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# Cost-effectiveness analysis for the implementation of the EU Water Framework Directive

Donné van Engelen[a], Christian Seidelin[b], Rob van der Veeren[c], David N. Barton[d] and Kabir Queb[e]

[a]*Corresponding author. Cartesius Institute, University of Twente, Druifstreek 72, 8911, LH Leeuwarden, The Netherlands. Tel.: +31 30 2348243. E-mail: dmvengelen@hotmail.com*

[b]*NIRAS, Sortemosevej 2, DK-3450, Allerød, Denmark*

[c]*Institute for Inland Water Management and Waste Water Treatment (RIZA), PO Box 17, 8200, AA Lelystad, The Netherlands*

[d]*Norwegian Institute for Water Research (NIVA), Gaustadalléen 21, NO-0349, Kjelsaas, N-0411, Oslo, Norway*

[e]*Skagerrak & Kattegat Water District Authority, County Administration Board of Västra Götaland, Södra Hamngatan 3, 403 40, Göteborg, Sweden*

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**Abstract**

The EU Water Framework Directive (WFD) prescribes cost-effectiveness analysis (CEA) as an economic tool for the minimisation of costs when formulating programmes of measures to be implemented in the European river basins by the year 2009. The WFD does not specify, however, *which* approach to CEA has to be taken by the EU member states. In this paper the lack of a standardised approach to CEA for the implementation of the WFD is taken as the point of departure. The aim of the paper is to discuss and evaluate two pragmatic approaches to CEA based on case studies recently performed in The Netherlands and Denmark. The case studies allow for the comparison of a quantitative and a qualitative approach to CEA at the water body and river basin level and for an evaluation of the approaches in terms of their practical applicability, their transparency and the extent to which they render sound results for decision-making. Conclusions are drawn with regard to the suitability of the two approaches for the implementation of the EU WFD.

*Keywords:* Cost-effectiveness analysis; European Water Framework Directive; Integrated river basin management; Investment decisions

## 1. Introduction

In December 2000 the European Water Framework Directive (WFD) came into act with the predominant goal to improve the functioning of aquatic ecosystems in Europe. In broad lines the WFD

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requires EU member states to characterise all water bodies in the European river basins, to assess whether these water bodies are at risk of falling short of desired ecological objectives and to formulate and implement programmes of measures to attain these objectives. With regard to the latter, cost-effectiveness analysis (CEA), the central focus of this paper, has been assigned an important role in the WFD: it is to be used to select combinations of measures at the water body and at the river basin level that allow for the attainment of the desired ecological objectives at the lowest costs to society (WATECO, 2002). In the scheme of the WFD these least cost programmes of measures will then serve as input for the debate on the disproportionality of the welfare costs of improving the aquatic ecosystems in Europe, a decision-making process that should ideally involve all relevant stakeholders and that can result in derogations in terms of extended deadlines and less ambitious ecological objectives. The decision-making process for the EU WFD relating to the costs and benefits to society of water quality improvements merits an article of its own and is considered beyond the scope of this paper.

Before CEA can be carried out, the EU WFD requires a clear problem definition following on from a so-called risk analysis. As Figure 1 illustrates, the risk analysis starts with a description of the present economic and environmental situation and expectations about impacts of present policies and autonomous trends, resulting in an expected ecological status of water bodies in the future if no additional measures are taken (for the EU WFD 2015 is used as the first-time horizon). By comparing this expected future water status—also referred to as the without scenario—to the objectives that have been set for those water bodies (following from the ecologic characterisation of the water bodies carried

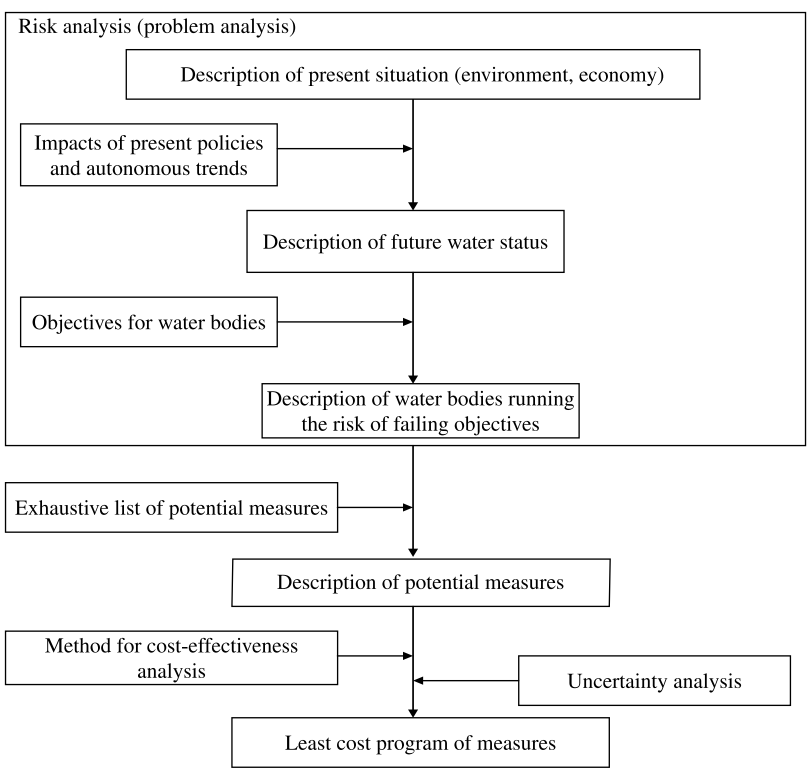


Fig. 1. Generic illustration of the WFD decision problem for CEA.

out during the description of the environmental situation), it becomes clear which water bodies run the risk of failing the objectives. For those water bodies that are at risk, additional measures will have to be considered to solve ecological problems.

Once the risk analysis has been carried out and the problem definition is clear the first step in a CEA is to compile an exhaustive list of potential measures. In this paper measures are defined as technical interventions that result in a physical contribution towards the achievement of an objective. Examples are investments in wastewater treatment plants and reductions in emissions from the agricultural sector. Measures should not be confused with policy instruments, defined in this paper as activities aimed at implementing measures (permits and price incentives, for instance).

The next step in the decision-making procedure is to limit the exhaustive list of measures down to a list of potential measures that focus on the problem at hand and that can be implemented by the contributors to the ecological problems identified. The risk analysis—that not only indicates the type of problems that need to be solved (too high concentrations of certain substances, or not enough species diversity), but also the potential causes and causers—can be used for this purpose. The resulting list of potential measures relevant for the specific problems should include an indication of the costs and effects of the measures.

The third step in the procedure is to combine the potential measures that have been identified into programmes of measures that allow for the achievement of the predetermined objectives for water bodies at the lowest costs to society. This step involves the choice for, and the application of, a specific CEA approach and is the focus of this paper. The final step for the formulation of programmes of measures shown in Figure 1 is the evaluation of uncertainty. This step is necessary to test the sensitivity of the formulated programmes of measures to decisions and assumptions made regarding the assessment of the costs and effects of measures.

Although the WFD requires CEA along the general lines described above, it does not prescribe a standardised methodology to be applied by the EU member states. This leaves room for the choice between a number of alternative approaches and hence raises the question: which approach or approaches are most appropriate? In our opinion a CEA approach has to meet three core requirements if it is going to be of use for decision-making for the implementation of the WFD. First, it has to be transparent so that the analysis and outcomes can be understood and used by decision-makers. Second, it has to be pragmatic so that it can be carried out by non-economists given time and resource constraints such as a large number of water bodies and a small number of economists involved in water management in many EU countries. Finally, it has to render sound enough results to base decisions on and should thus approach optimal solutions that are subject to acceptable margins of uncertainty.

In this paper we set out to compare two pragmatic approaches to CEA and to evaluate their suitability for the implementation of the EU WFD in terms of the three core requirements formulated above. Based on case studies recently performed in Denmark and The Netherlands Section 2 and 3, respectively, present a quantitative and a qualitative approach to CEA at the water body level. In Section 4 the possibilities for extending the two approaches to CEA at the river basin level are discussed and in Section 5 the two approaches are evaluated and compared and conclusions are drawn with regard to the suitability of the two approaches for the implementation of the EU WFD. Since we consider pragmatism and transparency important criteria for CEA for the WFD, extensive and complicated modelling approaches for CEA at the river basin level are only briefly mentioned in this paper.

## 2. Quantitative approach to CEA

The first approach to CEA presented in this paper is a quantitative approach that is oriented towards emission reductions and that prioritises and combines measures using the ratio of the costs and effects of individual measures. It is a pragmatic approach that is common both to water management and the management of other environmental resources (Krozer, 2002; WATECO, 2002; Gerasidi *et al.*, 2003; RIZA, 2003; Interwies *et al.*, 2005). In this quantitative approach, once a list of potential measures has been drawn up to solve an environmental problem (compare to Figure 1), the cost-effectiveness of each measure is derived by dividing the costs by the effects of the individual measures. The individual measures are then ranked to arrive at marginal cost curves, allowing for the selection of the least costly combination of measures to achieve the environmental objectives.

Table 1 presents such a ranking of potential measures drawn up by Danish experts to improve the water quality of the Mariager Fiord in Denmark (NIRAS, 2004). The fiord—located in an agriculturally intensive area on the border of the counties of Nordjylland and Århus—is eutrophic and in the past it has known incidents of widespread hypoxia leading to mass fish starvation. Nutrient loading has been identified as the major cause of the ecological problems for the water body and it has been estimated that the annual abatement targets for nitrogen and phosphorus emissions to the fiord should be 200 tons and 2 tons, respectively.

The cost-effectiveness of the measures presented in Table 1 is calculated by dividing the annual costs of a measure by the annual effect in terms of nitrogen abatement. This effect is expressed as the number

Table 1. Measures for cost-effective nitrogen (N) reduction (NIRAS, 2004).

| Measure | Cost-effectiveness (Kr/ton N) | Ton N reduction per year | Accumulated N reduction (ton) |
|---|---|---|---|
| Establishing mussel farms | −319.742 | 15.8 | 15.8 |
| Substituting fish farm 3 by model | −38.634 | 6.2 | 22.0 |
| Wetland—Barsbøl | 10.512 | 13.0 | 35.0 |
| Wetland—Fyrkat | 14.756 | 21.3 | 56.3 |
| Wetland—Møgelmose | 27.446 | 12.6 | 68.9 |
| 10% standard emission reduction | 28.359 | 151.0 | 219.9 |
| Winter catch crops | 45.783 | 92.0 | 311.9 |
| Land use subsidy 1–200 ha | 59.001 | 3.5 | 315.4 |
| Winter catch crops and spring seed | 67.090 | 316.0 | 631.4 |
| Land use subsidy 4–485 ha | 83.416 | 13.5 | 644.9 |
| Closing fish farm 2 | 101.016 | 19.6 | 664.5 |
| Land use subsidy 3–500 ha | 106.883 | 4.3 | 668.8 |
| Closing fish farm 1 | 115.926 | 6.9 | 675.7 |
| Private afforestation 500 ha | 194.415 | 15.0 | 690.7 |
| Public afforestation 265 ha | 237.785 | 6.5 | 697.2 |
| Treatment plant 2 | 253.791 | 10.8 | 708.0 |
| Overgaard's dams | 255.170 | 8.0 | 716.0 |
| Reduction in domestic animals | 365.125 | 24.5 | 740.5 |
| Rainwater related discharges 4 | 393.026 | 0.2 | 740.6 |
| Rainwater related discharges 3 | 417.762 | 0.1 | 740.7 |
| Rainwater related discharges 1 | 1,194.798 | 2.5 | 743.2 |
| Rainwater related discharges 2 | 1,357.725 | 1.8 | 745.0 |

of tons of nitrogen emission reduction per year and is quantified using models and expert judgement. The costs of the individual measures are defined as the annual costs directly related to the implementation of a measure, consisting of the annualised initial investment[1], the net annual operational costs, the net maintenance costs and the net annual productivity loss/gain[2]. Since the CEA for the Mariager Fiord is a so-called partial analysis, indirect costs incurred by sectors other than the sector implementing a certain measure are not considered here. They will, however, be considered in Section 3.

With regard to the pricing of the cost items a welfare economic approach is taken in the Danish CEA study. To apply such an approach in the case of distorted markets all factor prices (prices excluding taxes and subsidies) should be converted into shadow prices by multiplying by conversion factors. This will allow prices to reflect the relative scarcities in an economy. In a Danish context, where markets are generally not distorted, this can be done relatively easily by using two conversion factors: the net tax factor and the net tax factor for internationally traded goods. The net tax factor reflects the average tax pressure in Denmark (17%), whereas the net tax factor for internationally traded goods reflects the relation between the inland price level on internationally traded goods and their world market price levels. A third conversion factor that is being introduced in Danish guidelines to economic analyses is the tax loss factor. This conversion factor reflects the efficiency loss which results from tax financing of, for example, public investments. Applying these conversion factors in a CEA provides the theoretical link to welfare economic theory by rendering prices on which consumers base their decisions in accordance with their willingness to pay (leading to utility maximisation)[3].

The selection of the most cost-effective combination of measures to achieve the abatement target for the Mariager Fiord is based on the ranking of the measures according to their individual cost-effectiveness. As can be deduced from Table 1 the two most cost-effective measures identified in the CEA for the Mariager Fiord are characterised by a negative cost-effectiveness, meaning that money is generated by carrying out these measures[4]. The accumulated nitrogen abatement column of the table shows that the nitrogen abatement target of 200 tons per year for the fiord is reached by carrying out the top six measures in the table. This requires the establishment of a mussel farm, the conversion of three areas of farmland to wetlands, the substitution of four fish farms by a model fish farm and a 10% standard

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[1] As is common in investment analysis the initial investments are annualised in order to make measures with different economic lifetimes comparable. Using a discount rate $r$ an initial investment with a lifetime of $n$ years is annualised by dividing by a factor $((1-(1+r)^{-n})/r)$. In Denmark a discount rate of 3% is used for environmental investments. In the Netherlands a discount rate of 4% is used.

[2] To give an example: the conversion of farmland to a wetland necessitates an initial investment with an infinite economic lifetime. Due to the conversion, income from the land is lost (including EU-subsidies), but new subsidies are provided reducing the net income loss. With regard to net operation costs the same with–without approach has to be taken. Maintenance of the wetland requires annual costs to be made. However, costs are no longer incurred for working the land. Total costs of the measure are the sum of the annualised initial investment, the net operating costs and the net income loss/gain.

[3] Note that the use of a net tax factor across all measures alone is not expected to change the relative ranking (cost-effectiveness) of the different measures. However, the correct estimation of social costs of measures becomes material when evaluating disproportionate costs in a cost–benefit analysis.

[4] This contradicts the basic assumption in economic theory of profit maximisation by firms. According to this assumption, in the absence of, for example, nutrient abatement policies, profit-maximising firms will choose those levels of production and emissions that will maximise profits. This automatically implies that each measure to reduce or increase nutrient emissions reduces profits because, if these measures would be beneficial, these measures would have already been taken in the initial situation (before nutrient abatement restrictions apply). Therefore, measures with negative costs indicate sub-optimal production processes in the initial situation. This is often caused by incomplete information.

reduction of nitrogen emission in the area from farmland and pig farms. From the Danish CEA study it also appears that, by establishing the mussel farm, the phosphorus abatement target of 2 tons per year for the fiord can be reached. This implies that the six measures described above together form the least costly programme of measures to attain the nutrient abatement targets for the Mariager Fiord (NIRAS, 2004).

## 3. Qualitative approach to CEA

As opposed to the quantitative CEA approach discussed above, this section presents a CEA approach that is qualitative in nature. The qualitative CEA approach uses scorecards and an expert panel of aquatic biologists and ecologists to formulate a cost-effective programme of measures directed at the ecosystem of a water body. The methodology has recently been developed in Germany for the implementation of the EU WFD (Interwies *et al.*, 2003) and consists of six steps, starting with the systematic derivation of a list of potential measures based on causes and causers of the water problems (compare to Section 1). This section will illustrate the qualitative German CEA approach at the water body level using a cost-effectiveness analysis performed for the highly eutrophic Lake Leijen in the Dutch province of Friesland as a case study (van Engelen, 2005).

The starting point for the CEA for Lake Leijen is a list of 33 potential measures drawn up by local Dutch experts to address the ecological problems of the lake. The first step to arrive at a cost-effective programme of measures is then to identify the most effective measures from the long list of 33 measures. This is done using a scorecard called a cause-effect matrix, which is initially completed by each expert on the expert panel individually. For each measure each expert has to decide what the effect of the measure will be on the four most important ecological parameters for a lake (given by the WFD): the abundance, composition and quality of macrophytes, algae, benthic invertebrate fauna and fish fauna. The effect on each parameter can be scored as no effect (— or 0), a low effect ( × or 1), a medium effect ( × × or 2) or a high effect ( × × × or 3). The scores for the effects per measure on the individual parameters are then added up giving a total score for the measure ranging from 0 to 12 for each expert. Finally the individual scores of the experts are added up in order to arrive at a total score for the effectiveness of each measure, in the case of the CEA for Lake Leijen—with an expert panel of four persons—ranging from 0 to 48.

In Table A1 in the appendix an overview of the 33 measures, their total scores and their lowest and highest scores is given. In the last column of the table the measures are classified according to their priority for water quality improvements for the lake. The measures with total scores between 0 and 1 and between 2 and 17 are, respectively, assigned priority level 0 (no ecological effect) and 1 (low ecological effect) and are considered of too little impact to consider for the next step in the methodology. Measures with total scores between 18 and 33 are assigned priority level 2 (medium ecological effect) and are presented in italic in Table A1. For the CEA for Lake Leijen no priority 3 measures are identified (total score between 34 and 48—large ecological effect), so the 11 priority 2 measures are considered the most effective measures for the lake and are used in the next step of the methodology to identify the most effective combinations of measures to improve water quality. This is done using a scorecard called a combination matrix shown in Table 2.

Table 2 presents a systematical comparison of the effectiveness of pairs of measures and serves to identify the most effective combinations of measures suitable for formulating potential programmes of measures in the next step of the methodology. Similar to the cause—effect matrix, the combination

Table 2. Combination matrix Lake Leijen for four experts (van Engelen, 2005).

| Measure | 1.1 | 1.2 | 2.1 | 2.2 | 2.4a | 2.7 | 4.1 | 5.3 | 5.5 | 6.3 | 6.7 |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Renovation/expansion WWTP (1.1) | | 5 | **9** | 8 | 7 | 5 | 8 | 7 | **10** | **10** | 5 |
| Diverting boezem water (1.2) | | | 7 | 6 | 7 | 3 | 7 | 6 | 8 | 7 | 5 |
| Requirement oriented fertilisation (2.1) | | | | 5 | 4 | 7 | 7 | 6 | 7 | 6 | 6 |
| Land appropriation and extensivation (2.2) | | | | | 5 | **9** | 8 | 6 | 7 | 7 | 5 |
| Dredging bottom of Lake Leijen (2.4a) | | | | | | **9** | 6 | 5 | 4 | 5 | 8 |
| Wetlands boezem water (2.7) | | | | | | | 7 | 7 | 8 | 7 | 6 |
| Natural waterlevel Frisian boezem (4.1) | | | | | | | | 7 | 7 | 7 | 8 |
| Extension of banks into Lake Leijen (5.3) | | | | | | | | | 7 | 8 | 6 |
| Sediment trap (5.5) | | | | | | | | | | **10** | **9** |
| Elimination of bream (6.3) | | | | | | | | | | | 7 |
| Introduction submerged waterplants (6.7) | | | | | | | | | | | |

matrix is first completed by each expert individually after which the scores are added up to arrive at a total for each pair of measures. The experts can score a combination as having a minimal effect (+ or 1), a good effect (++ or 2) or a very good effect (+++ or 3), meaning that the total score for a pair of measures can range from 4 to 12. For the CEA for Lake Leijen it was decided that those pairs of measures that are scored as having a very good effect at least once (range from 9 to 12) would be selected for the formulation of potential programmes of measures. In Table 2 these pairs of measures have been highlighted and in Table 3 the five programmes of measures resulting from the seven most effective combinations of measures are presented. Note that programme A is a combination of the three most effective pairs (scoring 10 points each). As can be deduced from Table 3 the most effective pairs of measures in Table 2 serve as the base for the potential programmes of measures to improve the water quality of Lake Leijen. These measures are complemented by the experts by additional measures in order to ensure that the programmes address all major pressures on the water body and thus have the potential to attain the desired ecological objectives.

Before being able to decide which programme of measures in Table 3 is the most cost-effective two more tasks have to be carried out. First the effectiveness of each potential programme of measures has to be expressed in terms of the ecological effectiveness of the programme, the timescale within which the programme will have effect and the probability that the target for the lake will be attained within six years (required by the WFD)[5]. Then the costs of the programmes have to be estimated in terms of their direct and indirect costs. Direct costs refer to initial investment costs and the costs of operation and maintenance and can be expressed as total project costs or annualised costs. Since annual operation and maintenance costs were negligible for the measures, and all measures have a long economic lifetime, it was decided to use the sum of the initial investments in the measures as total direct project costs. Regarding the indirect costs of the programmes of measures—referred to as costs incurred by other sectors than the sectors implementing the measures—in this study the experts gave a qualitative estimate of their potential size (low, moderate, high or very high). Table 4 gives an overview of the characterisation of the five potential programmes of measures for Lake Leijen in terms of their effectiveness and costs.

[5] Target attainment: improbable–probable–highly probable; ecological effectiveness: low–moderate–good–very good; timescale: long term (LT >6 yr)–medium term (MT: ~6 yr)–short term (ST: <6 yr).

Table 3. Programmes of measures for Lake Leijen (van Engelen, 2005).

| Programme | Main combination(s) | Additional measures |
|---|---|---|
| A | WWTP + sediment trap + elimination of bream | Extension of banks |
| B | WWTP + requirement oriented fertilisation | Elimination of bream + submerged water-plants |
| C | Land appropriation + wetlands boezem water | Wetlands polder water |
| D | Dredging + wetlands boezem water | Elimination of bream + extension of banks + wetlands polder water |
| E | Sediment trap + submerged waterplants | Elimination of bream |

Table 4. Effectiveness and costs of programmes of measures for Lake Leijen (van Engelen, 2005).

| Progaramme of measures | Target attainment by 2006 | Ecological effectiveness | Timescale | Direct costs in Euro | Indirect costs |
|---|---|---|---|---|---|
| Programme A | Highly probable | Very good | MT | 16.6–16.7 M | Low |
| Programme B | Probable | Moderate/good | MT/LT | 15.8 M | Low |
| Programme C | Improbable | Moderate | LT | Relatively very high | Moderate |
| Programme D | Highly probable | Good | ST/MT | 4.0–5.1 M | Low |
| Programme E | Improbable | Moderate | ST | 0.2 M | Low |

The final step in the qualitative approach to CEA consists of the selection of the most cost-effective programme of measures to attain the desired ecological objectives. The first selection from Table 4 takes place by eliminating those programmes of measures that will probably not lead to target attainment (programmes C and E), since these programmes would have to be complemented by another programme in order to be certain to reach the goals of the WFD. Programme C is the least attractive programme in the table since it will probably not lead to target attainment and is the most expensive of all. In terms of costs programme E is very attractive, but unfortunately it is unlikely to be effective enough. It would be very attractive as an experiment given a longer time frame than applicable under the WFD. This leaves us with programmes A, B and D. Programme A is more attractive than programme B, because both programmes are comparably costly but programme A has a higher likeliness of attaining the targets for Lake Leijen on time and with a better ecological effectiveness. However, the most cost-effective programme of measures in the table is programme D, since it has the same chance of attaining the targets for the WFD as programme A but only necessitates a quarter to a third of the investment in programme A. Hence, at the level of the water body, the CEA for Lake Leijen shows that a combination of dredging, elimination of bream, extension of the banks of the lake and wetlands for polder and boezem water forms the most cost-effective programme of measures to reach the WFD water quality standards on time.

## 4. Extension of the two approaches to CEA at the river basin level

In the previous sections case studies performed in Denmark and The Netherlands have been used to respectively illustrate a quantitative and a qualitative approach to CEA at the water body level. One of the major challenges of the EU WFD is, however, to arrive at integrated management plans at the level of entire river basins. The goal of CEA in such a case is to identify in which regions or countries measures ought to be taken in order to arrive at lower total costs for achieving ecological objectives than reached by applying flat rate emission reductions in all the regions or countries of the river basin simultaneously. In this section the possibilites for extending the two pragmatic CEA approaches previously discussed are explored in order to gain insight into the applicability of the two approaches at the river basin level. Although there are other options to perform CEA at the river basin level using more extensive modelling approaches—possibilities are linear programming models (Gren *et al.*, 1997) and linear quadratic models (van der Veeren, 2002)—we consider these approaches too unpragmatic in terms of expertise and information requirements and too untransparent when it comes to the derivation of programmes of measures to be considered suitable for the implementation of the WFD. It is for this reason that we do not elaborate on these approaches in this paper and focus on an extension of the pragmatic CEA approaches to the river basin level.

The quantitative CEA for the Mariager Fiord is an example of a CEA for two or more sources in one region. For larger river basins that cross administrative borders—a common situation in Europe—an extension of the quantitative CEA approach to the river basin level implies considering the effects on a target area (lake, river, the sea) of two or more sources in two or more regions or countries. This complicates matters because differences between emission reductions at regional sources and effects in target areas (due, for instance, to soil retention and biochemical processes) will have a larger impact on the outcomes of a CEA. This can be illustrated by an example. Assume two wastewater treatment plants (WWTPs) in two different regions (regions A and B) are located along a river leading to a eutrophic lake (see Figure 2). The WWTPs discharge the same substances but have different abatement costs: 100 Euro/kg for WWTP A and 60 Euro/kg for WWTP B. If the impact of one kilogram reduction at WWTP A on the loads to the lake is equal to the impact of one kilogram emission reduction at the WWTP in region B, the most cost-effective solution to abate substances would be to have region B decrease emissions more than region A.

Now consider the case where, due to biochemical processes, there is a difference between emission reductions and nutrient loading to the lake between WWTP A and WWTP B. In order to arrive at a one kilogram reduction in loads to the lake, two kilograms emission reduction by WWTP A and four kilograms emission reduction by WWTP B is required. This would mean that the costs to reduce the loads to the lake by one kilogram would be 200 Euro for WWTP A and 240 Euro for WWTP B. In that case it would be more cost-effective at the level of the river basin to reduce emissions at WWTP A, even though the costs of emission reduction per kilogram at source level are lower at WWTP B.

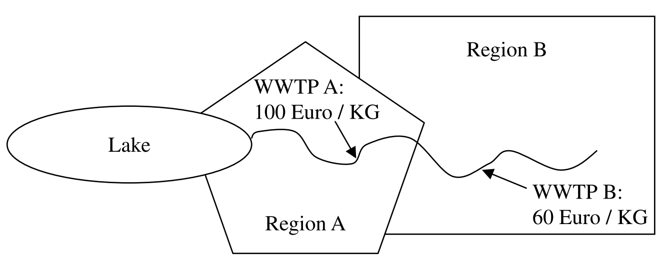


Fig. 2. Illustration of a CEA for two sources in two regions.

In order to systematically account for the effects of soil retention and biochemical processes in a quantitative CEA at the river basin level, transport coefficients have to be calculated for each measure oriented towards emission reduction. These transport coefficients are linear representations of transport mechanisms used in water quality models and describe the percentage of a particular substance emitted by an individual source in a specific region reaching a target area. For an extension of the quantitative CEA approach presented in Section 2 to a pragmatic quantitative CEA approach at the river basin level the calculations for the prioritisation of measures have to incorporate these transport coefficients in a manner similar to the example given above. The pragmatic approach to CEA developed in The Netherlands for the implementation of the WFD shows how local, regional and international CEA analyses based on the ratio of the costs and effects of measures can be linked using transport coefficients (RIZA, 2005).

With regard to the extension of the qualitative CEA approach presented in Section 3 a solution comparable to that of the use of transport coefficients has to be sought. In other words, a qualititative CEA approach at the river basin level has to combine separate programmes of measures and be able to compare the efficiency and the costs of the measures considered to be implemented in the river basin. In theory this would require experts to assess the effects of measures on one or more target areas in a river basin, taking into account such issues as soil retention and biochemical processes. Unfortunately, as much as Interwies *et al.* (2003) recognise the importance of integrating qualitative analyses performed for separate water bodies in a river basin, the design of an approach to achieve this goal was beyond the scope of their assignment when developing the qualitative approach to CEA in Germany.

## 5. Evaluation of approaches and conclusions

In this paper a quantitative and a qualitative approach to CEA at the water body level have been discussed, along with the possibilities for extending the approaches to CEA at the river basin level. Based on the experiences following from the Danish and Dutch case studies analysed, in this section the two approaches will be evaluated in terms of their transparency, their practical applicability and the degree to which they render sound results. By using these three criteria (formulated in Section 1) we are able to draw conclusions regarding the suitability of the quantitative and qualitative CEA approaches for the implementation of the EU WFD.

With respect to the quantitative CEA approach the case of the Mariager Fiord shows that the transparency of the selection of the programme of measures is a major advantage of the approach. The use of quantitative information (costs divided by effect) makes the selection of measures easy to understand and justify. This is particularly important for CEA at the river basin level, where the implementation of cost-effective programmes of measures may require compensatory payments between regions or nations. For the negotiations for such payments a transparent argument is essential, both in terms of the prioritisation of measures and the costs of implementing measures.

Regarding the practical applicability of the quantitative CEA approach, a quick and easy execution depends on the kind of measures considered and the models and information available to quantify the costs and effects of these measures. This presents a problem common to many investment studies: most of the work goes into the construction of a reliable data set. Carrying out the CEA calculations is relatively straightforward once reliable data are available. In this respect (inter)national databases with cost and effect data for measures common in a country or river basin will contribute considerably to reducing the time and costs of carrying out a quantitative CEA[6]. Furthermore, the uncertainty introduced by calculating or estimating the costs and effects of measures can be evaluated for quantitative CEA studies by using sensitivity analysis, scenario analysis and probability simulation techniques (see, for instance, Barton *et al.*, 2005; NIRAS, 2004).

A disadvantage of the quantitative CEA approach is the requirement that only those measures can be considered for which both costs and effects can be quantified. This means that measures hard to quantify (usually affecting hydromorphology or the ecosystem of a water body directly) either have to be omitted from the analysis or their effects have to be estimated very roughly. This can result in biases. Other biases in the formulation of programmes of measures can result from the omission of time lags, synergy effects and transport coefficients from the calculations[7]. Although these aspects can be taken into account in a quantitative CEA they are often omitted for practical convenience.

With regard to the ranking of measures biases can also occur if the multiple effects of measures are neglected and a selection is made taking only one of a series of multiple objectives for a water body into account. This problem can, in part, be dealt with by using eutrophication and dispersion equivalents or step-by-step ranking optimisation approaches (see van der Veeren, 1999; van der Veeren & Tol, 2001). A final point to be made regarding the quantitative CEA approach discussed in this paper is that it should not be forgotten that the approach is oriented towards emission reductions whilst the EU WFD aims at attaining ecological objectives. Uncertainty in the relation between emission reductions and ecological parameters introduces yet another bias.

The major advantage of the qualitative CEA approach as applied to the case of Lake Leijen in The Netherlands is the practical applicability of the methodology. Provided that a sufficient number of (local) aquatic biologists and ecologists are available, the methodology can be easily applied and renders results within a day. Understanding and carrying out the approach does not require specialised economists and this makes this approach ideal for the quick and relatively cheap screening of measures to support decision-making for the implementation of the EU WFD. Other advantages of the qualitative approach are the ease with which experts can take into account multiple ecological objectives, time lags, synergy effects and an unlimited range of measures.

The downside to the use of expert judgement for the formulation of programmes of measures relates both to the untransparency of the results derived and the uncertainty surrounding the expert judgements. With regard to the latter, biases result from the fact that experts are limited in their knowledge of measures, the water body and the surrouding area under consideration. On top of this, given the extensiveness of the scorecards and the fairly crude scaling, it is not easy for experts to remain consistent when applying the qualitative CEA approach. In fact, the CEA for Lake Leijen showed that individual judgements of measures can differ considerably between experts, and that outliers frequently occur (see Table A1)[8]. The uncertainty arising from the use of expert judgement can be reduced by increasing

[6] Currently in The Netherlands such a database is being drawn up for the implementation of the EU WFD (see www.paict.com).
[7] Time lags refer to a certain period of time elapsing between the implementation of a measure and the measure taking effect in the target area. Synergy effects refer to positive and negative reinforcements of measures considered for implementation. See Section 4 for an explanation of transport coefficients.
[8] For the CEA for Lake Leijen differences in scoring between experts were also caused by differences in understanding of the measures. When applying the qualitative approach a clear definition of measures for all is an important issue.

the number of experts on a panel and evaluated by using a control panel to carry out the analysis a second time for the purpose of comparison.

The untransparency of the results derived through expert judgement probably forms the most serious restriction for the application of the qualitative CEA approach for the implementation of the EU WFD. Regarding the opacity of the methodology problems can be foreseen when applying for exemptions from Brussels and when negotiating compensation payments between regions within river basins. Results derived using expert panels can be easily questioned and consequently do not form a solid ground for negotiations.

In conclusion, with regard to the suitability for the implementation of the EU WFD of the two CEA approaches discussed in this paper, the evaluation performed in this section strongly suggests that the applicability of the qualitative CEA approach is limited to performing CEA for an individual water body or for small river basins, common for instance in Denmark. Apart from this we conclude that the first step in the qualitative approach (carried out with the cause–effect matrix, see Section 3) will be very useful for screening measures and reducing a long list of potential measures to a list of effective measures. This shortlist can then be used as the point of departure for the quantitative CEA approach, meaning that less time and resources are needed to quantify the costs and effects of the measures considered and more attention can be devoted to the optimisation of the programmes of measures formulated for an entire river basin. Our analysis suggests that a merging of the two approaches along this line will provide a practical and transparent CEA approach that allows for the formulation of sound programmes of measures for the implementation of the EU WFD.

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**Appendix Table A1**. Summary of cause–effect matrices for Lake Leijen (van Engelen, 2005).

| Measure | Lowest individual score | Highest individual score | Total of four experts | Priority |
|---|---|---|---|---|
| *1.1 Renovation/expansion WWTP* | 3 | 8 | 24 | 2 |
| *1.2 Diverting boezem water* | 2 | 8 | 20 | 2 |
| 1.3 Handling of town area | 0 | 0 | 0 | 0 |
| 1.4 Disconnect rainwater & sewerage | 0 | 2 | 2 | 1 |
| 1.5 Stormwater overflows | 0 | 2 | 4 | 1 |
| *2.1 Requirement oriented fertilisation* | 2 | 8 | 19 | 2 |
| *2.2 Landappropriation/extensivation* | 1 | 12 | 18 | 2 |
| 2.3 Riparian buffer strips | | National policy from the year 2000 onwards | | |
| *2.4a Dredging of bottom Lake Leijen* | 2 | 8 | 18 | 2 |
| 2.4b Dredging of waterways | 0 | 2 | 4 | 1 |
| 2.5 P-fixation in bottom the Leijen | 0 | 3 | 5 | 1 |
| 2.6 Wetlands polder water | 0 | 2 | 4 | 1 |
| *2.7 Wetlands boezem water* | 2 | 8 | 19 | 2 |
| 2.8 Diverting polder water | 0 | 8 | 15 | 1 |
| 2.9 Hydrological isolation the Leijen | 0 | 6 | 13 | 1 |
| 2.10 Flushing the Leijen | 0 | 4 | 4 | 1 |
| 2.11 Variable waterlevels polder water | 0 | 2 | 4 | 1 |
| 2.12 Closed watercycle farmstead | 0 | 2 | 2 | 1 |
| 2.13 Stand alone toilets | 0 | 2 | 4 | 1 |
| *4.1 Natural waterlevel Frisian boezem* | 3 | 12 | 25 | 2 |
| 5.1 Waterways and islands | 1 | 1 | 4 | 1 |
| 5.2 Cleaning up illegal landing places | 1 | 3 | 6 | 1 |
| *5.3 Extension of banks into lake* | 1 | 8 | 19 | 2 |
| 5.4 Planting banks new islands | 1 | 7 | 16 | 1 |
| *5.5 Sediment trap* | 3 | 8 | 21 | 2 |
| 6.1 Submerged islands | 0 | 5 | 10 | 1 |
| 6.2 Gallery of wooden posts | 0 | 5 | 8 | 1 |
| *6.3 Elimination of bream* | 5 | 7 | 26 | 2 |
| 6.4 Introduction of pike | 1 | 6 | 12 | 1 |
| 6.5 Inundation for breeding purposes | 1 | 4 | 10 | 1 |
| 6.6 Introduction of clam shells | 1 | 4 | 9 | 1 |
| *6.7 Submerged waterplants* | 0 | 8 | 20 | 2 |
| 6.8 Mowing policy | 0 | 6 | 9 | 1 |