SEVENTH FRAMEWORK PROGRAMME

THEME 6: Environment (including Climate Change)

# Adaptive strategies to Mitigate the Impacts of Climate Change on European Freshwater Ecosystems

Collaborative Project (large-scale integrating project)
Grant Agreement 244121
Duration: February 1st, 2010 – January 31st, 2014

# Deliverable 6.10
# Pan-European review of cost-effectiveness analysis studies relating to water quality and Directive compliance challenges

Lead contractor: **JHI**
Other contractors involved: **UPAT, SYKE**

Due date of deliverable: **Month 30**
Actual submission date: **Month 32**

Work package: 6

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Estimated person months: 4

| Project co-funded by the European Commission within the Seventh Framework Programme (2007-2013)<br>Dissemination Level (add X to PU, PP. RE or CO) | | |
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## Abstract

*REFRESH seeks to develop a framework to help design cost-effective restoration programmes for freshwater ecosystems. In a number of REFRESH case study catchments cost-effectiveness analysis (CEA) of competing remediating strategies will be undertaken utilizing best practice models of CEA undertaken in Europe in relation to water quality and WFD compliance challenges. This Deliverable comprises a review of existing European CEA studies relating to water quality and WFD compliance. First, conceptual and measurements issues related to CEA are highlighted and methodological approaches to CEA and their applications are explored. Subsequently, climate change and uncertainty issues associated with CEA are reviewed and key applications and experiences in CEA studies in the five WP6's case study countries (UK, Norway, Finland, Greece and the Czech Republic) and also in several other European countries are presented.*

*Analysis presented here shows that within the economic analysis required for the delivery of WFD objectives, the role of CEA has been particularly highlighted as a prerequisite for the development of river basin management plans. Regarding the measurement of costs of environmental measures, an important issue is associated with data availability which would enable research efforts to account for not only private financial compliance costs but also for economic/social costs that are usually very difficult to monetize. Key issues, which have to be addressed in measuring effectiveness, include the basis on which effectiveness is assessed (e.g. 'pressures' or 'impacts') and the assignment of proper weights to individual pressures in cases of multi-pressure effects of different measures.*

*Various methodological approaches have been proposed in applying CEA to evaluate environmental measures and policies. The most commonly used approach involves linear optimization models based on mathematical programming framework. In general, the choice of a particular methodological framework in the CEA highly depends on the specific environmental problem to be dealt with, the availability and credibility of data, and the degree of uncertainty inherent in cost and effectiveness information.*

*In CEA, uncertainty can be inherent in the estimates of costs, effectiveness and time-lagged effects of measures and thus, can considerably affect the ranking of measures under consideration. To deal with this issue, the use of intervals of costs and effectiveness estimates as well as sensitivity and scenario analysis is advocated. Also, stochastic programming and Bayesian Belief Networks (BBN) can be applied to investigate water management decision-making under uncertainty. Climate change can create new and/or affect existing pressures on water bodies, and directly and indirectly affect the effectiveness of long-term water management measures. Importantly, the above-mentioned issues justify the "climate-proofing" investigation of proposed water management measures.*

*Finally, this review has shown that several studies have been carried out in Europe dealing with CEA applications related to WFD. In the case of the five countries specific to REFRESH demonstration catchments, one can observe a rich research tradition in the UK (including Scotland), Norway and Finland. This tradition seems much weaker in the case of the Czech Republic where such studies have been introduced in the context of the implementation of the WFD, and especially of Greece, where CEA applications are fragmented and progress in the*

*implementation of the WFD has shown a considerable lag. In the case of other European countries the review has shown a notable number of CEA studies dealing with agricultural abatement measures in Sweden, Germany and France and an impressive CEA study of reducing nitrogen emissions from agriculture in the Danube (i.e. specific to Austria, Bulgaria, Hungary and Romania). In Spain CEA efforts seem to focus more on water-saving measures and deal with both urban and rural activities.*

## 1 Introduction

The European Water Framework Directive (WFD) (EC, 2000) explicitly integrates economics into water management and water policy decision-making in Europe, calling for the application of economic principles (eg. polluter-pays principle (Article 9); economic tools (e.g. cost-effectiveness analysis (Annex III.b)) and economic instruments (e.g. water pricing (Article 9) in order to achieve its environmental objectives (European Commission, 2003). In this context, economic analysis has been given a key role in providing valuable information to aid the decision making process and developing river basin management plans (RBMPs).

Amongst the various economic analyses stated or implied in the WFD, cost-effectiveness analysis (CEA) has been given a paramount emphasis. Annex III (b) of the WFD stipulates that a CEA of water pollution mitigation measures should be conducted as a pre-requisite in formulating programmes of measures: "*The economic analysis shall contain enough information in sufficient details (taking into account the costs associated with collection of the relevant data) in order to make judgements about the most cost-effective combination of measures in respect of water uses to be included in the programme of measures under Article 11 based on estimates of the potential costs of such measures*". CEA required by the WFD entails identification of environmental objectives for each water body, assessment of possible measures to meet the pre-specified water management objectives set out in the Directive, and estimation of their costs and impacts on the status of the water bodies. However, whereas the identification of cost-effective programmes of measures is a requirement in WFD (Article 11), their implementation is not a rigid requirement. The Directive's emphasis on achieving good status of water bodies is accompanied by the need to demonstrate that failure to take relevant action is attributed to reasons of technical feasibility or disproportionately expensive/costly actions (Article 4).

As noted in the project's Description of Work (DoW) document, the CEA of environmental compliance envisaged in the Work Package 6 (WP6) of REFRESH links to the scenarios developed on future changes in climate, land use, nitrogen deposition and water resources to be used specified in the context of WP1 of the project (Strategies, Scenarios and Stakeholders). Also, it draws from the first three Tasks of WP6, which profiled the selected demonstration catchments and identified compliance threats (Task 1), screened catchments to identify sub-catchments which reflect the variety of compliance conditions (Task 2) and utilized local stakeholder consultation to scope out possible mitigation, adaptation and restoration options which might enable compliance (Task 3).

Further to the above, in the context of the effort to analyse competing remediating strategies at both land management unit and sub-catchment scales, WP6 will utilize best practice models of CEA undertaken in Europe, in relation to water quality and WFD compliance challenges, following the methodological proposal set out in Balana *et al.* (2010). Within this context, this Deliverable presents an analytical review of European CEA studies relating to water quality and Directive compliance. The next section highlights the conceptual and measurements issues related to CEA. Section 3 explores methodological approaches in CEA and their applications. As the CEA to be conducted in WP6 incorporates both the impact of climate change to reflect the uncertainty in costs and effectiveness estimates resulting from climate change, and uncertainty which likely to exist in either the effectiveness of measures or the costs of measures or both, section 4 reviews climate change and uncertainty issues associated with CEA. Subsequently, Section 5 reviews key applications and experiences in CEA studies in the five case study countries (the UK, Norway, Finland, Greece and the Czech Republic) and also in several other European countries. The last Section briefly summarizes and assesses commonalities and differences of Pan-European CEA applications related to water quality and WFD compliance challenges.

# 2 Conceptual and measurement issues

## 2.1 Conceptualizing the cost-effectiveness analysis (CEA)

Cost-effectiveness analysis is a decision-support tool that enables the assessment of the cost and the effectiveness of alternative technical measures or policy options in realising a predetermined goal. CEA aims at identifying a specific option/set of options of achieving a given goal at the minimum economic cost among a range of potential options. This method can be used either in ex-ante assessment, i.e., before the implementation of the relevant measures or in ex-post evaluation, i.e., when measures are already put in place. CEA is particularly suited to situations where the relevant actions involve environmental change; thus, making it difficult for the corresponding benefits to reliably assign monetary values. According to the WFD, an ex-ante CEA is required for the identification of the least-costly combination of measures to deliver established environmental objectives in water quality management.

A CEA applied in environmental decision-making is basically a multi-step procedure. Firstly, the environmental target to be met is clearly determined given the size of the environmental problem(s) in question. Then, all alternative measures to achieve the stated objective are identified. In the third stage, the total costs of all of the available measures are measured and assessed in monetary terms. In the fourth stage, the outcome or output of the implementation of each measure under consideration is evaluated and quantified in physical units (see section 2.4). Finally, the most cost-effective option among the available alternatives is identified. Usually, the cost-effectiveness ratio i.e. the cost per unit output is computed for each measure and measures are ranked on the basis of their cost-effectiveness.

## 2.2 CEA vs. other decision rules

### 2.2.1 CEA vs. cost-benefit analysis (CBA)

In CEA only the costs of implementing measures are measured in monetary terms but the effects of measures are expressed in physical units. In cost-benefit analysis (CBA), however, both the costs and benefits[1] are accounted in monetary terms. CBA is an applied economic tool often used to guide economic agents in resource allocation or investment project decisions or policy alternatives. It is a technique that is used to estimate and sum (in present value terms) the future flows of benefits and costs of society's resource allocation decisions or policy alternatives to establish the worthiness of undertaking the stipulated activity or alternative, and inform the decision maker about *economic efficiency*. CBA therefore addresses the question of whether the objective (or action) is economically

[1] In CBA, both costs and benefits of an undertaking are considered from the social point of view (i.e. economic, environmental, and social costs or benefits) and valued at shadow prices if markets fail to reflect the true value of a cost or benefit item.

worthwhile: do the benefits exceed the costs? In contrast, the cost-effectiveness approach does not deal with whether the policy objective or target has been set in an economically efficient way. As a tool, CEA is designed to help the user judge the most cost-effective options for achieving a particular environmental objective. Cost-benefit analysis is also present in the economic analysis of the WFD, on the analysis of disproportionality (Martin-Ortega, 2013), and also used in the REFRESH Project (Skuras *et al.* 2010).

### 2.2.2 CEA vs. Multi-criteria analysis (MCA)

Unlike in CEA, where alternative measures or policies are assessed against a single criterion, i.e., the cost of a measure or policy option per unit measure of a specified physical or environmental indicator of effectiveness, Multi-criteria analysis (MCA) involves multiple criteria that may include both qualitative and quantitative aspects. 'Cost of measure' becomes just one of the multiple criteria in MCA. MCA establishes preferences between options by reference to an explicit set of objectives and for which it has established criteria to assess the extent to which the objectives are achieved. The ingredients of MCA include: setting objectives, alternative measures /options/ interventions; criteria (effects or attributes); scores that measure the performance of an option against the criteria (its effects); and weights (applied to criteria/effects) and aggregating scores and weights in order to rank alternative options. For environmental decisions that involve various stakeholders with multiple objectives, impacts, and co-benefits, MCA offers an appealing analytical framework that can accommodate these factors. For instance, riparian buffers, in addition to improving water quality (the principal environmental objective) could also contribute in enhancing biodiversity and act as a wildlife corridor (co-benefits) or generate pollution swapping (negative impact). In such a situation, MCA appears to be an important assessment tool, and it has also been advocated in the context of the assessment of measures for WFD compliance (Messner, 2006, Berbel *et al.,* 2011). CEA, however, still remains a commonly applied assessment tool, perhaps mainly due to the relative importance of the 'cost criteria' in many decision environments, and has certainly been the predominant economic instrument in the context of the WFD up to date (Balana *et al.*, 2011). Defining objectives and formulating different options is not different in MCA or CEA. The difference lies in the selection of criteria (effects) and their weights. As each of the tools discussed above (CEA, CBA and MCA) has its advantages and limitations, Niang–Diop and Bosch (2004) have proposed the following schematic technique in choosing an appropriate decision support tool for the problem at hand.

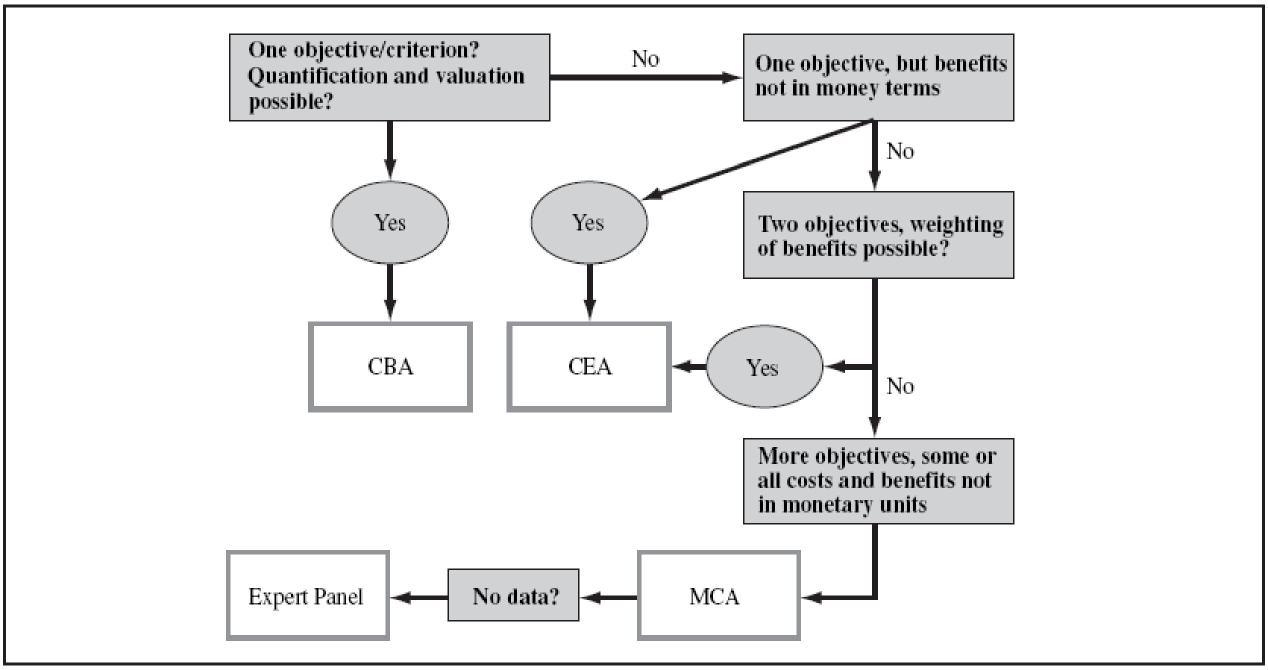


Figure 1. Schematic guideline for choosing decision support tools (Source: Niang-Diop and Bosch, 2004)

### 2.2.3 Limitations of CEA

Because estimating the costs of the policy is generally perceived easier than estimating its benefits, many economists would concede that CEA is somewhat easier to perform than a full CBA. However, CEA suffers from two noteworthy limitations:

a) *CEA does not enable the analysis of aggregate economic efficiency in resource allocation,* because it does not compare the benefits and costs of achieving the objective.

b) *CEA cannot readily be applied to compare decisions with multiple effects*, because it does not convert multiple effects to a common unit of measurement. This is particularly relevant in those cases where at least one option creates significant co–benefits in addition to its main effect. If there are multiple objectives, CEA can only be applied if one objective can, quantitatively, be expressed in the other by assigning importance (weight) to the objectives to arrive at a single yardstick. This is called “weighted CEA”.

As expressed by the figure above, CEA, CBA and MCA are simply different decision support tools aimed at different purposes and should be used accordingly to the policy or decision-making objectives. In the case of the WFD, CEA has been widely adopted for the analysis of economic efficiency of mitigation measures, while CBA is relevant in the context of the disproportionality analysis of the overall aims of the WFD.

## 2.3 Definition and measurement of ‘costs’ and ‘effectiveness’ in CEA

### 2.3.1 Costs

A crucial task to be undertaken in conducting a CEA of an environmental policy involves the identification of costs accruing from implementing the policy in question. Apart from the administrative costs incurred by the regulating authorities (costs of monitoring and enforcing compliance), the compliance process can entail a wide range of costs for the regulated economic units and society as well. Direct private costs are first order costs accruing from the policy implementation and, generally have a "local" character referring to specific target economic sectors. This category comprises all financial costs associated primarily with changes in firm's production process as a direct result of the particular policy. Since these costs are relatively easy to identify and quantify, they usually represent the main focus in compliance cost analysis.

On the other hand, when the target sector is integrated to a large extent with other economic sectors, the interactions among different economic agents operating in different markets tend to generate additional costs in terms of impaired competition, retarded innovation, distorted pricing behaviour etc. These indirect wider economic impacts are generally less tangible than the direct effects making their identification and estimation a more difficult task. However, in instances where the corresponding welfare losses are judged to be significant, the relevant indirect impacts can be assessed on a dynamic basis using general equilibrium estimates (Brouwer *et al.*, 2008).

In a similar context, complying with an environmental policy may impact on the labour market by increasing or reducing unemployment. However, it is a usual practice for CEA of environmental measures to exclude the corresponding indirect social costs. The rationale of such a practice is based on the assumption of great flexibility in labour markets according to neo-classical economic theory. Similarly, impacts on firm competitiveness are not usually included in the estimations of indirect compliance costs, especially in the case of EU environmental policies, where the degree of harmonization of relevant regulations across the EU member states is supposed to be relatively high.

In the light of the above, it can be argued that the decision on which costs to include in a CEA depends to a great extent on the particular case of environmental policy implementation and, of course, on the potential to select reliable and adequate data of different cost elements. In several applications, identifying and measuring the costs associated with environmental compliance has been guided by the principle of "opportunity cost". Implementing a specific environmental policy usually entails the commitment of economic resources with alternative uses. In this way, the opportunity to utilize these resources in an alternative use is lost along with the benefits that this alternative would produce. The opportunity cost of an action, in fact, reflects the foregone benefits of not directing the resources in their next best alternative use.

Another measurement issue is particularly relevant to instances where costs of complying with a policy flow differently over time. In CEA, costs that occur in different time periods must be expressed on a common temporal basis, taking into account the fact that individuals normally prefer to incur the costs later rather than sooner. Discounting is the method commonly performed for expressing future costs at today's equivalent (i.e. present) value accounting for the individuals' time preferences. A controversial issue in the discounting process concerns the choice of an appropriate discount rate. The relevant choice is of great significance since the use of different discount rates may affect radically the results of the analysis and consequently the decisions made. Ideally, for environmental decision-making, the discount rate should reflect the society's preferences for allocating natural resource use over time.

Measuring different types of costs in a CEA is subject to a degree of uncertainty. Sensitivity analysis is a simple way of dealing with uncertainty, providing a range of possible cost estimates. . In fact, determining whether the CEA results vary significantly when, for instance, some additional indirect costs are included or a higher discount rate is used may be valuable in dealing with uncertainty. Ultimately, it is crucial for all the costs selected to be included in a CEA to clearly state their nature, the estimation method and any assumptions made in the computation process, in order to offer better insight into the degree of uncertainty and the reliability of the results.

Taking the above-mentioned issues into account, Box 1 presents an illustrative example on how/where to obtain cost estimates.

**Box 1: Illustrative example on how/where to obtain cost estimates:**

Suppose that we are interested in estimating compliance costs of WFD measures related to land-based activities such as 'land use change' or 'livestock management'. In such a case, two broad approaches can be used to estimate costs of measures.

1. Based on ***farm typology*** (arable, beef, dairy, etc ). This may be appropriate at catchment or sub-catchment scale and commonly used in CEA studies based on 'model' farm types. In this approach, major farm typologies 'representing' the distribution of actual farming practices of a certain catchment or region is defined. The level of activities or scale of farm operations such as area size, number of livestock, input types and usages, existing management practices are defined for each of the 'model' farms. Based on the specific proposed measures, cost estimates (that may include initial investment cost, recurring costs, yield losses, etc.) are estimated for each measure and for each 'model' farm. A good example of such approach can be found in Cuttle *et al.* (2007) diffuse pollution user manual (http://www.lec.lancs.ac.uk/download/defra_user_manual.pdf).
2. Based on ***farm accounts*** (gross margin). This may be more appropriate at farm or even at field level analysis. In this approach actual farm account data (from sources such as farm business survey, agricultural census, etc) is utilized to generate estimates of costs of measures. Unlike the 'model' farm case, here changes in farm gross margins (FGM) as a result of implementing mitigation measures are the basis for cost estimates. A good example of this approach can be found in Bateman *et al.* (2007) (http://www.cserge.ac.uk/sites/default/files/ecm_2007_03.pdf).

### 2.3.2 Effectiveness

Another key issue in the cost-effectiveness assessment is measurement of the 'effectiveness' of measures. This question is central to determining whether the CEA methodology will provide a robust ranking of measures, as different effectiveness indicators may lead to different rankings (Berbel *et al.*, 2010). Most effectiveness assessments are based on estimating the effectiveness of mitigation measures on reducing 'pressures'. One can also assess 'policy instruments' in terms of the extent to which they lead to the successful implementation of those 'measures' which it is tasked to deliver. Also, it is possible to assess the effectiveness on the basis of the 'impacts' of a pollutant, for instance on the ecology of the water body, instead of basing the assessment on 'pressures'.

Different measures will affect different pressures and will have a different overall effect on the achievement of environmental standards. Many measures will only provide a partial improvement and, thus, will need to be combined with other measures. It is important, therefore, that the approach for assessing effectiveness takes this into account. Furthermore, it will need to take into account any overlaps that may exist in what different measures would deliver and any trade-offs or synergies when determining the cumulative level of effectiveness delivered. As a result, there are several factors that should be considered in determining how effectiveness is defined such as the natural characteristics of the water body, the activities leading to the pressures, and the timing of measures. For instance, effectiveness could be considered in dimensions specific to particular activities (industrial discharges, wastewater treatment discharges, abstractions, diffuse pollution, etc.) and the pressures that these give rise to.

Within this context, and in strict terms, the WFD would require effectiveness to be measured in terms of ecological impacts, since the objectives are to improve the ecological status. However, this is very difficult to achieve, and current hydrological models do not have full capacity of accounting for ecological changes (though this is one of the issues that REFRESH is trying to achieve). In any case, there can be estimates of pressure-specific effectiveness, which in turn, can lead to different ranking of measures.

One approach to assessing effectiveness is to consider 'pressure-by-pressure' basis, with the effectiveness of each measure in delivering a reduction in a particular pressure based on percentage reduction in risk (e.g. % reduction in total phosphorus load). This approach alone, however, would not enable a fair comparison between measures that would deliver a reduction in several pressures against one that would deliver a reduction against only one pressure. Furthermore, it may not adequately highlight synergies and trade-offs. Multi-pressure effects of different measures can be assessed by developing a certain form of index reflecting the aggregate pressure. A measure's effectiveness could then be assessed in relation to its relative performance against its capacity to reduce the aggregate level of pressure. A key question in developing such an index is how to assign weights to individual pressures. The manner in which such weights are developed could be critical to their acceptance by stakeholders, particularly, if different weights imply different programmes of measures. Further factors that need to be fed into the analysis are those of timing and the risk that a measure will not deliver as expected. The timing issue is critical to whether or not environmental objectives are met within the WFD timetable. It will also be important to ensure that issues associated with the even-handedness of measures are considered within the assessment; for example, should point sources that are readily and quickly controlled through existing regulatory frameworks be required to take a higher level of action than diffuse sources which may take longer to deliver a reduction. From an economic perspective, the answer to this question relates to a comparison of the costs and benefits over time associated with alternative measures aimed at point sources and diffuse sources. This indicates that for the purposes of the CEA, timing should be factored into the measure of effectiveness, for example through discounting the measure of effectiveness.

Postle *et al.* (2004) proposed a framework for that moves from an assessment carried out at a generic level and at a higher scale (national/regional) to one carried out at a lower scale (river basin/water body). They suggested that, the effectiveness of measures that apply nationally have to be re-assessed when the analysis moves to the more detailed water body level assessment. The national measures may form a 'core' set which should be factored into the baseline, with the effectiveness of water body specific measures then assessed in relation to this adjusted baseline. The outputs of either the generic assessment or the more detailed assessments of effectiveness would then be combined with costs to develop the most cost effective combination of measures under the programme.

'Effectiveness' is essentially a biophysical concept. Where and how to obtain effectiveness data is one the two major questions in CEA (the other being cots data). Data on effectiveness of measures can be obtained from various sources. This may include expert judgement; data from experimental works; previous modelling works; literature survey; using 'benefit transfer' method; and exploring the WFD River Basin Management Plan (RBMP) documents for different River Basin Districts. Based on the work by Balana *et al.* (2011), the following paragraph illustrates how effectiveness of buffer strips for phosphorus mitigation Scottish loch was estimated. Here we illustrate a case study on field-by-field estimates of buffer strips P trapping efficiency based on expert judgement and literature metadata analysis in Lunan catchment (Rescobie Loch catchment), East Scotland . Rescobie Loch is a shallow, eutrophic Loch in Angus, Eastern Scotland. The catchment is dominated by mixed arable farming, much of it on moderate to steep slopes. A number of small streams and ditches drain into the Loch, as well as the main stem of the Lunan Water. Much of the riparian land is poorly buffered mixed arable farmland, which leads to significant quantities of sediment and nutrients (N and P) being transported into the Loch from surface runoff and drain flow from farmland.

Box 2 presents an illustrative example on the measurement of effectiveness of a policy instrument/measure.

**Box 2: Buffer Illustrative example on how/where to obtain cost estimates:**
Consider a 5 ha farm field located at 0.5 km distance from the Loch and with four crop rotations: winter wheat-spring barley-spring barley-seed potatoes. Firstly, based on crop type (land use) expert judgement on P export coefficient in kg P/ha/yr was obtained. Multiplying this figure by the field size results in total P produced from the field. Secondly, putting buffer strip of a certain width traps a certain proportion of the P produced from the field. Estimates of buffer efficiency of various widths, say, 2 m, 6 m, 10 m or 20 m buffers were obtained from expert judgement and literature metadata analysis. The total P produced from the field was adjusted with buffer efficiency coefficient. Third, a delivery ratio figure to each field in geo-database was assigned based on distance of the field from the Loch. Adjustment with delivery ratio results in the P input to the Loch with buffers. Subtracting P input into the Loch with buffer from that of without buffer gives the amount of P reduced attributable to buffer strips. Repeating the exercise four times for the four rotations and calculating the average gives the mean buffer efficiency in terms of the amount of reduced P into the Loch from a particular field.

# 3 Methodological issues – approaches in CEA

A range of conceptual approaches can used to assess the cost-effectiveness of technical measures or policy instruments. Methods range from simple optimization (often operating in a spreadsheet) to complex mathematical programming and integrated bio-economic models. Most CEA models, however, are based on variants of mathematical programming/optimization models.

## 3.1 Optimization based CEA models

The mathematical programming formulations as applied in CEA can be summarized under two broad headings:

(i) *Cost minimization* - based on achieving an exogenously determined level of environmental target or quality standard at a least possible cost. Many CEA studies have pursued this approach (Atkinson, 1974; Gren *et al.*, 1997; Hanely *et al*., 1998; Elofsson, 2003; Zylicz, 2003; Khanna *et al*., 2003; Yang *et al*., 2003; Yang and Weersink, 2004; Yang *et al.*, 2005; Rashford and Adams, 2007; and Froschl *et al.*, 2008).

(ii) *Benefit maximization* - based on maximizing aggregate level of environmental benefits from a given budget outlay (i.e., maximizing environmental benefits by choosing abatement activities subject to a given budget outlay). For example, Azzaino *et al.* (2002) and Ancev *et al.* (2008) pursued this approach.

Mathematically, optimization models based on the first approach can be formulated in a generic form as follows (equation 1):

$$\left.\begin{array}{l} Min. \sum C_i(e_i) \\ \\ Subject\ to \\ \sum_i e_i \geq \overline{R} \end{array}\right\} \quad \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots \ (1)$$

Where $C_i(e_i)$ is the cost function (i.e. cost of reducing the pollutant from the $i^{th}$ source as a function of the amount of pollutant mitigated); $e_i$ is the amount of pollutant abated; and $\overline{R}$ is the desired level of pollutant reduction (i.e., specific environmental target). This simple representation can be expanded by incorporating, for example, various agri-environmental schemes, pollutant export coefficients, farm types and other sources of pollution (Gren *et al.*, 1997; Yang *et al.*, 2005). Specification/derivation of cost functions and levels of pollutant leaching and delivery depends on the nature of the problem under investigation.

CEA models based on the second approach can be conceptualized using the following general mathematical formulation (equation 2):

$$\left.\begin{array}{l} Max.\sum_i \theta_i(a_i) \\ Subjet\ to: \\ \sum C_i(a_i) \le \overline{C} \end{array}\right\} \quad \text{..........................................................} \quad (2)$$

where $\theta_i$ is the value of pollution abatement index from the i^th^ source; $a_i$ is the level of abatement activity at source $i$; $C_i(a_i)$ abatement cost function when abatement activity $a$ is implemented in source $i$; and $\overline{C}$ is the budget that the Environmental Agency has devoted to pollution abatement.

### 3.2 CEA models based on Bayesian Belief Network (BBN)

Bayesian Belief Network (BBN) methodology provides a powerful, intuitively and visually appealing tool for combining (uncertain) information from different sources into a common framework and for analysing a particular system's functioning and characteristics. Bayesian networks utilise probabilistic, rather than deterministic, expressions to describe relationships among variables. In a Bayesian network, the system is represented in a graphical model, in which the subsystems (i.e. variables) are represented by nodes and the causal interactions between the variables by arrows linking the particular nodes. Each dependence indicated by an arrow represents a conditional probability distribution (in the form of a discrete "conditional probability table") that describes the relative likelihood of each value of the down-arrow node, conditional on every possible combination of values of the parent nodes. A probability function is attached to each node, and probabilities are combined in the model using Bayes' theorem. Bayes' theorem shows the relation between one conditional probability and its inverse; for example, the probability of a hypothesis given observed evidence and the probability of that evidence given the hypothesis (Howson *et al.*, 1993). A BBN, therefore, provides both a tool for reasoning under uncertainty and a statistical model of the domain of interest (Jensen, 2001).

Bayesian analytical techniques began to be considered seriously by ecologists and resource managers largely as a result of a special edition of Ecological Applications published in 1996 (e.g. Ellison, 1996). Recent applications of BBNs in resource management include those developed by Rieman *et al.* (2001) and Marcot *et al.* (2001) who developed BBNs for aquatic and terrestrial vertebrate species found on federal lands within the interior Columbia River basin in the United

States, and Bromley *et al.* (2005) who explored their application to integrated water resource planning.

Bromley *et al.* (2005) describe integrated management as a key to the sustainable development of Europe's water resources. Using Bayesian beliefs networks allow a range of different factors to be linked together, based on probabilistic dependencies and, at the same time, BBN provides a framework within which the contributions of stakeholders can be taken into account. A further strength described in this report is that BBNs explicitly include the element of uncertainty related to any strategy or decision. The links are based on whatever data are available. This may be an extensive data set, output from a model or, in the absence of data, be based on expert opinions. Barton *et al.* (2006) used a Bayesian belief network approach to conduct decision analysis under uncertainty of nutrient abatement measures in the Morsa catchment, South Eastern Norway, structuring available cost-effectiveness studies, eutrophication models and data in a Driving Forces-Pressures-State-Impacts-Responses) (DPSIR) framework. The report demonstrates that BBN can be used to conduct cost-effectiveness and benefit-cost analyses under uncertainty, responding to the economic analysis requirements of the WFD. Forward and backward sensitivity analyses and information analysis are also demonstrated using the Hugin Expert software.

Barton *et al.* (2008) used a Bayesian network methodology in the catchment of Storefjorden, South Eastern Norway, to integrate models of phosphorous (P), abatement costs and effects, as well as models of lake P and eutrophication dynamics. The Bayesian network integrated model was used to explore and evaluate the probable and improbable outcomes and uncertainties of (i) the eutrophication problem and (ii) the CEA of the corresponding abatement measures. In addition, factors which affect the reliability of transferring cost-effectiveness data for nutrient abatement measures between river basins were detected with a view to informing Norwegian implementation of the WFD, and the relative uncertainty of model components within the Bayesian influence network was evaluated with an aim to uncover "information gaps" in abatement planning, and as a tool for prioritising future eutrophication research.

Fig. 1, adapted from Barton *et al.* (2008), illustrates the difference between a Bayesian network and influence diagram in the context of a generic driver-pressure-state-impact model, for example for water quality management. In this stylised example, the management context is made up of the states of an exogenous variable X conditioning water quality state S, and the decision D on whether the pressure P mitigating measure is implemented or not. In this framework, prior knowledge of water quality can be expressed as a probability of a state S given pressure P and exogenous variable X: $Pr(S \mid P, X)$. Similarly the probability of nutrient loading pressure P dependent on the decision D is $Pr(P \mid D)$. In an impact analysis, a manager may be interested in determining the posterior probability for a state given a pressure and the states of context variables $c = c(D, X)$: $Pr(S \mid P, c)$, or conversely a likelihood, expressed as a probability of pressure given the state of given context variables: $Pr(P \mid S, c)$. Bayes' rule (Eq. 3) expresses the relationship between the prior, likelihood and posterior probabilities.

$$Pr(S|P,c) = \frac{Pr(P|S,c) \times Pr(S|c)}{Pr(P|c)} \quad (3)$$

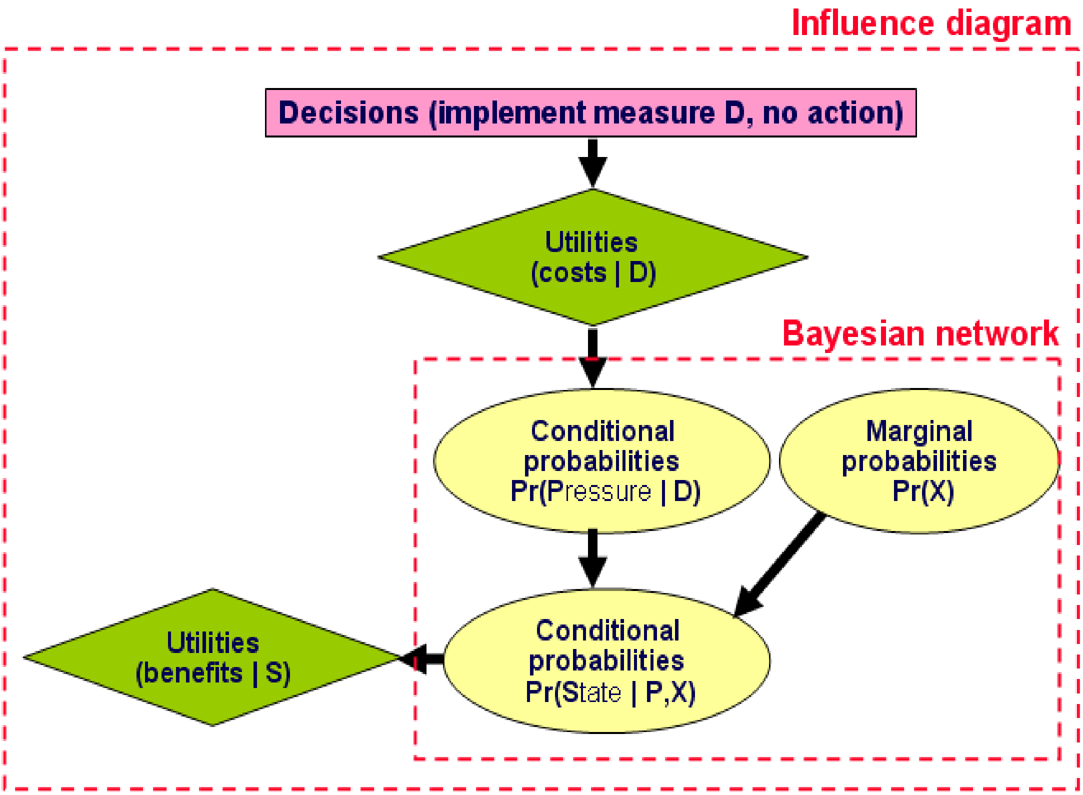


Fig. 1. Influence diagrams and Bayesian networks in the context of DPSIR (Driver–Pressure–State–Impact–Response). Note: Decision nodes are represented by rectangles, utility nodes by diamonds and chance nodes by ovals (Source: Barton *et al.*, 2008).

Cost-effectiveness can be calculated both outside and within the BBN. One can include nodes called "Euro/kg" (Euro per kilogram) to show each measure's cost-effectiveness. In Barton *et al.* (2008) cost-effectiveness is measured as Euro/kg phosphorus on-site at the 'end-of-pipe' for each nutrient abatement measure, i.e. without regard to the eutrophication effect downstream. Another way of looking at cost-effectiveness is to look at the increase in probability that the water body will be of "good ecological status" per million Euro spent on abatement.

### 3.3 CEA models based on Regression Analysis

The economic impacts on agriculture of policy measures (e.g., in terms of changes of farm gross margin (FGM) defined as the difference between farm revenues and variable costs) can be assessed using a regression analysis based on simple econometric relations between economic impacts and agricultural activities from farm-level data. These farm-level economic cost estimates i.e., changes in FGM) can be generalized to estimate the costs of agri-environmental measures (e.g., WFD measures) at a catchment level. This can be combined with prediction of hydrological model to allow an assessment of cost-effectiveness of WFD measures. In this respect, Fezzi *et al.* (2008) developed an integrated modeling approach and demonstrated the model's empirical appeal using real world data from UK farm business survey (FBS) and agricultural census. The model was tested in a case study in the Derwent catchment, North East England. This work pursued the following approaches:

(1) Derivation of farm-specific economic impacts of the implementation of WFD measures using FBS farm accounts data (see Fezzi et al., 2008),

(2) Generalization of the results by regressing those costs on spatially distributed explanatory variables such as land use and livestock numbers,

(3) Specification of the hydrological model which derive nutrient leaching and concentrations, and

(4) Selection of case study area (e.g. catchment), simulate the implementation of WFD measures in terms of costs and changes in nutrient leaching and concentrations and combine these to investigate cost-effectiveness.

Regression models can also be used to predict the amount of abatement of pollutant from i-th source $\theta_i$, as described by the Equation 2 in section 3.1, which depends on various characterizations of the i-th source, e.g. land use, number of livestock, soil types, topography, and weather conditions. Mathematical statistics provide suitable tools to estimate the relative importance of each type of "contributor" to the pollution of the given type of the water-body. The dependence between the severity of each stressor and the quality of water in the given water-body is determined by various factors. We can estimate this dependence on a limited representative sample of case studies and then extrapolate the results to the whole watershed or even wider area. Regression analysis can be used to estimate the 'dose-response' relationship in the water quality. For example, Davies and Neal (2007) used the regression method to establish the empirical relationship between the concentration of nitrate and the catchment characteristics. The regression equation was estimated based on the measurements of the nitrate distribution at 51 sites across UK.

## 3.4 Comparative discussion of applied CEA modelling approaches

Variants of the conceptual approaches described in the preceding sections have been applied in various CEA studies: mathematical programming (Bartolini *et al.*, 2007); bio-economic modelling (Semaan *et al*., 2007; Mouratiadou *et al.,* 2007); non-linear mathematical programming models (Brady, 2002); linear programming (Azzaino *et al.*, 2002); agriculture sector mathematical programming (Ribaudo *et al.,* 2001); integrated agri-environmental models (Schou *et al.*, 2000); damage index methods (Schleiniger, 1999); expert ecosystem attribute scoring methods (Macmillan *et al.*, 1998); regression models (Szoege *et al*., 1996); BBNs (Barton *et al*., 2008); and simulations and optimization models (Paulsen and Wernstedt, 1995; Lise and van der Veeren, 2002). A brief review of some of these studies is presented below.

As discussed in the previous section, mathematical programming or optimization models are the most widely used methods in applied CEA studies. The level of complexity depends on the purpose, scale, data availability, and scope of the problem under investigation. Azzaino *et al.* (2002) applied a simple binary optimization model for selecting the best parcels of land for riparian buffers to prevent sources of drinking water pollution in the Skaneateles Lake watershed in the eastern USA. Defra (2004) and Cuttle *et al.* (2006) used linear programming models to estimate the economic costs and effectiveness of a range of measures that could be adopted to reduce diffuse water pollution from agriculture in the United Kingdom. Forschl *et al.* (2008) used a linear optimization model in their empirical appraisal of the costs to farmers and effectiveness of four nitrogen load reduction measures to waters entering the Black Sea from Austria, Bulgaria, Hungary and Romania. They found that optimization at international level is more cost effective than measures implemented independently in each country.

Among the key criticisms of linear optimization models are their empirical weakness in the use of 'average farm' data and inability to incorporate non-linear relationships and uncertainties. Several studies attempted to deal with these issues. With regard to uncertainty, for instance, McSweeny and Shortle (1990); Gren *et al.* (2000); Gren *et al.*, (2002); Elofsson (2003); Lacroix *et al.* (2005); and Ancev *et al.* (2008) explicitly incorporated uncertainty and applied chance-constrained programming modelling approaches in their cost-effectiveness studies. Similarly, the non-linear nature of effectiveness and costs of measures were recognized in a number of CEA studies. In this regard, considerable progress has been made in developing nonlinear optimization models by integrating bio-physical process models (including simulation models) and economic behavioural models, commonly termed as 'bio-economic models' (Vatn *et al*., 1999; Rashford and Adams, 2007; Khanna *et al.,* 2003; Semaan *et al.,* 2007; Booker *et al.,* 2005; Ward *et al.,* 2006; Ward, 2009; Gurluk and Ward, 2009; Schou *et al*., 2000; van der Veeren and Lorenz, 2002; Lise and van der Veeren, 2002; Pualsen and Wernstedt, 1995; Volk *et al.,* 2008; Yang *et al.*, 2003) for analysing agri-environmental measures and water policy.

For instance, using a bio-economic modelling approach (combined agronomic simulation and multi-objective programming models), Semaan *et al.* (2007) have analyzed the effect of agricultural policies on irrigation water application efficiency, farmers' revenue and nitrate leaching reduction and the cost-effectiveness of various non-point pollution abatement policy measures in Southern Italy. Similarly, by combining a partial-equilibrium behavioural agricultural sector model, farm account statistics, GIS-based spatial production structure, and nitrate leaching models, Schou *et al.* (2000) analysed alternative nitrogen policies in terms of their effects on farm value added, nitrate leaching, and cost-effectiveness for Danish agriculture. Brady (2003) has applied a spatially distributed nonlinear programming model to link changes in agricultural production on crop farms and coastal nitrogen loads in southern Sweden. In the United States, Yang *et al.* (2003) applied an integrated economic, environmental and GIS model to study the cost-effective retirement of cropland to achieve a 20% reduction in sediment loadings in the Illinois River.

Several studies applied other approaches, such as regression models (Szoege *et al.,* 1996; Marshall *et al.,* 2000); numerical analysis (Iho, 2004); damage index method (Schleiniger, 1999); and expert-based system for scoring in a GIS (Macmillan *et al.,* 1998). To assess the cost-effectiveness of nitrogen reduction measures in Switzerland, Schleiniger (1999) used the damage index method. Using the tool of CBA, Hanley and Black (2005) have evaluated the costs and benefits of implementing the WFD in Scotland at two scales – micro and macro scale analysis. In the macro level analysis, they compared the national benefit of implementing WFD and the costs to impacted industries and found positive net social benefits for Scotland as a whole.

In general, the choice of a particular methodological approach or combination of techniques in cost calculations, emission reduction estimation, and combining costs and effectiveness in a given CEA study depends on the specific environmental problem at hand, the availability of data and the degree of uncertainty inherent in cost and effectiveness information. The method chosen in empirical CEA assessments should thus enable us to rank or prioritize alternative environmental remediation measures on the basis of the cost-effectiveness criterion for achieving environmental objectives.

# 4 Climate Change and uncertainty issues

Even though the WFD does not explicitly address climate change issues, frequent references to future forecasts imply the need to account for climate change impacts into the planning process and economic analysis (EEA, 2007; Dworak and Leipprand, 2007). Typically, future trends referring to climate change impacts are assessed and incorporated in economic analysis through the use of climate projections and scenarios. Utilizing such scenarios in conducting the CEA can potentially reduce the uncertainty in cost and effectiveness estimates that stems from climate change.

However, there are additional sources of uncertainty that could affect CEA results and consequently, decision-making. As referred in the WATECO document "uncertainty on costs, effectiveness and time-lagged effects of measures needs to be dealt with throughout the economic analysis process, and more generally throughout the process of identifying measures and developing the river basin management plan" (European Commission, 2003). Importantly, uncertainty and its incorporation in decision making required for the implementation of the WFD has been identified as one of the key issues that require further investigation (European Commission, 2003).

Sigel *et al.* (2010) provide a conceptual framework for perceiving and describing uncertainty in environmental decision-making. Brouwer and Blois (2008) recognize several forms of uncertainty concerning water resources management. *Environmental uncertainty* refers to imperfect knowledge and information about the functioning of the water environment, while *economic uncertainty* is related to imperfect information about the current and future social and economic values of water systems. In the case of the WFD, *political uncertainty* also exists about the decision issues, decision criteria, decision-making procedures and political, institutional, social, financial and economic consequences of decisions to be taken. In the context of political decision-making, a distinction has furthermore been made between *goal uncertainty* (policy objectives), *action uncertainty* (composition of alternative sets of measures) and *yield uncertainty* (costs and benefits of alternative sets of measures).The next two sections review basic studies that address uncertainty and climate change issues in the context of CEA to serve water management decisions.

## 4.1 Incorporating uncertainty in CEA

CEA is a time-defined analysis based on cost and effectiveness estimates for alternative options referring to a specific environmental target. Estimating/predicting different types of costs and effectiveness in a CEA given a specific time-scale is subject to a degree of uncertainty. This degree is likely to vary depending on a series of factors. The particularities of the specific environmental problem at hand, the nature of costs to measure, the way effectiveness is assessed, the estimation method and any assumptions made in the computation process, as well as the time and spatial scales are the most common factors introducing uncertainty in CEA. The uncertainty being inherent in the cost estimates as well as the effectiveness is likely to affect the cost effectiveness outcome and subsequently the ranking of the measures under consideration. Even though uncertainty issues are considered crucial in water resources decision-making (Xu and Tung, 2008; Tsakiris and Spiliotis, 2011; Sigel *et al.,* 2010), only few CEA studies explicitly account for uncertainties in their models (Balana and Slee, 2011; Berbel *et al.*, 2010).

The use of intervals of costs and effectiveness estimates, instead of point estimates as well as sensitivity and scenario analysis are probably the most common ways to deal with uncertainty in CEA. A simple sensitivity analysis would be to calculate the change in the cost-effectiveness ratio, while altering key parameter values one at a time, while holding all other parameters constant. If this procedure yields the same outcome, then the results of the initial CEA can be considered as robust. Berbel *et al.* (2010) conduct a sensitivity analysis to consider uncertainty in both costs and effects estimates based on three scenarios derived using a simulation tool. They found that the ranking of some measures in fact depends on the scenario considered. Brouwer and Blois (2008) used statistical analysis based on Monte Carlo simulation along with expert judgment to quantify and assess the combined impact of environmental and economic uncertainty on the selection of cost-effective water policy measures. Even though they find the same ranking of the considered measures as applying a deterministic analysis, they argue that the probability distributions for the cost and effectiveness estimates may provide significant information to policy makers.

Following the seminal works of McSweeny and Shortle (1990) and Shortle (1990) some studies have used stochastic programming to account for uncertainty, where the decision problem includes minimisation of costs for pre-specified environmental targets under probabilistic constraints. In this framework, Bystrom *et al.* (2000) studied the issue of controlling nitrate pollution in wetlands, illustrating the impact of uncertainty on the optimal solution. Gren *et al.* (2000) and Gren *et al.* (2002) incorporate stochastic pollutant transports in their cost-minimization models to account for uncertainty related to the quantification of land-based loads from non-point emission sources. In a similar context, Gren (2008) analysed the impact of risk linkages between mitigation and adaptation measures on cost-effective solutions to given pollution reductions under conditions of stochastic loads to water recipients. Elofsson (2003) utilised a chance-constrained programming model to calculate cost effective solutions to nutrient load reductions taking into consideration the stochastic relationship between abatement measures and the corresponding impact on nutrient loads. Extending the probabilistic cost-effectiveness approaches, Lacroix *et al.* (2005) used bio-physical model predictions to evaluate the cost-effectiveness of a series of farm management scenarios for reducing the nitrate concentration in drained water.

A more recently-developed strand of literature recognizes the usefulness of Bayesian belief networks (BBN) in water management decision making under uncertainty. Contrary to the above mentioned approaches where uncertainty assessment is an integral part of the original analysis, BBN modelling can be seen as a 'holistic' meta-analytical approach imposed on a number of disparate original studies (Brouwer and Blois, 2008). Regarding uncertainty considerations in the context of

WFD, BBN offer a framework for documenting and assessing the probability of non-compliance against uncertain abatement costs, as well as the probability that expected costs exceed expected benefits, by jointly assessing conditional probability distributions for abatement costs, water quality effects and benefits (Barton *et al.,* 2008). So far there is limited research on applying BBN to water management (e.g. Varis and Lahtela, 2002; Ames *et al.,* 2005) and even more limited to CEA. In this respect, the works of Barton *et al.* (2006) and Barton *et al.* (2008) are considered as most promising. These studies demonstrate the application of BBN to integrated decision-making process in economic analysis (that is CEA and cost-benefit analysis) under the WFD, discussing the relevant pros and cons of this approach in the light of the results from a relevant application.

## 4.2 Considering the impact of climate change on cost and effectiveness estimates

The time horizon is of great significance when conducting a CEA. In the water management decision-making, the relevant objectives are commonly designed to be accomplished within a specific time-frame. Climate change has brought into the scene a number of crucial issues referring to floods, water scarcity and droughts impacting on water quantity and quality over time (EEA, 2007; IPCC, 2007). Such impacts entail significant implications for designing water management plans to realize specific environmental objectives based on cost-effectiveness criteria. This is relevant to the case where CEA results are utilized in order for cost-effective and viable water management measures to be implemented on a long-term basis under changing climate conditions.

Recent attention has been drawn to the assessment of the main risks and uncertainties which climate change could be posed to the delivery of the WFD objectives. Failure to allow for climate variability and change at critical stages in the WFD could lead to disproportionate costs, undermining of benefits from programs and/or measures over time, and ultimately failure in meeting the environmental objectives set out in river basin management plans (Wilby *et al.,* 2006). In a CEA context, the potential impacts of climate change on costs and effectiveness of water management measures can be critical. First, climate changes are likely to create new and/or affect existing pressures on water bodies which will potentially affect both the gaps in water status and the costs of reducing the relevant gaps. On the other hand, the effectiveness of long-term water management measures (e.g. measures concerning agricultural practices) may be adversely affected by extreme weather events. Also, indirect impacts of climate change arising from changes in land use could modify patterns of diffuse pollution and require new abatement strategies (Whitehead *et al.,* 2006).

To account for climate change in CEA-based decision making, simple sensitivity analysis using the range of possible climate impact futures, may be sufficient, and substantially computationally less intense (Patt, 2011). On the other hand, probabilistic cost-effectiveness approaches generally fail to distinguish between uncertainty due to climate variability and uncertainty due to lack of knowledge. Lacroix *et al.* (2005) propose the use of bio-physical models which include climatic variables in assessing the probabilistic cost-effectiveness of the farm management practices, considering in this way, climate variability. Meta-models based on Bayesian network structures may also be utilized for analyzing the role of climate change for water management decisions. In this direction, the next generation of probabilistic climate change scenarios can create new opportunities for options appraisal and impact assessment within risk-based frameworks that enable weighting multiple uncertainties (Wilby *et al.,* 2006).

Incorporating climate change effects in cost-effectiveness studies implies a number of issues to be properly addressed. Scale is considered to be one of them. Climate change constitutes a global problem, but the kind and the intensity of effects tend to vary at the international or even national level. Hence, the scale at which climate change impacts are assessed is a significant issue to address in CEA studies. Importantly, the way in which water management measures aiming to achieve WFD targets counteract with measures of adapting to climate change is usually viewed as an additional challenge (Dworak and Leipprand, 2007). The "climate-proofing" character of the proposed water management measures should be investigated to ensure that the related measures support adaptation to climate change or at least do not run counter to it. Finally, the cost-effectiveness of options may vary according to the discount rate used, and this may be important particularly for longer-term options.

From the above it can be deduced that there are several issues to consider when climate change impacts are taken into account in the long-term water management planning in order for the most cost effective and viable package of measures to be selected and successfully implemented. Not considering climate change impacts and relevant uncertainties in CEA may result in selecting packages of measures that are not viable or cost-effective over time.

# 5 Review of CEA applications related to WFD

## 5.1 REFRESH demonstration catchment countries

### 5.1.1 The United Kingdom

One of the most comprehensive and earliest scoping studies on outlining alternative methodologies for conducting WFD-related CEA in UK has been provided by Postle *et al.* (2004). This study suggested that the CEA methodology to be applied in the UK should adopt full economic costs (as opposed to financial costs) as a basis for measuring costs to appraise potential measures. Within the context of delivering good environmental status, a range of economic costs to be considered in CEA were identified including: direct costs of complying with the requirements, any welfare losses to consumers, any non-water environmental costs, induced effects to the wider economy, and transaction costs. The study also proposed a generic framework for estimating and aggregating costs at different scales (local, basin, and national levels). With regard to assessing the effectiveness of measures, this document set out a staged approach moving from the technical screening of measures to a generic assessment and finally to a detailed assessment (see Postle *et al.,* 2004, pp. 56-58). Formal optimization methods were suggested for combining the estimates of costs and effectiveness in order to develop the most cost-effective combination of measures.

Building on Postle *et al.* (2004), the UK Collaborative Research Programme (CRP) on River Basin Management Planning Economics has developed six sequential projects. They are focused on the progressive development of the approaches and data required to assess the costs and benefits of Programmes of Measures (PoMs) proposed under WFD. One of the core projects of CRP (project 2) was on "developing a methodology and guidance to assess the cost-effectiveness of measures and combination of measures for the WFD" (Risk and Policy Analysis (RPA), 2005). This project provides an overarching framework for undertaking CEA under WFD in the UK. Since the UK Government Department – the Department for Environment, Food and Rural Affairs (Defra) – is the responsible body for policy and regulations on the environment, food and rural affairs; chaired the UK CRP on River Basin Management Planning Economics; and commissioned a number of CEA case studies related to WFD measures, our review in this sub-section gives special attention to these and provides a brief summary of WFD implementation in Scotland.

Defra (2003) assessed the cost-effectiveness of 40 mitigation options against 15 farming scenarios for England and Wales. The 'Phosphorus loss estimation for agricultural systems with mitigation measures' (PLEASE) expert system was used to estimate P losses and explore the effects of different mitigation options under different model farm scenarios. Enterprise level data were employed to calculate economic costs. On the other hand, the Defra (2004) study assessed the costs and effectiveness of 49 nitrate mitigation measures against the base-line scenario in one arable farming and three grassland systems. Various N-cycling models were used in estimating nitrate leaching from each of the baseline systems and simulated the impacts of different measures on nitrate leaching. The cost of implementing each measure was calculated using available data and expert consultation. The results from both the Nitrate and Phosphorus studies (Defra, 2003; 2004) show that generally, there is greater scope for implementation of cost-effective measures in arable systems than in grassland systems. Both studies focused on a single pollutant ('pollutant-centric' approach). The approaches utilised in both studies are basically similar.

The COST-DP project (Defra, 2005) examined a range of pollutants (ammonium, nitrite, pathogens and biological oxygen demand) and adopted a 'measure-centric' approach. A new model framework (Cost-Cube export model), based on the 'export coefficient' approach (Johnes, 1996), was developed. The new framework provides explicit representations of the fractions of the total pollutant loss by aspects of *source, mobilization* and *transport* dimensions. Accordingly, for each of the three stages of the pollutant flux pathways, lists of measures were identified, and their effectiveness and costs were assessed for each model farm scenario. Cost estimation was based on capital investment and annual maintenance costs.

A study by Cuttle *et al.* (2006) on "An inventory of methods to control diffuse water pollution from agriculture (DWPA)", provides more detail than the three earlier Defra-commissioned CEA studies (Defra, 2003; 2004; 2005). Cuttle *et al.* (2006) developed a 'User Manual' for DWPA which provides a platform for model framework development and succinct information on individual mitigation methods to assist the user in developing policies to control diffuse water pollution. By combining the empirical and expert assessments, the Manual presents the impact of 44 different selected methods against model-based baseline losses for three main pollutants – nitrate, phosphorus, and faecal indicator organisms (FIO) – from seven representative model farm systems on clay loam and sandy loam soils and assuming a medium rainfall of 850mm/year. The information provided in Cuttle *et al.* (2006) served as reference points and building blocks for many of the subsequent CEA studies in UK, for instance, Anthony (2006), Bateman *et al.* (2008), and Fezzi *et al.* (2008).

The CEA approaches examined so far were either 'pollutant-centric' or 'method-centric' in which a particular pollutant or mitigation method was examined individually. Anthony (2006) developed a new methodology for calculating combined effectiveness of multiple mitigation methods and the cost of implementing them. By allowing integrated assessment of the effect of combinations of methods, the new approach has brought about novel ideas compared to the previous approaches. With the help of selected environmental models (PSYCHIC, NITCAT, MANNER and N-CYCLE) and simulation models, baseline losses of nitrate, phosphorus, FIOs and sediment from six model farm systems were calculated for various environmental conditions.

The WEWS Act identifies the Scottish Environment Protection Agency (SEPA) as the competent authority for the Scotland river basin district (RBD) and gives certain duties to Scottish Ministers. As part of a coordinated RBMP process, SEPA has established a national advisory group and eight area advisory groups for the Scotland RBD. One of the major tasks of these groups is to provide advice on cost-effectiveness of PoMs. Cost-effective, on-the-ground solutions to reduce pressures are at the heart of implementing the WFD in Scotland. The government pursues a strategy of working with those responsible for taking action to reduce pressures, encourage innovation, provide information and advice; and encourage them to explore their different options. In the Solway Tweed RBD, SEPA and the Environmental Agency (England) work jointly to establish a co-ordinated approach to implementing the WFD obligations. Paying special attention to participatory approaches, stakeholder engagement and co-ordinating the work of different public sector bodies and other private and voluntary organisations to encourage and ensure action to control and reduce pressures cost-effectively is a distinguishing feature of the WFD implementation in Scotland.

### 5.1.2 Norway

Magnussen *et al.* (2003) produced an overview over CEA related to water environment in Norway. This report shows that Norway has a relatively long tradition of undertaking CEA related to environmental impacts on water bodies. In Norway, this type of analysis has often been named "analysis of measures". Work with "analysis of measures" started in Norway in the middle of the 1980s when the Norwegian Pollution Control Authority carried out two analyses of measures for Lake Mjøsa and the inner Oslo Fjord. Later, several analyses have been carried out, often related to a certain water body under the direction of municipalities or by the developer of hydropower. The costs of measures related to hydropower have in these studies mainly been related to costs for the developer.

The Magnussen *et al.* (2003) review on several CEAs related to eutrophication shows that the CEA studies in Norway were not based on certain generic principles and frameworks. The assumptions, cost and effectiveness estimation approaches/methods, and reporting formats pursued in different studies vary markedly and hence are not comparable. Thus, the review report recommended that CEA studies conducted and reported need to adopt certain generic procedures and framework (but could be adapted to local conditions) to ensure consistency in the interpretation of various research findings and provide valuable inputs to policy/decision making. The report also indicates that geographic scale of most of the CEA studies in Norway is at catchment level. Lyche *et al.* (2001) conducted a CEA with an objective to suggest relevant abatement measures necessary to improve the environmental conditions in the Vansjø-Hobøl watercourse in South-East Norway. This analysis is related to effects of bio-available phosphorous on the water body and assumes that efficiency linked directly to environmental goals in water bodies, requires modelling of proliferation and bioavailability and/or exposure in the waters. In cases where e.g. eutrophication is the main problem and the effect parameter is phosphorous, this study has taken into account that only a component of phosphorus is directly available for the algae production, and that this proportion varies with the source and type of the water body the phosphorous is affecting.

The Lyche *et al.* (2001) study also discusses the complication of costs assessments of agricultural measures. These complications are connected to significant variations in crops and whether to use Norwegian or world market prices. If farm costs are assumed, they mean that the most obvious will be to use Norwegian prices but if the societal costs are assumed, one can discuss if the world market prices should be used in the assessment. The subsidy from the State could be considered to be an income for the farmer or an expense for the State. In this study, they have not considered the subsidy grants in the cost figures. From a societal perspective, the state grants are considered as an expense for the society along with the "administrative" costs associated with management of these subsidies. Lyche *et al.* (2001) found that the most important abatement measures to obtain the environmental objectives in the Vansjø-Hobøl watercourse refer to changes in tillage practices, establishment of vegetation zones at the field edges along the watercourse, construction of sedimentation ponds/artificial wetlands, establishments of grassed waterways in the agricultural sector, removal of direct outlets and solid materials from the separate household sewage sector, and proper connection of misconnected sewage pipelines, as well as reduction of overflow and leakages in municipal sewage pipelines. They also found that climate change may counteract the effect of implemented abatement measures and aggregate the water quality.

One of the recent studies on CEA related to WFD in Norway is a study on improving the cost-effectiveness estimation of measures for reducing agricultural runoff (Refsgaard *et al.,* 2010). The basis for this study is several erosion and run-off experiments. The cost-effectiveness of measures was estimated as a change in costs for the farmer divided by reduced loss of phosphorous. This means that measures with small economic impact for the farmer combined with a good effect on loss of phosphorous will be pointed out as the most cost-effective measures. Refsgaard *et al.* (2010) based their CEA on the direct costs each farmer bears by implementing measures to reduce phosphorous run-off. A combination of in depth-interviews, surveys and data from the Norwegian Advisory Service was used to come up with the cost data. Loss of phosphorous was calculated for each erosion class and the effect of a certain change in a tillage practice was given by the difference between standard tillage (autumn ploughing) and the investigated tillage practice. The study showed that the effect of reduced tillage is dependent of the risk of erosion in the area.

The Refsgaard *et al.* (2010) study assesses a change in the gross margin induced by a change in tillage practices. The assessment is based on crop prices in an unchanged policy regime. They have not excluded the price subsidies on crop because they mean that it is not likely that this will change although the instruments for reduced phosphorous will change. They believe that this gives a good picture of what are the real costs for the farmer of changing the tillage practice in relation to effects on the water environment. The study is thus not a study where the costs and benefits for society as a whole are measured. This cost-effectiveness study does not include transaction costs and subsidy schemes are not included in the gross margin calculations since a part of the purpose is to estimate farmers' costs. This could at a later stage form the basis for evaluation of the contribution rates. The report concludes with that it is not enough to focus on measures in the most erosion prone areas where the measures are most cost-effective if we are to achieve the goal of good ecological status in the water bodies. The cost-effectiveness will, however, vary from farm to farm. It is therefore important to target grant schemes in The Regional Environmental Program to compensate farmers for environmental measures on the most erosion prone areas of the farm.

Despite the above, a review of the Norwegian water management plans shows that Norway has a long way to go when it comes to economic considerations in connection with the WFD. The economic analyses of measures undertaken so far are not sufficient enough to guide cost-effective decisions to manage water resources in order to comply with the WFD requirements. It appears that the uncertainty overshadows the cost estimates which are largely based on expert judgements. Measurement of the effectiveness of various mitigation measures is also not free from the shadow of uncertainty. This, therefore, needs for clarity and guidelines for how this should be handled in future plans (Vannregionmyndigheten for Glomma/Indre Oslofjord, 2010). These examples show that there is a need for standards for conducting CEA studies. In May 2010 the Norwegian authorities published a tender for a report on the economic aspects for the implementation of the Directive with an objective to come up with guidelines for the societal-level analysis in relation to the WFD (Direktoratet for Naturforvaltning, 2010), i.e., a need for systematic assessments of uncertainty in the different parts of CEA as well as the overall uncertainty when several measures are considered together as an alternative (Barton *et al.*,2006; Barton *et al.*;2010).

### 5.1.3 Finland

In Finland, agricultural nutrient releases account for more than half of all the nutrient discharges into water bodies, in spite of recent reductions in the use of fertilizers, the widespread establishment of buffer zones, and the adoption of farming practices that reduce erosion. Research is targeted to identify new cost-effective water protection measures focusing on the Baltic Sea coastal zones as well as on the inland rivers and lakes (mostly southwest Finland, because of the intensive agriculture). Studies specific to the cost-effectiveness of agricultural nutrient reductions in the Baltic basin include Byström (2000), Ollikainen and Honkatukia (2001), Gren (2001), and Eloffson (1997). Hart and Brady (2002) examine the issue of different target types in their study on efficiency of agricultural policies in reducing pollution of the Baltic Sea. Their focus was on regulator’s optimal responses to three alternative ways of setting the target. Brady (2003) introduces regional aspects and agricultural policy into a nitrogen abatement model. He defines a least-cost solution based on present levels of agricultural subsidies as second best and compares it with a least-cost solution based on the situation where subsidies are decoupled from production. The optimal solutions differed substantially in land use allocations, abatement intensity and total costs of abatement. Both Brady (2003) and Hart and Brady (2002) used mathematical programming models.

Cost effectiveness studies on Finnish inland rivers and lakes restoration are rather scarce, though some projects are currently being carried out by the Ministry of Agriculture and Forestry, such as *'Cost-efficiency in agricultural water protection', 'Cost-effective measures and multi-criteria strategies for the management of water resources' (VeKuMe project), and 'cost-efficiency and implementation of nutrient abatement measures and identifies the associated risks of pesticide use'* projects.

The first project studies the cost-efficiency and implementation of agricultural nutrient abatement measures. The cost of achieving water protection targets will be calculated for two key sub-basins, which have been identified based on existing water quality monitoring data. The project will develop improved modelling tools for assessing the cost-efficiency of agricultural nutrient abatement measures; examine and rank different policy instruments according to their cost-efficiency; and give guidelines for designing environmental policy. The goal of the second project is to create guidelines for planning the restoration methods of lakes and their drainage basins in an integrated and cost-effective way. The third project aims at developing an improved modelling tool for cost-efficiency calculations of agricultural nutrient abatement measures. The project will examine and rank different policy instruments according to their cost-efficiency.

One of the most studied areas in Finland is the catchment of Lake Pyhäjärvi in the Southwest of Finland. It has been studied extensively in the last 20 years, as the general usability of water has deteriorated due to the increased algal blooms but shown some signs of recovery during the recent years. Pyhäjärvi has been the object of intensive bio-manipulation for decades, carried out by commercial fishermen. Ventela *et al.* (2007) discuss the long-term management perspectives of Pyhäjärvi lake: eutrophication, restoration –recovery and conclude that it is not currently realistic to achieve the level of water quality of the 1980s due to intensive agricultural use of catchments, lack of cost-effective tools for load reduction and climate change threats.

Iho (2004) studied a cost-effective reduction of 10% from the phosphorus load carried by the river Yläne into lake Pyhäjärvi. Three methods of reducing agricultural runoff are considered: buffer strips, constructed wetlands and reduced use of fertilizer. For solving the problem a general model of minimizing social costs originating from multiple nutrient reduction methods in different regions with a common target level of pollution was created. Cost-effective allocations of methods are solved for 14 sub-basins independently; also a coordinated cost-effective solution is solved for two chosen sub-basins. Iho (2005) continues the study of cost–effectiveness of agricultural nutrient abatement and develops a model which determines the cost-effective allocation of three measures to reduce phosphorus loss from the fields. The model allows for comparisons of costs and shows that the cost effective allocation strongly depend on a target level and even within homogeneous region using the allocation from one reduction level as a guideline for other levels violates the cost effectiveness seriously. Helin et al. (2005) researched abatement costs for agricultural nitrogen and phosphorus loads in the case study of Southwestern Finland, where they develop an empirical framework for estimating abatement costs for nutrient loading from agricultural land. Nitrogen abatement costs and the phosphorus load reductions associated with nitrogen abatement are derived for crop farming in southern Finland. The model is used to evaluate the effect of the Common Agricultural Policy reform (CAP) on nutrient abatement costs.

Varis *et al.* (1994) asserted that uncertainty associated with decision making is one of the primary functions of modeling and monitoring targeted to assist decision making in reservoir, river, and lake water quality management. In many practical activities such as environmental impact assessment, the inference is bound to be based primarily on subjective, expert judgments, supported by empirical data and models. The paper discusses decision analytic approaches to the handling of uncertainty and subjectivity associated to information available, as a decision criterion, and as a component influencing the model structure. Bayesian belief networks were used to estimate uncertainty in the functioning of established buffer zones in Finland (Varis 1999). Tattari *et al.* (2003) analyze the use of belief network modeling to assess the impact of buffer zones on water protection and biodiversity.

### 5.1.4 Greece

Since the implementation of the WFD in Greece has shown a considerable lag with respect to the majority of the Directive's requirements, the progress made thus far in conducting economic analyses implied by the WFD is very limited. Regarding the application of CEA or other valuation. methods, there are only few fragmented studies specific to Greece that are summarised below.

Among the first attempts to evaluate environmental measures targeted to Greek sites is that of Zanou and Kopke (2001). More specifically, this study compares the costs of alternative measures intended to improve the quality of bathing waters in the Gulf of Kalloni located in the Greek island of Lesvos. Taking into account the main sources of eutrophication in this area, four options were examined with roughly comparable effectiveness measured by the percentage reduction in nitrates and phosphates. Using data on direct financial costs, two indicators, i.e. the Net Present Value of total costs and the Equivalent Annual Costs are computed for the four alternatives. Comparing the indicator values across the examined options reveals that the most cost-effective option is the construction of a municipal wastewater treatment plant followed by organic farming and training. However, the authors noted that the results should be treated with caution due to uncertainty arising from the rapid changes of nutrient concentrations in coastal waters.

In a quite different methodological framework, Gerasidi *et al.* (2003) developed a cost-effectiveness method for evaluating water management interventions following the nine standard steps introduced by Orth (1994). The proposed method was applied to assess a series of measures intended to deal with current and future water shortages in the Greek island of Paros. More specifically, using data and information provided by the Water Utility of Paros, the annual total cost of various options was estimated as the sum of annual depreciation of capital cost and the annual operation and maintenance costs. A single indicator of effectiveness is used, that is the percentage of shortage coverage achieved by implementing the alternative options. Applying the proposed CEA methodology and taking into consideration future water demand estimates, a thirteen-year water management plan was designed based on five combinations of measures that were identified as cost-effective. However, the analysis did not account for environmental or socio-economic dimensions in estimating the cost and the effectiveness of alternative water management interventions.

Kontogianni *et al.* (2004) attempted to provide measures of both costs and benefits of improving coastal water quality of the Inner Thermaikos Gulf, i.e. the Thessaloniki bay. The present value of the total cost and the annual economic cost of municipal waste water treatment were estimated based on cost data referring to the existing waste water treatment plants (WWTPs), the WWTPs that were on schedule, and those that needed to be constructed in order to serve the municipalities lacking any waste water treatment. The annual social benefits of cleaning up the bay are estimated using the contingent valuation approach. The relevant survey is based on 466 interviews intended principally to evaluate the maximum willingness to pay of Thessaloniki's inhabitants for improving the water quality of the bay. A significant net benefit on annual basis is estimated by comparing the corresponding measures of benefits and cost. However, the authors draw attention to the existence of potentially high uncertainty in money estimates of both costs and benefits due to a number of factors that are not taken into account in the analysis.

In a rather wider context, the CBA employed by Birol *et al.* (2006), the economic benefits of sustainable management of the Cheimaditida wetland in Greece were measured using total willingness to pay estimates derived in the framework of a choice experiment (CE) methodology. The relevant data came from 407 interviews in the CE survey and were used for the estimation of alternative econometric models (logit, random parameter logit with interactions and latent class models) accounting for heterogeneity in the public preferences. In the light of the above, three potential wetland management scenarios were designed and evaluated weighing their economic benefits against their costs. The net benefit estimates seem to favour the "high impact" scenario, i.e. management interventions aiming at preserving high ecological status of the wetland, offering significant research and educational opportunities and re-training a large number of farmers in environmentally friendly employments.

Psychoudakis *et al.* (2005) propose an integrated framework for the assessment of wetland management scenarios which was applied in the case of the Zazari-Cheimaditida wetland. All costs and benefits of three management scenarios were indirectly valued on the basis of a contingent valuation approach used to estimate the use value of several ecological functions of the examined wetland. A stakeholder analysis, based on a relevant survey was also undertaken to investigate the acceptability and the social impact of realising the scenarios in question. CBA and MCA were employed to compute quantitative performance indicators for the assessment of the scenarios. The results indicate that the most efficient management option is a no-intervention policy, unless a conservatively budgeted management plan is implemented.

### 5.1.5 Czech Republic

In relation to WFD, the Czech Republic is divided into five river basin districts each of which is being administered by its own River Board, a State enterprise. These River Boards are obliged to conduct an economic analysis in accordance with annex III of WFD. The CEA is part of each of these documents. To fulfill this task, a similar methodology, outlined below, has been used by all River Boards.

The cost effectiveness analysis is assembled from three parts – economic evaluation of the programs of measures, analysis of recovery of the cost of water services, and ability of households to pay for drinking water supply and sewage discharge.

In the first part, the so called ‘programs of measures’ (i.e., a group of actions aimed for solving water-management problems) are evaluated from various aspects. These actions are planned, or already are being carried out. The following Programs of Measures are evaluated in the CEA analyses in great detail:

a) Program of measures for water used or intended to be used for human consumption.
b) Program of measures for point emissions control.
c) Program of measures to ensure proper hydromorphological conditions of the bodies of water.
d) Program of measures for control of hazardous substances entry to water, included leaking from old industrial sites.
e) Supplemental measures.

In the framework of the CEA analysis, each Program is evaluated on the basis of efficiency, feasibility, urgency in regard of real needs, and urgency in regard of obligations agreed. The costs of these measures are compared with the available financial means and their feasibility is evaluated.

The environmental measures are grouped into ten clusters:

a) Measures applied to water which is or will be used for human consumption
b) Control on abstraction and impoundment of water.
c) Prevention of direct discharge to groundwater.
d) Control of emission from point sources and control of other impacts of human activities on the status of waters.
e) Control or cessation of hazardous substances entry to water.
f) Prevention or reduction of accidental pollution incidents.
g) Supplementary measures necessary for fulfilling the environmental quality objectives.
h) Polluter-pays principle application.
i) Measures to ensure that the hydromorphological conditions of the bodies of water are consistent with the achievement of the required ecological status or good ecological potential.
j) Control of diffuse sources of pollution

In the second part, the cost of recovery for use of water and water services is calculated including environmental costs. The tighter the bound between the cost and payments for the water supply and discharge the more the “polluter pays principle” is followed. The analysis of the costs is based on the following services: significant watercourses administration, minor watercourses administration, drinking water supply, waste-water collection and treatment. On the side of revenues, there are payments for drinking water consumption and sewage discharge, for surface water abstraction, for point source discharges, and other payments for water services. Another important part of revenues taken into the account are the subsidies from EU and the state. In the third part, social capacity of households to pay for the drinking water and sewage treatment is investigated. Average income of households is compared with the average payment for the water and sewage services. The results are compared with similar data from Western Europe.

However, the methodology used by the River Boards has a number of shortcomings. Some these shortcomings were also reflected at the end of the document. The most serious of these is in the field of cost estimates. According to common practice, in the CEA, only the environmental costs are taken into account, as no sufficient methodology is available for water resource cost estimation. The water resource costs are a phrase used for opportunity costs for different usage of water in terms of various scenarios of development. Water resource costs arise when an alternative usage of water generates a higher economic value in comparison to actual or considered future usage. Since no sufficient methodology is available to assess the water resource costs, these costs have not been taken into account in the CEAs.

For the needs of analysis, environmental costs have been stated as the cost of recreation and restoration of environmental services and spared costs. In other words, the costs incurred by the need of compensation of water services and influence which harm the status of water. According the CEA methodology in Czech Republic, environmental costs are incurred by the following three categories of water services:

a) Pollution of surface water and groundwater
b) Abstraction of surface water and groundwater
c) Hydromorphological influences in regard to watercourses

The costs of compensation of the negative consequences of these services according to the national legislation should be included into the expenses of the provider of these services.

In a nutshell, in the Czech Republic, each water body is involved in some of CEAs conducted by River Boards by the demand of WFD. All the River Boards use the same methodology, which is a plus, because it facilitates to compare various attitudes to water management, to compare efficiency and sustainability, to share good practices, to compare the economic, environmental, and social consequences of water resource exploitation. A significant gap in the methodology is that it only evaluates the actual policy; it does not allow comparison of various scenarios in the terms of cost-effectiveness and thus does not allow ex-ante evaluation of competing policies in the field of water resource usage.

## 5.2 Other European countries

### 5.2.1 Sweden

Swedish national eutrophication policy aims at reducing phosphorus and nitrogen emissions to the Baltic Sea from three sectors: wastewater, agriculture, transport and energy. Elofsson (2012) compares the cost-effectiveness of the past and present national nutrient reduction policy to the international targets set by Baltic Sea Action Plan (BSAP). Each Baltic Sea country has a catchment specific target for phosphorus and nitrogen reductions set in the BSAP. The national "zero eutrophication" target is to reduce 16,890 tons of nitrogen load and 350 tons of phosphorus load by 2010 compared to the 1995 level. The nitrogen load reduction target set by BSAP for Sweden is somewhat bigger, 20,948 tons and the phosphorus target slightly smaller, 291 tons.

Elofsson (2012) uses an empirical programming model that includes all Baltic Sea countries and their nutrient reduction measures. Abatement measures for Sweden include increased cleaning at sewage treatment plants, private sewers, P-free detergents, selective catalytic reduction on power plants, ships and trucks, reduction in cattle, pigs and poultry, fertilizer reduction, catch crops, energy forestry, grassland, creation of wetlands, changed spreading time of manure and buffer strips. Some of these measures reduce both phosphorus and nitrogen emissions and others affect only one or the other.

In the 1995-2005 period, national policy has emphasized wastewater treatment measures while nutrient abatement from agriculture has been quite small. The nitrogen emission reduction achieved during that time period was 15,474 tons and the equivalent phosphorus reduction was 527 tons. The total cost of the nutrient abatement measures was 336 million EURO. In the current national policy the cost-effective solution stresses the nutrient reduction potential from agriculture compared to the wastewater, transport and energy sector which have smaller reduction targets. Almost the same emphasis can be seen in the BSAP target for Sweden. The total cost of the current national policy is 299 million EURO and 585 million EURO for the BSAP target.

To reach the Swedish national target cost-efficiently, 138 of the 196 million EURO directed to abatement measures in agriculture should be used to implement measures that reduce both nitrogen and phosphorus emissions simultaneously. Equivalently in the BSAG, 394 of the 404 MEUR directed to agriculture is targeted to simultaneous nitrogen and phosphorus reduction measures. Meeting the BSAP emission reduction targets for Sweden would entail considerably higher total costs compared to the current cost-effective national target. This also means that the BSAP target cannot be met with the current annual budget for nitrogen and phosphorus emission abatement measures in Sweden.

### 5.2.2 Germany

In the German cost-effectiveness study, Mewes (2012) examines a group of agricultural sector abatement measures to see if a reduction target of 25% reduction and an even stricter target of 50% reduction in both nitrogen and phosphorus emissions can be achieved in the German Baltic Sea catchment. A total of 19 river catchments are included in the MONERIS model and a time period 1998–2000 is used as a basis to compare the achieved nutrient reduction to. During this time period the nitrogen nutrient emissions to the German Baltic Sea catchment were estimated to be 13,910 t N $y^{-1}$ and the phosphorus emissions 409 t P $y^{-1}$. A 25% reduction compared to the 1998–2000 nutrient emission level would amount to a decrease of 3,480 t N $y^{-1}$ and 102 t P $y^{-1}$. Respectively a 50% reduction equals 6960 t N $y^{-1}$ and 204 t P $y^{-1}$.

The abatement measures considered in the German CEA case study include advisory service about the use of organic and inorganic fertilizer, extensively used buffer strips coupled with a ban to use fertilizer, grassland buffer strips, restoration of wetlands and converting arable land into either less intensely used arable land, grassland, set-aside land or afforestation with or without use. All the measures affect both nitrogen and phosphorus emissions. The implementation of these measures can mean additional costs to the farmers such as running costs, wetland restoration costs, extension service consultation fees and income loss due to lower crop yield. Cost savings are also possible. Transaction costs and argo-environmental subsidies are left out of the annual abatement cost calculations. The cost-effectiveness of a measure is calculated by dividing the total average cost by the reduced unit of nutrient emissions (river specific retention capacity). The aim is to define a set of measures for each river catchment that enables the reduction target to be reached in the most cost-efficient way.

First Mewes (2012) looks at the nitrogen and phosphorus targets separately. The total cost of meeting the 25% German Baltic Sea catchment nitrogen nutrient reduction target for the cost-effective combination of measures is 9–34 million € $y^{-1}$. The total cost depends on how well the extension service recommendations about fertilizer use are adopted by farmers. A success rate of 50% and 100% for the extension service is considered in the analysis. Respectively the total costs for achieving the 25% reduction target for phosphorus emissions is 12–35 million € $y^{-1}$. The 50% reduction cannot be achieved without new additional measures for either nutrient. Simultaneous reduction of nitrogen and phosphorus was also considered in the case study. If the aim is to reduce 25% of both nitrogen and phosphorus emissions the costs can be slightly higher than for nitrogen emissions alone. The maximum total cost is 103 million € $y^{-1}$ assuming a 50% success rate for extension service efforts and a maximum reduction of 35 % nitrogen and 27 % phosphorus.

### 5.2.3 France

Lacroix *et al.* (2005) use a bio-physical model in their study to evaluate the probabilistic cost-efficiency of different combinations of measures designed to reduce nitrogen concentration from agriculture to the Bruyéres catchment area of the Parisian Basin in the northeast of France. The costs and effectiveness of six farm management practice scenarios were estimated for the study period of 1991–1997. Implementation of three different nitrogen fertilization restrictions, sowing of catch crops and converting fields with the lowest productivity into grassland (set aside) areas were the measures considered in the scenarios. Climatic variability and uncertainty were also addressed in the cost-effectiveness analysis.

The results of the different scenarios were compared to the status quo scenario of conventional farming practices where nitrogen emissions to the Bruyéres catchment area stay unchanged, no catch crops is sowed and 1.5% of the agricultural area is allocated to set aside. The first scenario called the 'integrated fertilization' (Intfert) was the same as the status quo except for the assumed restriction of nitrogen fertilization. The fertilization restriction is defined with optimal yield and environmental goals in mind. In the next two scenarios this restriction is combined with catch crops sown either in August (IntfertC1) or September and (IntfertC2). The next two scenarios (RedinpC1 and C2) are more demanding combining catch crop cultivation with a 20% reduction of fertilizer use compared the optimal level for yield and pollution. In the last scenario (Setas) only the set aside measure is considered. In this scenario 17% of the agricultural area of Bruyéres catchment area is converted to grassland.

The Monte Carlo method was applied to identify the scenario with the optimal environmental and economic impact. The nitrogen concentration reductions achieved with the different scenarios were also compared to the long-term goal defined by the European maximum limit. The best scenario in terms of cost-effectiveness was IntfertC2, in which case catch crops were planted in August. This result also showed how big a difference the timing of the catch crops planting has on nitrogen emissions to the Bruyéres catchment. The use of catch crops was also found to decrease the variability of nitrogen concentration from year to year in the study period, especially so in scenario RedinpC2. None of the above scenarios succeeded in keeping the nitrogen concentration below the European maximum 50 mg $NO_3$ $l^{-1}$ each year of the study period.

### 5.2.4 Austria, Bulgaria, Hungary and Romania

Fröschl *et al.* (2008) present a cost-effectiveness study of reducing nitrogen emissions coming from agriculture to the Black Sea through the Danube River. Four countries were included in the study. Austria was used as an example of an old European Union member country and Bulgaria, Hungary and Romania represented new EU member countries. The objectives discussed in the article were: reduction of fertilizer use by 10% ($M_1$), reduction of ammonia emissions from manure by 25% ($M_2$), increase plant productivity in Austria by 10% and 20% in other countries ($M_3$), a reduction of erosion by 75% and surface run-off by 20% ($M_4$). The goal was to achieve the above reduction objectives with a combination of best available technique measures by 2015.

Effects to the farmer income include costs of implementing a measure and the possible change in the revenues. Subsidies are not included in the calculation. A linear optimization model was used to find the optimum measure combination that would minimize the total costs of achieving a given nitrogen emission reduction. The effectiveness of a specific measure in reducing the nitrogen emission was estimated using a MONERIS model.

Fröschl *et al.* (2008) consider two CEA-situations: a situation where countries act on their own without international cooperation and a situation where countries cooperate in meeting the nitrogen emission reduction objective. In the first case the most profitable measure combination to implement for the farmers in Austria and Hungary are improving plant production techniques ($M_3$) and the second best is reduction of erosion and surface run-off ($M_4$). For Bulgaria and Romania the situation is reversed, the most profitable measure combination is $M_4$ and the second best is $M_3$. To use all of the different measure combinations ($M_1$–$M_4$) at the same time would cause the largest costs to the farmers. The maximum nitrogen emission reduction of all four countries is 18,000 t/a with a total cost of 1,010 m€/a. A total of 54% of this maximum reduction would be implemented in Romania with a share of 37% of the implementation costs.

In the second case international cooperation with or without transfer payments between countries was illustrated. A transfer payment would enable the countries with the lowest implementation costs to implement a larger share of measures. A reduction target of 13,500 t/a was considered. If transfer payments are possible the cost-effective solution would be the following: Austria would implement measure combination $M_3$ to the full extent, Bulgaria and Romania would implement $M_1$, $M_3$ and $M_4$ to the full extent and Hungary would fully implement measure combination $M_3$ and $M_4$ only partially. The total cost is negative (-90 m€) and the transfer payment to Bulgaria would be 21 m€ and 11 m€ to Romania. If transfer payments between countries with lower and higher implementation costs would not be possible the total cost of a 13,500 t/a nitrogen emission reduction objective would yield a positive total cost of +137 m€. Fröschl *et al.* (2008) conclude that international cooperation in nitrogen emission reduction is more cost-effective than unilateral action.

### 5.2.5 Spain

The Spanish cost-effectiveness study (Berbel *et al.,* 2011) focuses on water-saving measures to reach good ecological status defined by the EU Water Framework Directive (WFD) for the Guadalquivir River Basin. The different measures discussed in the article are: improvement of urban distribution networks, modernization of irrigation systems, service cost recovery in both urban sector and irrigation, volumetric billing for irrigation, extension services for irrigators and strict groundwater abstraction control. The study highlights the differences of using either pressure reduction or impact reduction as a measure of effectiveness. Comparative statistics is used to test how sensitive the measures are to uncertainty about the cost and effectiveness level. Both of the factors mentioned above can change the priority in which the measures are implemented in order to achieve the environmental goal in the most cost-effective way.

The water-saving measures are ranked using a cost-effectiveness index. The annual costs of a specific measure can include investment, maintenance and operational costs. The use of impact reduction as a measure of effectiveness is recommended in the EU WFD. This means that the CE index of a measure is calculated dividing the annual cost by the change in the water flow in the river. The strict ground water abstraction control has the lowest cost-effectiveness index and is therefore the most cost-efficient measure. The second and third measures to be implemented are service cost recovery for irrigation and volumetric billing for irrigation. All three measures have also a low CE pressure reduction index. The impact and pressure reduction index give conflicting information about the priority of the use of the following measures: improvement of urban distribution networks, modernization of irrigation systems and extension services for irrigators. For these measures the CE pressure reduction index is low and CE impact reduction index is high.

In the sensitivity analysis ‘optimistic’ and ‘pessimistic’ cost and effectiveness values were estimated to account for the uncertainty related to the analysis. This altered the ranking of some of the measures. The biggest difference in the priority of implementation can be seen for the extension service for irrigators. In the optimistic scenario extension service is the second measure to be implemented but in the most probable and pessimistic scenario it's sixth. One of the key findings of the study is that the aim of reducing water consumption by 202.13 $Mm^3$ per year cannot be achieved with the suggested program of measures (PoM). Additional more drastic measures are required to meet this goal.

Other analysis of measures for implementing WFD’s programmes of measures in Spain include the cost-benefit analysis of the environmental restoration of a coastal lagoon (Martinez-Paz *et al.,* 2012) and the cost-benefit analysis of reclaimed waste water for agricultural use (Alcon *et al.,* 2012)

## 6 Conclusions

Within the economic analysis required for the delivery of the WFD objectives, the role of CEA has been given a particularly important role, as a prerequisite for the development of river basin management plans. Regarding the measurement of costs of environmental measures, an important issue is associated with data availability which would enable research efforts to account for not only private financial compliance costs but also for economic/social costs that are usually very difficult to monetize. Key issues, which have to be addressed in measuring effectiveness, include the basis on which effectiveness is assessed (e.g. 'pressures' or 'impacts') or the assignment of proper weights to individual pressures in cases of multi-pressure effects of different measures.

Various methodological approaches have been proposed in applying CEA to evaluate environmental measures and policies (e.g. optimization techniques, bio-economic modeling, regression analysis and damage index methods). The most commonly used approach in applying CEA involves linear optimization models based on mathematical programming framework. Main criticisms of this approach refer to its inability to control for farm-specific characteristics and non-linear relationships; and to account for uncertainty. To address non-linearity, alternative non-linear optimization models have been proposed, while uncertainty has been explicitly accounted for in chance constrained programming models. In general, the choice of a particular methodological framework in the CEA study highly depends on the specific environmental problem to be dealt with, the availability and credibility of data, and the degree of uncertainty inherent in cost and effectiveness information.

In CEA uncertainty can be inherent in the estimates of costs, effectiveness and time-lagged effects of measures and thus, can considerably affect the ranking of measures under consideration. To deal with this issue, the use of intervals of costs and effectiveness estimates as well as sensitivity and scenario analysis is advocated. Also, stochastic programming and Bayesian Belief Networks (BBN) can be applied to investigate water management decision-making under uncertainty. In a CEA context, the potential impacts of climate change on costs and effectiveness of water management measures can be significant. Climate change can create new and/or affect existing pressures on water bodies, and also affect the effectiveness of long-term water management measures. Also, indirect impacts of climate change arising from land use changes could modify patterns of diffuse pollution and require new abatement-strategies. Importantly, the above-mentioned issues justify the "climate-proofing" investigation of proposed water management measures. In this context, the scale at which climate change impacts are assessed and the level (i.e. international, national or more local) at which CEA is undertaken are critical issues that may significantly affect cost and effectiveness estimates. To account for climate change in CEA-based decision-making, several approaches have been proposed, including simple sensitivity analysis, bio-physical models including climatic variables in assessing probabilistic cost-effectiveness and meta-models based on BBN.

Several studies have been carried out in Europe dealing with CEA applications related to WFD. In the case of the five countries specific to REFRESH demonstration catchments, one can observe a rich research tradition in the UK (including Scotland), Norway and Finland in conducting such studies and have made significant progress in complying with the WFD requirements. This tradition seems much weaker in the case of the Czech Republic where such studies have been introduced in the context of the implementation of the WFD, and especially of Greece, where CEA applications are fragmented and progress in the implementation of the WFD has shown a considerable lag. In the case of other European countries the review has shown a notable number of CEA studies dealing with agricultural abatement measures in Sweden, Germany and France and an impressive CEA study of reducing nitrogen emissions from agriculture in the Danube (i.e. specific to Austria, Bulgaria, Hungary and Romania). Finally, in Spain, CEA efforts seem to focus more on water-saving measures and deal with both urban and rural activities.

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