassland areas were multiplied by the appropriate per hectare cost and N loading reductions for each measure (see Table 3-31). Overall costs and N input changes were then derived for each measure and the former divided by the latter to produce cost-effectiveness ratios (C/E, i.e. the cost (in £) per kg of N reduction). Costs and N reductions for individual measures were also summed to produce totals for each scenario package. With the scenarios involving more extensive applications of measures (e.g. Utopian) it was also necessary to take care to avoid double-counting (e.g. situations where two incompatible measures were applied to the same piece of land). In the small number of cases where this situation occurred, the solution adopted was to implement only the most cost-effective measure.

Table 3-31
Summary of cost and
effectiveness information
for the seven measures.

Measure	Cost	N Effectiveness	Application Constraints
Establish Cover Crops in Autumn	£67.5/ha/yr to cover seed purchase and cultivation (Cuttle et al., 2007, p.94).	Reduce N loading for relevant crops by 40% (based on results from Lord et al., 1999 and Johnson et al., 2002).	Only relevant for spring cultivated crops. Applied to spring barley, potatoes, sugar beet, field beans, peas and other arable (e.g. linseed and carrots) categories.
Adopt Minimal Cultivation Systems	Saving of £40/ha/yr compared to conventional ploughing (Cuttle et al., 2007, p.13).	Reduce N loading for relevant crops by 2.5 kg N/ha (Cuttle et al., 2007, p.13).	Applied to winter cereals and oilseed rape grown in the 15 two km cells that have the majority of their land on the limestone outcrop.
Reduce Fertiliser Application Rates by 20%a	£5.3/ha/yr for arable land, £18.9/ ha/yr for grassland based on proportion of beef cattle in study area livestock (Cuttle et al., 2007, p.45).	Reduce N loading by 7.5 kg N/ha for arable land and temporary grass, 3.5 kg N/ha for permanent grass (Cuttle et al., 2007, pp.45-6).	Not relevant for crops where no N is applied e.g. legumes.
Establish 6m in-field Grass Buffer Strips	£31.6/ha/yr to reflect loss of output and management costs (Cuttle et al., 2007, p.20).	Replace the loading for 10% of the relevant arable land with that for unfertilised grass (i.e. 5 kg/ha).	-
Convert Arable Land to Grassland	£350 ha/yr to cover loss of output and management costs (Cuttle et al., 2006, p.5). Note that this figure would need to be increased by £50 to £100 if any compensation was to reflect recent increases in cereal prices (e.g. compare Nix, 2006, p.17 and 2007, p.16).	Replace N loadings for arable land with unfertilised grass value of 5 kg/ha (from Water4all study, Lovett et al., 2006, pp.48-50).	-
Convert Arable Land to Farm Woodland	Planting cost of £2,500/ha equals an annual payment of £257/ha (based on a 6% interest rate and 15 year repayment period). Adding an average annual maintenance cost of £76 and loss of output compensation at £300/ ha/year totals £633/ha/yr (based on advice from the Forestry Commission).	Replace N loadings for arable land with woodland value of 8 kg/ha (from Water4all study, Lovett et al., 2006, pp.48-50).	-
Convert Arable Land to Perennial Energy Crops (Miscanthus)	Planting cost of £2,500/ha equals an annual payment of £257/ha (based on a 6% interest rate and 15 year repayment period). Compensation for difference in gross margins (compared to wheat/oilseed rape) would add £200/ha/year to total £457/ha/yr (based on Nix, 2006, pp.71-3 and SAC/Uni. Cambridge (2005) p. vii).	Replace N loadings for arable land with woodland value of 8 kg/ha (from Water4all study, Lovett et al., 2006, pp.48-50).	-

Table 3-32
Packages of measures
examined in the CEA.

Measure	Baseline	Realistic Scenario	Optimistic Scenario	Utopian Scenario
Establish Cover Crops in Autumn	25% of relevant crops	50% of relevant crops	75% of relevant crops*	100% of relevant crops
Adopt Minimal Cultivation Systems	10% of relevant crops	20% of relevant crops	30% of relevant crops*	100% of relevant crops
Reduce Fertiliser Application Rates	-	20% reduction in N application to 30% of cropland and pasture	20% reduction in N application to 60% of cropland and pasture*	20% reduction in N application to 100% of cropland and pasture
Establish Grass Buffer Strips	5% of arable land with 6m grass buffer strips	20% of arable land with 6m grass buffer strips	50% of arable land with 6m grass buffer strips*	100% of arable land with 6m grass buffer strips
Convert Arable Land to Grassland	342 ha of unfertilised grass in study area	684 ha of unfertilised grassin study area	1,080 ha of unfertilised grass in study area	1,550 ha of unfertilised grass in study area
Convert Arable Land to Farm Woodland	264 ha in study area	Plant 20 ha	Plant 50 ha	Plant 540 ha
Convert Arable Land to Energy Crops	None	Plant 40 ha of Miscanthus	Plant 100 ha of Miscanthus	Plant 250 ha of Miscanthus

* Only these four measures were included in variant 1 of the Optimistic scenario. Variant 2 involved all seven changes.

Table 3-33
CEA results for
the Realistic scenario.

Measure	N Reduction (kg)	Cost (£)	C/E Ratio (£)	C/E Ratio (€*)
Minimal Cultivation	834	-13,336	-16.0	-23.5
Reduced Fertiliser	14,376	16,321	1.1	1.7
Cover Crops	18,265	89,690	4.9	7.2
Buffer Strips	4,249	38,561	9.1	13.3
Arable to Grassland	11,970	119,700	10.0	14.7
Arable to Energy Croaps	1,280	18,280	14.3	21.0
Arable to Woodland	640	12,660	19.8	29.1
Total	**48,585**	**223,073**	**4.6**	**6.7**

* Using exchange rate of £1 = €1.47

3.4.4 Narrow CEA results

Table 3-33 shows the cost-effectiveness results for individual measures in the Realistic scenario. The measures are ordered from the lowest cost-effectiveness ratio upwards and the results indicate that those involving land management perform rather better than those concerned with land-use changes. Minimal cultivation stands out as the most cost-effective measure, but this technique is only applicable in certain soil conditions and is more expensive for farms that need to maintain conventional ploughing as well. Adoption is therefore difficult in some regions.

Most measures had the same C/E ratio in all the scenarios, but one exception was that for reducing fertiliser application rates which increased slightly from £1.1 per kg N in the Realistic scenario to £1.3 per kg N in the Utopian one. This was because temporary and permanent grass (which have a higher C/E ratio than arable land) occupied a larger proportion of the area to which the measure was applied in the scenarios involving greater land-use change.

Table 3-34 Summary of CEA scenario results.

Scenario	Total N (kg)	Cost (£ 000)	Cost (€ 000)	C/E Ratio (£)	Soil NO_3 mg/l	Groundwater NO_3 mg/l in 2027
Baseline	364,206				106.0	57.0
Realistic	312,592	281.8	414.4	5.46	90.8	51.1
Optimistic 1	291,898	258.5	380.1	3.58	84.8	48.8
Optimistic 2	270,724	561.0	824.6	6.00	78.7	46.4
Utopian	214,085	1,149.6	1,689.9	7.66	62.2	40.1
Groundwater target NO_3 50 mg/l in 2027	302,920				88.0	50.0
Groundwater target NO_3 37.5 mg/l in 2027	191,779				55.7	37.5
Soil NO_3 50 mg/l target	172,112				50.0	35.3

Results from applying the different scenario packages are summarised in Table 3-34. A key feature of these findings is that they suggest that the two Optimistic scenarios could meet a groundwater target of 50 mg/l NO3 in 2027. The Utopian combination does not meet a 37.5 mg/l NO3 objective, but the predicted soil NO3 level equates to an estimated groundwater concentration of 40 mg/l NO3 in 2027. None of the scenarios are close to meeting a 50 mg/l NO3 in soil objective, which highlights the importance of maintaining natural attenuation and mixing processes in the aquifer for the future viability of drinking water abstraction at sites such as the Clay Hill borehole. Another important detail in Table 3-34 is the way in which the average C/E ratio increases between scenarios. This is essentially because more expensive measures (such as conversion of arable land to woodland or grass) are needed to achieve larger amounts of N reduction. A final point is that these costs would increase further (e.g. by at least £75,000 for the Utopian scenario) if the recent increase in cereal prices was reflected in compensation payments.

3.4.5 Extending the CEA to consider non-market benefits

A literature review was carried out to assess the non-market (or side) benefits associated with the seven measures, with the aim of determining whether the inclusion of these would alter the ranking of options. The results of this exercise suggested some potential benefits (or negative impacts) could be identified for each measure (e.g. the scope for cover crops to reduce run-off and soil erosion, advantages of grass margins for biodiversity), but that the overall extent of these was unlikely to be significant for the four land management measures. Attention was therefore focused on the three measures which involved land-use changes, particularly in relation to non-market benefits associated with enhanced visual amenity, recreation or reductions in greenhouse gas emissions.

Converting arable land to grassland

Cuttle et al. (2007, p.8) noted that this measure would create savings in greenhouse gas emissions due to reduced nitrous oxide emissions and improving the soil's ability to sequester carbon. There is a small amount of evidence in the economic valuation literature that the public prefers extensive grassland over arable farmland. For example, Willis and Garrod (1993) undertook a study on preferences for hypothesised landscape changes in the Yorkshire Dales and found that very few respondents expressed preferences for more intensive or semi-intensive agricultural landscapes; however this could be due to an attachment to what is clearly a very high quality landscape. Many studies on agricultural landscapes (e.g. Bullock and Kay (1997), eftec (2006), Willis and Garrod (1993)) have examined changes to upland landscapes, which are rather untypical agricultural landscapes. eftec (2002) reviewed all available industrialised country studies on the value of undeveloped land; no UK studies were found valuing intensive farmland, but studies from North America and Sweden suggested that intensive farmland is valued less in terms of the visual amenity it provides than extensively farmed land, for example in Environmentally Sensitive Areas. eftec (2006) undertook a valuation of upland landscape attributes and found in focus group discussions that both arable land and improved grassland were relatively unpopular landscape types. As a consequence they were not examined further in subsequent stages of the valuation study. Therefore, while there might be a non-market benefit in terms of visual amenity from converting arable land to extensive grassland, particularly if it were managed in an environmentally sensitive way, the evidence is not particularly strong or conclusive.

Converting arable land to farm woodland

There have been many studies on the non-market benefits provided by woodland. The most comprehensive meta-study of these in the UK is that by Willis et al. (2003), the results from which are summarised in Table 3-35. Woodlands provide a range of non-market benefits such as recreation, landscape amenity, habitat provision, carbon sequestration and air quality regulation. The benefits provided differ according to the type of woodland i.e., broadleaved or coniferous, upland or lowland, natural or plantation. One example

Table 3-35
Estimates of non-market benefits of woodland

Recreation	£1.66 - 2.75 per visit
Visual amenity	£269 per annum per household for urban fringe households with a woodland view
Biodiversity	£0.35 - £1.13 per annum per household, depending on the type of woodland
Carbon sequestration	£6.67 per tonne (note that this is substantially lower than official UaK government estimates of the social cost of carbon i.e. £70 per tonne)
Air quality regulation	£125,000 for each death avoided by 1 year and £602 per hospital stay avoided due to reduced respiratory illness

of an individual study examining the non-market benefits of woodland is Bateman et al. (1996), which investigated household willingness to pay for a 100-acre community woodland near Wantage in Oxfordshire and found an estimated value of £250 per acre.

Converting arable land to energy crops
There is a limited literature on the non-market benefits of planting perennial energy crops, though it is widely recognised that they can contribute to reductions in greenhouse emissions (e.g. Defra, DTI and DfT, 2007). On the other hand, some concerns have been expressed about the potential environmental damage resulting from extensive planting of such crops (e.g. European Environment Agency, 2006; Wildlife and Countryside Link, 2007). For instance, the dense and relatively tall (3-4 m) appearance of Miscanthus may mean that it is not perceived as a particularly natural landscape feature and so is unlikely to enhance visual amenity.

3.4.6 Implications for CEA results

Based on the above review it was decided to focus on the non-market benefits of woodland. A 'break-even' analysis was carried out to assess the extent to which the non-market benefits associated with the amount of woodland planting would be sufficient to overturn the differences in costs between scenarios. This approach owes much to the opportunity cost concept which underpins much of standard economic theory (Freeman, 2003) and can be explained as follows:

In order for a more costly option J to be preferred to a cheaper option, K, the following inequality must be satisfied:

$B_J - C_J > B_K - C_K$

where BJ, BK are the non-market benefits of J and K, and C_J, C_K are the costs. Assume that the non-market benefits per hectare of J and K are the same, i.e. equal to b (note that this may not be a safe assumption if the area of woodland under the two options is vastly different).

K

J	1	2	3	4
1	–	$\frac{23,330}{20} = 1,166$	See cell (3,1)	See cell (4,1)
2	See cell (1,2)	–	See cell (3,2)	See cell (4,2)
3	$\frac{279,095}{30} = 9,303$	$\frac{302,425}{50} = 6,049$	–	See cell (4,1)
4	$\frac{867,726}{520} = 1,669$	$\frac{891,056}{540} = 1,650$	$\frac{588,630}{490} = 1,201$	–

Table 3-36
Break-even comparison for scenarios.

$bA_J - C_J > bA_K - C_K$

where A_J, A_K are the areas of woodland under the two scenarios. This gives:

$b > C_J - C_K$

$A_J - A_K$

Therefore we have to decide whether the non-market benefits per hectare of woodland are likely to be greater than the difference between costs divided by the difference between areas.

Table 3-36 shows this calculation for each combination of the scenarios presented in Table 3-32; note that this is arranged so that in each case $C_J > C_K$. Costs are in £. The scenarios are numbered from 1 to 4 as follows: 1 = Realistic, 2 = Optimistic Variant 1, 3 = Optimistic Variant 2 and 4 = Utopian so that:

$C_1 = 281,877$	$A_1 = 20$
$C_2 = 258,547$	$A_2 = 0$
$C_3 = 560,972$	$A_3 = 50$
$C_4 = 1,149,602$	$A_4 = 540$

i.e. C is the cost associated with the scenario and A is the additional area of woodland involved.

Each cell shows the figure which annual per hectare woodland benefits would have to exceed in order for scenario J to be preferable to scenario K. The next step is to examine the main potential benefits to see if this is possible.

Carbon sequestration
According to Willis et al. (2003), a hectare of lowland woodland sequesters about 100 tC over the lifetime of the trees. Using Defra's official figure for the social cost of carbon, this gives:

100 x £70 = £7,000 per ha.

However, note that this is not an annual benefit, takes no account of discounting over time and relies on the woodland being around for the foreseeable future. In addition, there are many uncertainties in estimating the social cost of carbon (Tol, 2005) and some of the calculated values are much lower than £70 per tC. For example, using the figure of £6.67 tC reported by Willis et al. (2003) would reduce the above total by more than 90%. Overall, given that figures of £1,201 to £9,303 per ha per year are required, it seems fair to conclude that carbon sequestration alone does not provide a substantial enough benefit over several years for more costly scenarios to be preferred.

Recreation

Bateman et al. (1996) estimated recreation/amenity values for a 100 acre community woodland near Wantage, Oxfordshire of £9.94 per household per year, or £0.82 per visit. This value is lower than that reported by Willis et al. (2003) in Table 3-35.

To transfer these values to Sleaford, we ideally need to adjust for average (mean) household income in the area, which is probably lower around Sleaford than in Oxfordshire. However, mean household income is not disclosed by the Wantage study, so we will leave the values unadjusted.

Using the 'per household' figure

$B = V_H \times N_H$ and $N_H = N_P / P_H$

where:

B = total benefit (per annum)

V_H = value per household (i.e. £9.94 = 10)

N_H = number of households

N_P = population in study area

P_H = average number of people per household (i.e. N_P / N_H)

Values for NP and PH were estimated from official statistics as follows:

$N_P = 8,700$ and $P_H = 2.38$ so

$$B = \frac{10 \times 8700}{2.38} = £36,600$$

This suggests that the recreational benefits of a small woodland (of about 50 ha) are likely to be around £40,000 per annum. For the Realistic and Optimistic Variant 2 scenarios such a recreational benefit total translates to about £800 - £2,000 per hectare. However, it is inappropriate to transfer the per hectare figure to the Utopian scenario which has a woodland area approximately ten times larger. This is because it is well-recognised by economic theory and empirical research that the principle of diminishing marginal returns applies to the value which individuals derive from consuming units of a good (e.g. see Loomis and Helfand, 2001, pp. 298-305). So, while the provision of initial, small recreational woodland may generate a substantial benefit for nearby populations, doubling the size of the woodland will not double the benefit. Rather, the increase will be more modest. This diminishing relationship will continue with even larger increases in size.

Visual amenity

It is not possible to transfer the value of around £270 per annum reported by Willis et al. (2003) without data on the number of urban fringe houses which would have a view of the woodland. Assuming that Sleaford is large enough to be regarded as urban, consider the smallest difference in cost between scenarios:

$C_1 – C_2 = £23,330$

For visual amenity benefits alone to make Scenario 1 (Realistic) preferable there would have to be:

$$\frac{23,330}{270} = 109$$

households with a view of the woodland which is not unfeasible. This total increases to about 3,300 households for the largest variation in costs (C4 – C2). Such a situation is unlikely given that the total population of Sleaford is approximately 15,000 people. It might be argued that amenity values would be less while trees were growing, but there would still be some benefits associated with the provision of an open-access recreation area.

Assessment

Several factors complicate the break-even analysis of non-market benefits. These include:

- the available value estimates being reported in different units;
- limited data related to the specific case of Sleaford (and hence a need to rely on benefit transfer);
- the fact that the woodland area in the Utopian scenario is much greater than the others.

The combination of these factors means that although relatively low thresholds of per hectare benefits from woodland (£1,201/ha to £1669/ha in Table 3-36) are required to prefer the Utopian scenario over other options, it is difficult to be confident that the values available can be reliably applied to the larger area of planting involved.

There is more evidence that where both the woodland size and cost difference is small then the benefits of the woodland would outweigh the additional cost. For example, it could be argued that the Realistic scenario is preferable to Optimistic Variant 1, as the cost difference of £23,330 could easily be negated by the combined recreational and visual amenity benefits if at least 50 houses could see the wood (e.g. 20 x £800 for recreation + 50 x £270 for visual amenity = £29,500). On the other hand, if the costs associated with converting arable land to grassland are increased by £50 per ha (to reflect a rise in cereal prices) then the difference between the scenarios exceeds £40,000 and the case is less clear-cut. This consideration also applies to the Optimistic Variant 2 scenario where the additional non-market benefits do not seem sufficient to outweigh the cost differences compared to the Realistic and Optimistic Variant 1 packages of measures.

3.4.7 Conclusions and recommendations

It would appear that meeting a future groundwater NO3 target of 50 mg/l is feasible with enhanced resources to existing schemes or proposals currently under discussion

The results of the Narrow CEA suggest that either of the Optimistic scenarios could meet a groundwater NO3 target of 50 mg/l in 2027. The most cost-effective solution would be the first variant of the Optimistic scenario which involves land management measures such as minimal cultivation, reduced fertiliser inputs, cover crops and grass buffer strips rather than extensive changes of arable land use to grass or woodland. However, assessment of non-market benefits suggests that the planting of 20 ha of woodland in the Realistic scenario could generate sufficient benefits to more than cover the difference in costs compared to the Optimistic scenario. The strength of this conclusion depends on the compensation cost assigned to the conversion of arable land to unfertilised grassland, but it also implies that a combination worthy of further investigation could be the first variant of the Optimistic scenario along with the element of woodland planting from the Realistic scenario. Calculated on a similar basis to the other scenarios this combined package of measures would have a total N input of 291,467 kg and an annual cost of £270,390, giving a C/E ratio of 3.72. The estimated soil zone NO3 level would be 84.7 mg/l and the predicted groundwater NO3 concentration in 2027 is 48.8 mg/l.

Results from the Narrow CEA also suggest that the Utopian scenario could reduce future groundwater NO3 levels to around 40 mg/l in 2027. This is a similar conclusion to earlier analysis in the Water4all project. However, the cost of such a package of measures is much greater (at least four times that for the first variant of the Optimistic scenario) and the assessment of non-market benefits does not suggest that these are sufficient to cover the difference. However, it should be noted that there is some uncertainty in this assessment as it is difficult to translate the benefit levels estimated for smaller areas of woodland to the much larger amount of planting envisaged in the Utopian scenario. This situation suggests that there could be some merit in future research to assess the benefits of larger areas of woodland planting.

The combined package of measures identified in this case study could be implemented through extensions of existing schemes in the UK (e.g. Environmental Stewardship and the English Woodland Grant Scheme) or as part of the Water Protection Zone concept outlined in a current policy consultation document (Defra, 2007b). This approach would reduce administrative costs, though it would require increases in the budgets of organisations such as Natural England and the Forestry Commission. The cost of some measures would fall upon the farming community and therefore a key issue would be the balance between a regulatory approach (e.g. a proposal to require cover crops is included in the current consultation on changes to Nitrate Vulnerable Zones in the UK, Defra, 2007c) and incentives in a scheme such as Environmental Stewardship. For the latter a key issue is likely to be world prices for products such as grain, since the recent increases have made it much more valuable to keep yields as high as possible.

Overall, the results of this study suggest that CEA can be used to help identify a package of measures that could achieve potential groundwater quality targets under the Water Framework Directive. It would appear that meeting a future groundwater NO3 target of 50 mg/l is feasible with enhanced resources to existing schemes or proposals currently under discussion. Measures can also be identified that would meet a more demanding NO3 target of around 40 mg/l, but the costs of implementing this package would be considerable and at present do not appear to be compensated for by the non-market benefits that would be generated.

4

Recommendations for a CEA methodology

As stated in the introduction of this report, the main objective has been to test available methodologies for CEA with regard to their applicability in a groundwater context and to find a way to include wider impacts to the CEA in a sound and feasible way.

The methodological point of departure was taken in the Entec (2006) methodology. Working from that, the 4 case studies show that the methodology is applicable in a groundwater context. This work furthermore shows that the Entec (2006) provide a sufficiently flexible framework for CEA to be undertaken at different levels (from screening to decision support). As for the latter, the Entec (2006) was extended to include specific welfare economic features (see chapter 2). It is generally recommended to apply this approach in analyses providing decision support in order to make sure that investment decisions are taken on a theoretically consistent background in each case, and that investment decisions become comparable across river basins and countries.

A so-called "modified benefit transfer" approach, including wider impacts, was suggested during the project. The modified approach to benefit transfer was first used in Denmark, and has been applied in a study for the Danish EPA. The approach takes point of departure in a narrow CEA. The narrow analysis ranks different measures with respect to their relative cost-effectiveness. The question is whether the wider impact can be expected to change the ranking of the measures. In order to provide an answer to this question, first the kind of wider impacts associated with the different measures are identified. When this is done, research on relevant studies of the value of the wider impacts are carried out. These studies give an indication of the relative sizes of the wider impacts. If these relative sizes point to the same ranking as was done by the narrow CEA, no further action is taken. If the wider impacts suggest that the ranking might be changed, the actual relative size of the value of the wider impacts needs to be considered. Based on the studies, an argument is produced either in favour of maintaining or changing the ranking.

It is recommended to apply the modified benefit transfer approach to include wider impacts in the CEA

The four case studies all take wider impacts into consideration, using aspects of the modified benefit transfer approach ranging from multi-criteria analysis based on expert judgement (German and Dutch case studies), over research based semi-quantification (the Danish case study) to development of a stringent mathematical framework for inclusion of wider impacts (the British study).

The application of the modified approach to benefit transfer in the case studies explores a wide range of possible levels for analysis. It is recommended to apply the modified benefit transfer approach to include wider impacts in the CEA, since the approach is found to be feasible and yet resting on a sound methodological background.

References

5

A

Aalborg Kommune (2001): Delindsatsplan for Grundvandsbeskyttelse. Drastrupområdet.

Aalborg Municipality (2006): The Concept of Cost Effectiveness – What It Is and How To Define It.

B

Bateman, I.J., Diamand, E., Langford, I.H. and Jones, A. (1996), Household willingness to pay and farmers' willingness to accept compensation for establishing a recreational woodland, Journal of Environmental Planning and Management 39: 21-43.

Bateman, I., Day, B., Dupont, D., Georgiou, S., Gonca Noceda Matias, N., Morimoto, S., Subramanian, L. (2006). Does Phosphate Treatment for Prevention of Eutrophication pass the Benefit-Cost-Test? CSERGE Working Paper EDM 06-13. Norwich: University of East Anglia.

Bateman, I.J., Diamond, E., Langford, I.H., Jones, A. (1996). Household Willingness to Pay and Farmers' Willingness to Accept Compensation for Establishing a Recreational Woodland. Journal of Environmental Planning and Management 39 (1): 21-43

Beckmann, V., Tiemann, S., Reuter, K. and Hagedorn, K. (2003). Kosten der Erreichung von Umweltqualitätszielen in ausgewählten Regionen durch Umstellung auf Ökologischen Landbau im Vergleich zu anderen Agrarumweltmaßnahmen unter besonderer Berücksichtigung von Administrations- und Kontrollkosten. Report,

Bullock, C.H. and Kay, J. (1997), Preservation and change in the upland landscape: the public benefits of grazing management, Journal of Environmental Planning and Management 40:315-334.

C

Cuttle, S.P., Haygarth, P.M., Chadwick, D.R., Newell-Price, P., Harris, D. Shepherd, M.A., Chambers, B.J., Humphrey R., (2006), An Inventory of Measures to Control Diffuse Water Pollution from Agriculture: User Manual, Interim Report for Defra, London.

Cuttle, S.P., Macleod, C.J.A., Chadwick, D.R., Scholefield, D., Haygarth, P.M., Newell-Price, P., Harris, D. Shepherd, M.A., Chambers, B.J., Humphrey R., (2007), An Inventory of Measures to Control Diffuse Water Pollution from Agriculture: User Manual, Final Report for Defra, London.

D

Dabbert, S. and Häring, A.M. (2003). Vom Aschenputtel zum Lieblingskind. Zur politischen Förderung des Ökolandbaus. GAIA 12, No. 2: 100-156

Damgaard, C., E. Erichsen & H. Huusom (2001): Samfundsøkonomisk projektvurdering af skovrejsning ved Vollerup.

Danish Forest and Nature Agency (2007): Privat Skovrejsning Tilskud, Vejledning, 2007 www.skovognatur.dk/Service/Tilskud/PrivatSkovdrift/Skovrejsning/

Defra (2007a), Water Framework Directive, available at www.defra.gov.uk/ environment/water/wfd/index.htm.

Defra (2007b), The Protection of Waters Against Pollution from Agriculture: Consultation on Implementation of the Nitrates Directive in England, Department for Environment, Food and Rural Affairs, London.

Defra (2007c), The Protection of Waters Against Pollution from Agriculture: Consultation on Diffuse Sources in England, Department for Environment, Food and Rural Affairs, London.

Defra (2003) Economic Evaluation of the Organic Farming Scheme. Annex 6: Evidence of the wider public benefits of organic agriculture – literature review. Centre for Rural Economics Research, Department of Land Economy, University of Cambridge. Department for Environment, Food and Rural Affairs (DEFRA). www.defra.gov.uk/farm/

Department for Environment, Food and Rural Affairs, Department of Trade and Industry and Department for Transport (2007), UK Biomass Strategy, Industrial Crops Division, Department for Environment, Food and Rural Affairs, London.

Donné van Engelen, Christian Seidelin Sørensen, Rob van der Veeren, David Barton, Kabir Queb (2007): Cost-effectiveness analysis for the implementation of the EU Water Framework Directive. Being published.

Dubgaard et al. (2003): Cost-benefit analysis of the Skjern river restoration in Denmark, Fødevareøkonomisk Institut.

Dubgaard, A. (1998). Economic Valuation of Recreational Benefits from Danish Forests. In: Dabbert, S., Dubgaard, A., Slangen, L. And Whitby, M. The Economics of Landscape and Wildlife Conservation. Oxon, UK: CAB International.

E

eftec (2002), The External Benefits of Undeveloped Land: a Review of the Economic Literature, Report to the Department for Transport, Local Government and the Regions, London.

eftec (2006), Economic Valuation of Environmental Impacts in the Severely Disadvantaged Areas, Report to the Department for Environment, Food and Rural Affairs, London.

Elsasser, P. (1996). Der Erholungswert des Waldes. Monetäre Bewertung der E holungsleistung ausgewählter Wälder in Deutschland. Schriften zur Forstökonomie. Frankfurt am Main, Sauerländer

Entec (2006), Project 2C: Benchmark Costs Database and Guidance on the Application of Cost-Effectiveness Methodology, Report to Defra Collaborative Research Programme Steering Group, London.

Entec (2006): CEA Guidance Document – Final Report. Report for Collaborative Research Programme. Steering Group. DEFRA. October 2006

Environment Agency (2004), Water Framework Directive Characterisation Atlas – Pressures and Impacts Assessment Review, available at www.environment-agency.gov.uk.

European Environment Agency (2006), How much Bioenergy can Europe Produce Without Harming the Environment?, EEA Report No 7/2006, European Environment Agency, Copenhagen, Denmark.

F

Falconer, K. and Saunders, C. (2002). Transaction costs for SSSIs and policy design. Land Use Policy 19(2), 157-166

Freeman, A. M. (2003): The Measurement of Environmental and Resource Values: Theory and Methods, 2. Edition, Resources for the Future, Washington D.C.

G

Gamboa, G and Munda G. (2007). The problem of wind-farm location: A social multi-criteria evaluation framework. Energy Policy 35(3), 1564-1583

Gamper, C.D., Thöni, M. And Weck-Hannemann, H. (2006). A conceptual approach to the use of cost benefit and multi criteria analysis in natural hazard management. Natural Hazard and Earth System Sciences 6, 293-302

Geschäftsstelle Bundesprogramm Ökologischer Landbau, Bundesanstalt für Landwirtschaft und Ernährung (BLE), Bonn.

Goffe, Ph. Le (1995). The Benefits of Improvements in Coastal Water Quality: A Contingent Approach. Journal of Environmental Management 45: 305-317.

H

Hall, C., McVittie, A. and Moran, D. (2004). What does the public want from agriculture and the countryside? A review of evidence and methods. Journal of Rural Studies 20:211-225

Hasler, B. & J.S. Schou (2004): Samfundsøkonomisk analyse af sikringen af naturvenlig drift på paragraft 3-arealer og naturskovsarealer. DMU

Hasler, B., C.K. Damgaard, E.H. Erichsen, J.J. Jørgensen & H.E. Kristoffersen (2002): De rekreative værdier af skov, sø og naturgenopretning – værdisætning af naturgoder med husprismetoden. AKF

Hiscock, K.M., Lovett, A.A., Saich, A., Dockerty, T.L., Johnson, P.A., Sandhu, C., Sunnenberg, G., Appleton, K.J., Harris, B. and Greaves, J. (2007), Modelling land-use scenarios to reduce groundwater nitrate pollution – the European Water4all project", Quarterly Journal of Engineering Geology, 40:417-435.

I

Interwies, M.A., Kraemer, R.A., Kranz., N., Görlach, B., Dworak, T., Borchardt, D., Richter, S., Willecke, J. (2003). Basic principles for selecting the most cost-effective combination of measures for inclusion in the programme of measures as described in article 11 of the Water Framework Directive. Berlin: Ecologic.

J

Johnes, P.J. (1996), Evaluation and management of the impact of land use change on the nitrogen and phosphate load delivered to surface waters: the export coefficient modelling approach, Journal of Hydrology 183: 323-349.

Johnson, P.A., Shepherd, M.A., Hatley, D.J. and Smith, P.N. (2002) The effects of crop husbandry and nitrogen fertiliser on nitrate leaching from a shallow limestone soil growing a five course combinable crop rotation, Soil Use and Management 18: 68-76.

K

Kratochvil, R., Kaliski, O., Kirner, L. and Freyer, B. (2003): Betriebs- und volkswirtschaftliche Effekte einer großflächigen Bewirtschaftung nach den Prinzipien des Ökologischen Landbaus – dargestellt am Beispiel der Milchproduktion in der Region Mostviertel-Eisenwurzen (NÖ). Ber. ü. Ldw. 81 (2), 223-253.

L

Latacz-Lohmann U. and Renwick, A. (2002). An economic evaluation of the Organic Farming Scheme. In Powell et al. (eds): UK Organic Research 2002: Proceedings of the COR Conference, 26-28 March 2002, Aberystwyth. pp. 311-312

Loomis, J. and Helfand. G (2001) Environmental Policy Analysis for Decision making, Kluwer Dirdrecht, The Netherlands

Lord, E.I., Johnson, P.A. and Archer, J.R (1999) Nitrate Sensitive Areas: a study of large scale control of nitrate loss in England, Soil Use and Management 15: 201-207

Lovett, A.A., Hiscock, K.M., Dockerty, T.L., Saich, A., Sandhu, C., Johnson, P.A., Sandhu, C., Sunnenberg, G. and Appleton, K.J. (2006), Assessing Land-use Scenarios to Improve Groundwater Quality: A Slea Catchment Study, Environment Agency Science Report SC030126/SR, Environment Agency, Bristol.

M

Meyerhoff, J. (2002). Der Nutzen aus einem verbesserten Schutz biologischer Vielfalt in den Elbauen: Ergebnisse einer Kontingenten Bewertung. In Dehnhardt, A. and J. Meyerhoff, (eds.) Nachhaltige Entwicklung der Stromlandschaft Elbe. Wissenschaftsverlag Vauk Kiel KG, Kiel

Meyerhoff, J., Hartje, V., Zerbe, S.(2006). Biologische Vielfalt und deren Bewertung am Beispiel des ökologischen Waldumbaus in den Regionen Solling und Lüneburger Heide. Forschungszentrum Waldökosysteme der Universität Göttingen. Reihe B, Heft 73, Göttingen

Miljøstyrelsen (2006): Samfundsøkonomiske analyser i forbindelse med klimatilpasninger. Miljøprojekt nr. 1121. NIRAS.

Miljøstyrelsen (2005):Samfundsøkonomisk analyse af naturgenopretnings- og kultursikringsprojekt af den østre del af Åmosen. Miljøstyrelsen

Moons, E. (2002). Cost-benefit analysis of the location of new forest land. Working Paper Series, No 2002-05, Faculty of Economics and applied economic sciences, Center for Economic Studies. Energy, Transport and Environment. Katholike Universiteit Leuven

Møller, F., S.P. Andersen, P. Grau, H. Huusom, T. Madsen, J. Nielsen & L. Strandmark (2000): Samfundsøkonomisk vurdering af miljøprojekter. Miljø- og Energiministeriet.

N

Nix, J. (2006) Farm Management Pocketbook, 36th Edition (2006), The Andersons Centre, Melton Mowbray

Nix, J. (2007) Farm Management Pocketbook, 37th Edition (2007), The Andersons Centre, Melton Mowbray

O

Olschewski, R. (1997). Nutzen-Kosten-Analyse des Wasserschutzes durch eine Aufforstung. Schriften zur Forstökonomie. Frankfurt am Main, Sauerländer.

Oppermann, R. (2004). Naturschutzleistungen des Ökolandbaus und Perspektiven der Entwicklung. In Kärcher, A., Reiter, K. and Wiersbinski, N. (eds.) Ökologischer Landbau – Quo Vadis? Zwischen Ideologie und Markt. BfN-Skripten 105. Bundesamt für Naturschutz, Bonn, pp33-39

O'Riordan, T. and Cobb, D. (2001). Assessing the consequences of converting to organic agriculture. Journal of Agricultural Economics, 52 (1) pp. 22-35

Ott, W. and Baur, M. (2005). Der monetäre Erholdungswert des Waldes. Umweltmaterialien – Nr. 193. Bundesamt für Umwelt, Wald und Landschaft. Bern, Switzerland

P

Pretty, J.N., Brett, C., Gee, D., Hine, R.E., Mason, C.F., Morison, J.I.L., Raven, H., Rayment, M.D. and van der Bijl, G. (2000). An assessment of the total external costs of UK agriculture. Agricultural Systems, Vol. 65(2), pp. 113-136

R

Reiter, K. (2004). Auswirkungen konventioneller und ökologischer Landbewirtschaftung auf die Biodiversität – Literaturzusammenstellung aus Vergleichsuntersuchungen. In Kärcher, A., Reiter, K. and Wiersbinski, N. (eds.) Ökologischer Landbau – Quo Vadis? Zwischen Ideologie und Markt. BfN-Skripten 105. Bundesamt für Naturschutz, Bonn, pp7-22

Rorstad, E., Vatn, A. And Kvakkestad, V. (2007). Why do transaction costs of agricultural policies vary? Agricultural Economics 36(1), 1-11

Rüffer, C. (2006). Politisch geforderte Einbindung von Bevölkerungspräferenzen in die Honorierung ökologischer Leistungen der Landwirtschaft. Forschungs- und Studienzentrum Landwirtschaft und Umwelt (ZLU); Georg-August-Universität Göttingen. http://zlu.agrar.uni-goettingen.de/download/

S

Schou J.S. (2003): Miljøøkonomisk analyse af skovrejsning og braklægning som strategier til drikkevands-beskyttelse. DMU

Schou, J.S. & J. Abildtrup (2005): Jordrentetab ved arealekstensivering i landbruget. DMU

Scottish Agricultural College and University of Cambridge (2005), Farm Level Economic Impacts of Energy crop Production, Report to Defra, London

Shephard, M., Pearce, B., Cormack, B., Philipps, L., Cuttle, S., Bhogal, A., Costigan, P. and Unwin, R. (2003). An Assessment of the Environmental Impacts of Organic Farming. A review for Defra-funded project OF0405. http://orgprints.org/

Silgram, M., Williams, A., Waring, R., Neumann, I., Hughes, A., Mansour, M. and Besien, T. (2005) Effectiveness of the Nitrate Sensitive Areas Scheme in reducing groundwater concentrations in England, Quarterly Journal of Engineering Geology 38: 117-127.

Soil Association (2000). The Biodiversity Benefits of Organic Farming. www.soilassociation.org/

Stolze, M., Piorr, A., Häring, A. and Dabbert, S. (2000). The Environmental Impacts of Organic Farming in Europe. Organic Farming in Europe: Economics and Policy, Volume 6, Stuttgart-Hohenheim

Söderqvist, T. and Scharin, H. (2000). The regional willingness to pay for a reduced eutrophication in the Stockholm archipelago. Beijer International Institute of Ecological Economics. Beijer Discussion Papers Series No.128, Stockholm.

T

The National Environmental Research Institute (2005): Miljøøkonomisk analyse af skovrejsning og braklægning som strategier til drikkevandsbeskyttelse.

Thoroe, C., Elsasser, P. and Dieter, M. (2005). Economic accounting of carbon sequestration by forests. The International Forestry Review, Vol. 7(5) p. 299

Tol, R.S.J (2005), The marginal damage costs of carbon dioxide emissions: an assessment of the uncertainties, Energy Policy 33: 2064-2074

Turner, R. K. et al. (1999). Managing nutrient fluxes and pollution in the Baltic: an interdisciplinary simulation study. Ecological Economics vol.30 issue 2. pp.333-352.

U

UK Technical Advisory Group on the Water Framework Directive (2007), Proposals for a Groundwater Classification System and its Application in Regulation, available at www.wfduk.org.

V

Vatn, A. (2002). Multifunctional agriculture: Some consequences for international trade regimes. European Review of Agricultural Economics 29(3), 309-327

W

Wildlife and Countryside Link (2007), Bioenergy in the UK – Turning Green Promises into Environmental Reality, available at www.wcl.org.uk.

Willis, K.G., Garrod, G., Scarpa, R., Powe, N., Lovett, A., Bateman, I.J., Hanley, N. and Macmillan D.C. (2003), The Social and Environmental Benefits of Forests in Great Britain, Report to the Forestry Commission, Edinburgh.

Other types of literature

Literature regarding Multicriteria analysis can be looked up under www.defra.gov.uk/environment/economics/

www.communities.gov.uk/index.

Examples of MCA in practice relating to water management, including cost scoring and criteria weighting:
www.futurewater.nl
(Lasage 2007)
www.feem.it/NR/
http://alba.jrc.it/
www.nat-hazards-earth-syst-sci.net/

and to agriculture see
http://merlin.lusignan.inra.fr/ITAES/

Authors and contact details

Denmark
Per Grønvald,
Aalborg Municipality
Water Supply Department
(pg-forsyning@aalborg.dk)

Lars Delfs Mortensen,
Aalborg Municipality Nature
and Environment Department
(ldm-teknik@aalborg.dk)

Christina Folmand Knudsen,
North Denmark EU-Office
(czf@aalborg.dk)

Rasmus Bohr,
Copenhagen Energy
(rabo@ke.dk)

Gyrithe Brandt,
Copenhagen Energy
(gybr@ke.dk)

Dorthe Bülow,
Copenhagen Energy
(doby@ke.dk)

Christian Seidelin,
NIRAS konsulenterne
(css@niraskon.dk)

Trine Nielsen,
NIRAS konsulenterne
(trn@niraskon.dk)

Germany
Dr. Christina Aue,
Oldenburgisch-Ostfriesischer
Wasserverband
(aue@oowv.de)

Onno Seitz,
Landwirtschaftskammer Weser-Ems
(onno.seitz@lwk-niedersachsen.de)

Sweden
Michael Dahlman,
C4 Teknik, Kristianstad Municipality
(michael.dahlman@kristianstad.se

The Netherlands
Auke Kooistra,
Province Drenthe
(a.kooistra@drenthe.nl)

Nico van der Moot,
Water Supply Company Drenthe
(nico.van.der.moot@wmd.nl)

Ben L.J. van Os,
Province Drenthe (b.os@drenthe.nl)

United Kingdom
Andrew Lovett,
University of East Anglia
(a.lovett@uea.ac.uk)

Kevin Hiscock,
University of East Anglia
(k.hiscock@uea.ac.uk)

Ian Bateman,
University of East Anglia
(i.bateman@uea.ac.uk)

Gilla Sunnenberg,
University of East Anglia, Norwich
(g.sunnenberg@uea.ac.uk)

Helen Jackson,
eftec, London
(eftec@eftec.co.uk)

Paddy Johnson,
Independendent consultant, Sleaford
(pjasoc@soil.freeserve.co.uk)

Sean Burke,
Environment Agency, Birmingham
(sean.burke@environment-agency.gov.uk)

Project web site: www.watercost.org

Layout and print by: Stormtryk
Coordinated by: North Denmark EU-office

1st Issue January 2008

The publication has been supported by the European Union, The Interreg III B North Sea Programme. It should be noted that The contents of this publication reflect the views of the authors. The European Commission is not responsible for any use that may be made thereof.

The WaterCost project
is funded by the
EU Interreg IIIB
North Sea Programme