Ecosystem Services

In Nordic Freshwater Management

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TemaNord 2014:561

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ISBN 978-92-893-3851-6 (PRINT)
ISBN 978-92-893-3853-0 (PDF)
ISBN 978-92-893-3852-3 (EPUB)
http://dx.doi.org/10.6027/TN2014-561

TemaNord 2014:561
ISSN 0908-6692

Layout: Hanne Lebech
Cover photo: SignElements

Print: Rosendahls-Schultz Grafisk

Printed in Denmark

This publication has been published with financial support by the Nordic Council of Ministers. However, the contents of this publication do not necessarily reflect the views, policies or recommendations of the Nordic Council of Ministers.

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Content

Foreword

This report has been commissioned by the Nordic working group for environment and economy in collaboration with the Nordic working group for terrestrial ecosystems. The aim of the report is to explore the use and usefulness of the ecosystem services framework in freshwater management in Nordic countries, addressing the following four topics:

- Ways and methods for using ecosystem services in assessing the benefits of ecological improvements in water courses.
- Ways and methods for assessing costs, particularly disproportionate costs in line with the water framework directive.
- How the ecosystem services framework might contribute to develop targeted and locally adapted instrument mixes at the level of each river basin or water region.
- Possible use of payment for ecosystem services as an instrument for targeted freshwater management.

The structure of the report reflects these central themes. The report is a follow-up of a report from 2012 on ecosystem services in Nordic watersheds (Valuation of Ecosystem Services from Nordic Watersheds, NCM 2012). In that report the emphasis was on describing and mapping the ecosystem services provided by different ecosystems on a more general level. The aim of this report is to provide a more policy oriented approach by exploring how the ecosystem services concept can be applied. Management of ecosystem services has lately been among the corner stones in Nordic activities aiming at enhancing green economy.

The report has been written by the Norwegian consultancy Vista Analyse. The core team responsible for the report consisted of Kristin Magnussen (project leader), Berit Hasler (Aarhus University), and Marianne Zandersen (Aarhus University). Comments and guidance on the report have been provided by the two Nordic working groups. The interim results of the project were presented and discussed at a Nordic seminar in Mariehamn in March 2014. The authors of the report are however responsible for the content of the report which does not necessarily reflect the views and positions of the governments in the Nordic countries.

The main contribution of this report is to focus on practical issues and provide examples on how the ecosystem services framework has been used in this respect, mainly in a Nordic context. The report does not provide a complete overview of Nordic studies of ecosystem services, or valuation of ecosystem services, as such an overview has been given before. Examples have been chosen in order to demonstrate usage of the ES framework in different countries and with different purposes, in the hope that they may inspire and potentially be useful for others.

The report reveals that there are several practical examples of use of the ecosystem services framework in water framework directive related studies in all the Nordic countries. Most of the examples involve listing, description and categorization of freshwater ecosystem services, while there are few comprehensive cost benefit analyses and analyses of disproportionate costs that apply this framework. Relatively few studies in the Nordic countries value ecosystem services per se, while there are some more which value improved water environment, including reaching good ecological status.

The examples provided illustrate that the ecosystem services framework is used increasingly in Nordic aquatic management. More knowledge about ecosystem services and the value of ecosystem services for freshwater systems is however needed. Despite the scarcity of empirical studies, the examples and the discussion in this report demonstrate that the ecosystem services framework may be useful in Nordic water resource management, including in the implementation of the water framework directive.

October 2014

Magnus Cederlöf

Magnus Cederlöf
Chairman of the Working Group on Environment and Economy under the Nordic Council of Ministers

Acknowledgement

Kristin Magnussen (Vista Analyse), Berit Hasler (Danish Centre for Environment and Energy (DCE)/Department of Environmental Science, Aarhus University (AU)), and Marianne Zandersen (Danish Centre for Environment and Energy (DCE)/Department of Environmental Science, Aarhus University (AU) have written the report. Henrik Lindhjem (Vista Analysis and Norwegian Institute for Nature Research) has reviewed the report.

We would like to thank the members of our advisory group for their contribution, particularly in providing us with case study examples. The members of this group were:

- Anni Huthala, Government Institute for Economic Research, Finland.
- Virpi Lehtoranta, Finnish Environment Institute, Finland.
- Bjørn Walseng, Norwegian Institute for Nature Research, Norway.
- Ann Kristin Lien Schartau , Norwegian Institute for Nature Research, Norway.
- David Barton, Norwegian Institute for Nature Research, Norway.
- Simon Haraldsen, fylkesmannen in Oslo and Akershus, Norway.

We would like to thank Virpi Lehtoranta in particular, for writing text and providing figures for the two recent Finnish examples on valuation of reaching good ecological status (chapter 3.3.4.).

Any errors or omissions remain the responsibility of the authors.

8th September 2014

Kristin Magnussen
Project manager
Vista Analyse AS

List of abbreviations

AEP	Agri-Environmental Policy
CAP	Common Agricultural Policy
CBA	Cost Benefit Analysis
CEA	Cost-effective Analysis
CICES	Common International Classification of Ecosystem Services
DG	Directorate General
EAFRD	European Agricultural Fund for Rural Development
EFAs	Ecological Focus Areas
EQR	Environmental Quality Ratio
ES	Ecosystem Services
ETS	Emission Trading System
EU	European Union
GAEC	Good Agricultural and Environmental Condition
GEP	Good Ecological Potential
HMWB	Heavily Modified Water Body
GES	Good Ecological Status
MA	Millennium Ecosystem Assessment
MAES	Mapping and Assessment of Ecosystems and their Services
NWRM	Natural Water Retention Measures
PES	Payment for Ecosystem Services
PoMs	Programme of Measures
RBMP	River Basin Management Plan
TEEB	The Economics of Ecosystems and Biodiversity
WFD	Water Framework Directive

Summary

Abstract

Ecosystem Services (ES) are the contributions which ecosystems make to human well-being. Ecosystem services can be mapped and assessed consistently within an ES framework, building on the understanding of the link between ecosystems and human well-being. This project aims to explore the use and usefulness of the ES framework in freshwater management, particularly water management according to the Water Framework Directive (WFD) in the Nordic countries by providing examples. The examples provided in this report illustrate that the ES framework is used increasingly in Nordic aquatic management, but that relatively few studies in the Nordic countries value ecosystem services per se, while more value improved aquatic environment, including reaching good ecological status, according to the WFD. There are several examples of studies using various techniques to value ecosystem services related to the WFD in all the Nordic countries. Most of the examples involve listing, description and categorization of freshwater ecosystem services, while there are few comprehensive Cost Benefit Analyses and analyses of disproportionate costs that apply this framework.. There are several projects that study targeted and locally adapted instruments in the Nordic countries, mainly in the agricultural sector, and targeted and locally adapted instruments are increasingly used for ES management. Local adaption and use of the ES framework is emphasized, however, the link between improved ES flows and the economic mechanisms and size of payments is often indirect. More knowledge about ES and the value of ES for freshwater system management is still needed. The examples and the discussion in this report demonstrate that the ES framework may be useful in Nordic water resource management, including in the implementation of the WFD.

Background and motivation

Ecosystem Services (ES) are the contributions that ecosystems make to human well-being. By different classification schemes ecosystem services can be mapped and assessed consistently within an ES framework, building on the understanding of the link between ecosystems and human well-being.

In the project VALUESHEDS ("Valuation of Ecosystem Services from Nordic Watersheds" by Barton *et al*. 2012) and several other projects concerning ecosystem services in the Nordic countries, emphasis has been on describing and mapping the ecosystem services provided by different ecosystems. Now there is a need to further explore how to apply the concept and valuation of ecosystem services in practical water resources management.

The ES framework is not a part of the Water Framework Directive (WFD). When discussing ecosystem services in freshwater systems, however, it may be appropriate to relate to the WFD, which is one of the key pillars of water management in all the Nordic countries. Hence, a useful next step for considering ecosystem services in freshwater is to explore what the role of the ES framework may be for different water management tasks in general, and more specifically according to the WFD.

Project goals

This project aims to explore the use and usefulness of the ES framework in freshwater management in the Nordic countries, addressing four related topics in particular:

- Ways and methods for using the ES framework in assessing the benefits of ecological improvements in water courses.
- Ways and methods for assessing costs, particularly what the WFD calls disproportionate costs, of improvement measures.
- How the ES framework might contribute to developing targeted and locally adapted instrument mixes at the level of each river basin/water region.
- Possible use of Payment for Ecosystem Services (PES) as an instrument for targeted freshwater management.

Our approach

These four key topics, being the foci of this report, have to some extent been described and discussed before in a WFD context. The main contribution of this report is to provide examples on how the ES framework has been used in a Nordic WFD-context. While the VALUESHEDS report (Barton *et al*. 2012) mainly discussed basic methodological and principal issues, this report will focus on the practical issues and provide examples. We will not provide a complete overview of Nordic studies of ecosystem services, or valuation of ecosystem services, as such an overview was given in Barton *et al*. (2012). The approach in this report is to pick examples in order to demonstrate uses in different countries and with different purposes, hoping that they may inspire and potentially be useful for managers in the Nordic countries.

Ecosystem Services, Payment for Ecosystem Services and the Water Framework Directive

The ES framework has received much attention and substantial work is currently underway to develop this framework further and to implement it in practical management. The ES framework can be used to map and measure the value of the changes in supporting, provisioning, regulating and cultural services, and the trade-offs between these.

The Water Framework Directive (WFD) is the main Directive regulating the quality and the use of freshwater as well as coastal waters in the EU-countries, and Norway and Iceland have adopted the requirement in the Directive as well.

The aim of the WFD is to maintain and improve the aquatic environment, with specific emphasis on the ecological and physical-chemical quality of the water bodies concerned in order to obtain good ecological status (GES), and good ecological potential for those water bodies that are classified as modified. The main areas where economic analysis in the WFD can be linked to the ES framework are the required river basin characterization in the WFD (Article 5), the use of water pricing and cost recovery (Article 9), the assessment of disproportionate costs (Article 4), and finally the requirement for identification and implementation of cost-effective combinations of measures to achieve good ecological status of water bodies as a part of the Program of Measures (PoMs) (Article 11).

Water services are defined as part of the WFDs article 2(38) ("Definitions"):

> "Water services means all services which provide, for households, public institutions or any economic activity: (a) abstraction, impoundment, storage, treatment and distribution of surface water or groundwater, (b) waste-water collection and treatment facilities which subsequently discharge into surface water."
>
> EU commission, 2000

It is clear that the ES concept and framework has a broader definition of services than the WFD. Still, the ES framework can be used in analyses which are part of the implementation of the WFD.

The primary suggestion from this report is that the use of the ES framework can be very helpful to assess and illustrate how goods and services are affected by implementation of the WFD, and the trade-offs between different goods and services. In particular, it can illuminate how different water policy implementation strategies might lead to different results for the provision of ecosystem services, and hence demonstrate differences between the total benefits of different implementation strategies and the distribution of benefits between different users or beneficiaries across space and time.

The ES framework offers a more thorough way to assess benefits of positive environmental changes in a complex ecological system. It can help improve the evaluation methodology of disproportionate costs in the WFD. Furthermore, the ES services framework can be used to assist the analysis of the Programme of Measures and the cost-effectiveness of the measures.

The ES framework is one of the cornerstones in a number of economic policy instruments relating to water pollution, comprising both voluntary and mandatory instruments. The voluntary policy instrument PES is based on a payment made for the delivery of ecosystem service(s). Water quality cap-and-trading is an example of a mandatory regulatory instrument which is also based on the ecosystem services concept, where ecosystem based quotas for e.g. nutrient loads are traded between polluters. PES schemes that target water quality pollution are already in use in the Nordic countries and Europe. These PES schemes are not initiated because of the WFD, but are typically firmly established in the Common Agricultural Policy (CAP) of the EU, in drinking water policies (targeted drinking water protection) and aquifer replenishment. Nevertheless, these economic policy instruments contribute significantly to meeting the obligations under the WFD and may have the potential to play a larger role for the WFD than they do today. Common for policy instruments aiming at improving water quality is the growing recognition that they need to be adapted to local conditions, since both costs and benefits (ecosystem services) differ substantially between areas.

Use of the ES framework to describe and value benefits of improved ecological status in water

The necessary steps for benefit assessment of water status improvements based on the ES framework are identification, quantification and valuation. Identification of ecosystem services can be done, and is done, on different scales (water body, river basin, country, region) depending on the purpose. In some studies the identification and valuation is carried out with focus on one or a few selected ecosystem services. In a WFD context the most interesting question is how the benefits from all ecosystem services are changed (enhanced) when reaching the goal of good ecological status.

The included examples show that it is demanding to identify, and particularly to quantify and, when relevant, value in monetary terms the benefits of reaching good ecological status.

There are many interesting examples of the use of the ES framework in order to identify, quantify and value the benefits provided by freshwater in general, and the improvement of freshwater conditions (ecological and chemical status in WFD terms) in particular, across the Nordic countries. Most of the studies and reports so far do not, or only to a minor extent, take into account the need to consider trade-offs, or double counting. In the ecosystem services literature there is an on-going discussion of these issues. Probably, the issues of concern will be taken more into account as the framework is more commonly applied.

The ES framework can be a tool for systematic identification of benefits and to investigate the connection between ecological changes and welfare gains, and the examples show that the framework is coming into use across the Nordic countries. However, this framework is clearly no "quick fix". Much work is still needed on all aspects of identifying, quantifying, mapping and not at least valuing) ecosystem services (by monetary and non-monetary approaches), both with respect to the ecological underpinnings and the economic methodology.

Assessment of disproportionate costs

There are relatively few examples of Cost Benefit Analysis (CBA) in the context of the Water Framework Directive, and even fewer where the ES framework is used for benefit assessment. This is the case in Europe, as well as in the Nordic countries.

Martin-Ortega (2012) in her paper on economic perspectives and policy applications in the implementation of the WFD concludes that "… while CEA [Cost Effectiveness Analysis; authors note] has been widely adopted by most national guidelines in Europe, and the estimation of the environmental benefits has received a significant attention from the literature, the way these two should be joined up in a CBA has received much less attention".

We could add that even if the benefits are estimated, the ES framework is not commonly used. For example, some studies value "good ecological status", which is the aim of the WFD. Still, it can be difficult to retrieve information about the value of the specific ecosystem services, like recreation, fisheries and fish habitats etc., from these studies.

There are some examples though, mainly used as screening procedures, on national, regional and local (water body) levels, where the ES framework is used in evaluation of disproportionate costs. This is exemplified in Jensen *et al*. (2013) who use information on the values of the ecosystem services included in the Aquamoney study, i.e. the economic valuation results of water quality and ecological improvements in Odense river basin, in a benefit transfer[1] to other Danish water bodies. The benefit transfer results by river basins are subsequently used in a cost-benefit analysis for the WFD implementation in Denmark. The CBA is used as a conservative screening of where costs appear to be disproportionate, i.e. exceed the benefits provided by these ecosystem improvements. Much of the same procedure and framework is used on the local water body scale in two rivers in urban Oslo as a screening procedure to evaluate benefits and potentially disproportional costs (Magnussen *et al*. 2014).

The ES framework is seen as useful, because it helps provide a systematic and comprehensive picture of all benefits (valued in monetary terms, quantified or qualitatively described) which is necessary to assess benefits of the improvements in water status. The conclusion in

[1] Transfer of benefit estimates from one location where a valuation study has been carried out to another place of study where no valuation study exists.

Jensen *et al*. (2013) is, however, that a more comprehensive application of the ES framework should include more services into the assessment of those areas where the screening indicates that the costs exceed the benefits, because not all affected ecosystem services were valued in the primary study. This is an area where more work is needed and probably will be carried out in the coming years.

Locally adapted instruments, including PES, for enhanced provision of ecosystem services

There are a number of examples and lessons of locally adapted or targeted policy instruments that contribute to meeting WFD objectives and targets. Some of the examples are applied in practice and show results whereas other examples represent trends, recommendations, pilot studies or on-going research. PES schemes vary in the degree to which they are locally adapted to the circumstances and characteristics of land owners and/or physical and biological conditions of catchment areas.

Mixed instruments are frequently used in the Nordic countries (for example in agriculture), however, most of the mixed instruments used are general and not locally adapted. There is therefore a substantial potential for more targeted adaption, differentiated to local conditions for example creating or restoring wetlands. The examples we present focus on market-based policies and frameworks for managing non-point pollution from land use (primarily) in agriculture because associated problems and examples are found relevant in the Nordic context.

Non-point pollution is difficult to control in practice, in particular when using uniform instruments that ignore differences in soil retention capacities, farm typologies and costs as well as farmer characteristics. This so-called wicked problem requires a mix of instruments and measures that are adapted to local conditions as well as the involvement of a mix of stakeholders. The three examples of comprehensive water quality management programmes at watershed levels from Morsa in Norway, Munich in Germany and Catskill Mountains in the State of New York, USA, represent programmes that appear to produce significant and positive results for water quality within relatively few years using the ES framework and to a large extent PES. The motivation behind the Catskill Mountains case described in literature has been contested, however. Common for the programmes is locally adapted measures and instruments, some voluntary and others mandatory; an appropriate mix of different policies and the active involvement and engagement of land owners and households.

The idea of developing locally adapted PES instruments at the catchment level was also part of pilot projects in Denmark to look at how farmers could enter into contracts with towns and cities to provide ecosystem services on their land that would regulate excess water and avoid inundations in the built environment. It is also used in a proposed regulatory approach for targeted regulation of nutrient reductions in Denmark, where the nutrient management will be differentiated according to the resilience of the agricultural soils, the retention capacity (i.e. the regulating ecosystem service) and the effect on the ecosystem services of the water body (Kjær, 2014). Wetland PES schemes, which have a direct relevance to the WFD, are found in the three Nordic EU member state countries, co-financed through the second Pillar of the EU Common Agricultural Policy (CAP). Whereas the measure and objectives are largely similar across the countries, the payment levels and conditions in the contracts differ.

Water quality trading does not currently exist in the Nordic countries or in the EU, but could in principle be established as a measure at the river basin level as a cost-effective way of reducing emissions. The EU Commission proposed in the Communication "A Blueprint to safeguard Europe's water resources"[2] to develop Common Implementation Strategies (CIS) Guidance on trading schemes by 2014. Another example, outside the EU, include the nitrogen sourcing and trading in the lake Taupo catchment in New Zealand that aims at maintaining current good water quality, at risk from intensified agriculture and expanding urban areas. According to Stanton *et al*. (2010) there are currently 66 water quality trading programmes in the US, four in Australia and one in each of New Zealand and Canada. Voluntary off-sets of nutrient loads to recipients have been attempted in Sweden, and a full-scale pilot in Denmark has recently been carried out, indicating that compensatory mussel farming can be both an environmentally and economically efficient and effective measure.

Generally, when targeting economic policy instrument to catchment or even sub-catchment levels the challenge becomes striking the right balance between policies and measures that make sense locally while keeping transaction costs down in relation to management, coordination and control of both measures and policies.

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[2] (COM (2012) 673).

Conclusion

There are several examples of the use of the ES framework in WFD-related studies in all the Nordic countries. Most of the examples involve identification/listing, description and categorization of freshwater ecosystem services, while there are relatively few comprehensive CBAs and analyses of disproportionate costs that use this framework.

Relatively few studies in the Nordic countries value ecosystem services per se, while there are some more that value improved water environment, including reaching good ecological status. Apart from the Aquamoney study described in VALUESHEDS (the Morsa and Odense studies) there are a couple of new Finnish studies that value improved fresh water status according to the objectives of the WFD. These do not use the ES framework per se, but the improvement in water status can be linked to affected ecosystem services. Benefit transfer is, when performed, frequently used to value improved water status, and there exist a number of examples that transfer benefits within Denmark, from Denmark and Norway to Sweden, from one river in Oslo to other rivers in Oslo etc. However, there is a shortage of relevant primary studies to transfer from, and particularly there is a lack of good primary valuation studies which use the change in water status as their point of departure to elicit which ecosystem services are affected and to what extent.

Several studies, pilot projects and full scale projects use targeted and locally adapted instruments in the Nordic countries, mainly applied to the agricultural sector. In many of the studies the use of the ES framework is emphasized. However, the direct link between improved ecosystem services, the economic mechanisms and size of payment may not be so direct. One will need to know even more about the ecosystem services and the value of ecosystem services in order to target these instruments further. Still, there is a growing awareness that water pollution instruments need to be locally adapted and that the ES framework can be of great use.

It is perhaps not surprising that it takes some time to incorporate the ES framework in actual management of fresh water resources, and that the more economic parts of the framework, valuation in monetary terms and uses in CBA, take more time than the rest. The notion of ecosystem services has been around for a while, however it was not until the TEEB project was launched from 2008 and onwards that the foundation for the more economic and practical uses of the framework was developed. It does take time to integrate new ways of thinking into public resource management. However, much has been done, and there is much ongoing work in this field in the Nordic countries, as the examples in this report fully illustrate.

1. Introduction

1.1 Background and motivation

With ecosystem services (ES) we mean the benefits – goods and services – we receive from ecosystems. Water ecosystems provide for example drinking water and nutrition in the form of fish and shellfish, and they provide basis for recreation like swimming and angling.

In the project VALUESHEDS ("Valuation of Ecosystem Services from Nordic Watersheds" by Barton, Lindhjem, Magnussen and Holen 2012) and several other projects concerning ecosystem services (ES) in the Nordic countries, emphasis has been on describing and mapping the ecosystem services different ecosystems provide (e.g. watersheds in VALUESHEDS; freshwater ecosystem services in Maes *et al*. 2012; or all ecosystems in the Nordic TEEB[3] (Kettunen *et al*. 2013) and the TEEBS for separate countries (NOU 2013:10 for Norway, SOU 2013:68 for Sweden, the ongoing Finnish and Danish processes). This work is important and necessary as a starting point for describing and demonstrating the values associated with different ecosystems.

Currently there is a need to further explore the question of how to integrate and use lessons from work on the concept and valuation of ecosystem services in practical management, and how to integrate this in an overall framework of ecosystem management, e.g. related to the implementation of the European Water Framework Directive (WFD). For water management all the Nordic countries are currently implementing the WFD, as this EU directive is also made part of the European Economic Agreement for Norway and Iceland. The ES Framework[4] is not mentioned in the WFD, but when discussing ecosystem services in freshwater, however, it may be appropriate to relate to the WFD. Hence, a useful

[3] TEEB – The Economics of Ecosystems and Biodiversity is described in chapter 2.1.

[4] With the ES Framework we mean an analytical framework where "Ecosystem Services are derived from ecosystem functions and represent the realized flow of services for which there is demand. For the purpose of this framework, ecosystem services also encompass the goods derived from ecosystems. People benefit from ecosystems (goods and services). These benefits are, among others, nutrition, access to clean air and water, health, safety, and enjoyment and they affect (increase) human wellbeing which is the key target of managing the socio-economic systems" (COM 2013: Mapping and Assessment of Ecosystems and their Services).

next step for considering ecosystem services in freshwater seems to be to explore more in depth what the role of the ecosystem services framework may be for water quality management and administration, especially connected to the requirements for economic information connected to the implementation of the WFD, and this is one of the main purposes of this project.

1.2 Project goals

This project aims to explore the use and usefulness of the ecosystem services framework in freshwater management in the Nordic countries, addressing four related topics in particular:

- ways and methods for using ecosystem services in assessing the benefits of ecological improvements in water courses
- ways and methods for assessing costs, particularly what the WFD calls disproportionate costs of improvement measures
- how the ecosystem services framework might contribute to develop targeted and locally adapted instrument mixes at the level of each river basin/water region
- possible use of Payment for Ecosystem Services (PES) as an instrument for targeted freshwater management.

1.3 Our approach and outline of the report

The key topics of this report; benefits of improved freshwater quality, disproportionate costs, targeted and local instrument mix and payment for ecosystem services have to some extent been described and discussed before in a Water Framework Directive (WFD) context. The main contribution of this report is to explore how the ecosystem services framework may be used in this respect and mainly in a Nordic context. While the VALUESHEDS report mainly discussed basic methodological and principal issues, this report focuses on the practical issues and provides examples. Examples from different Nordic countries, and different uses, will be the main contribution of this report. We will not provide a complete overview of studies of ecosystem services, or valuation of ecosystem services, as such an overview was given in Barton *et al.* (2012). We have picked examples in order to demonstrate uses in different countries and with different purposes, hoping that they may inspire and potentially be useful in others' water management work.

Chapter 2 gives an introduction to the WFD and the ecosystem services framework, and some of the main tasks in WFD where economic benefits and costs need to be assessed, and where we believe the ecosystem services framework may be of added value for this assessment. We will also introduce the payment for ecosystem services (PES) framework and locally adapted measures/instruments and how the ecosystem services framework can be helpful in this respect. The main part of the report will present and discuss examples in order to illustrate how the ecosystem services framework can be used to describe and value improved environmental status in fresh water, and discusses topics which are important for how this can be done (chapter 3). Chapter 4 in a similar way provides examples of how disproportionate costs may be assessed using an ES framework and chapter 5 present examples of PES and locally adapted measures/instruments.

Summary and final conclusions are presented in the summary and conclusions chapter in English at the beginning of the report, and in Norwegian at the end of the report.

2. Introduction to Ecosystem Services, Payment for Ecosystem Services and implementation of the Water Framework Directive

In this chapter we:

- Present background for the presentations and discussions of examples in the following chapters.
- Introduce the ES framework and describes the use of this framework in the Nordic countries (section 2.1).
- Give an overview and example of classification of ecosystem services in freshwater (section 2.2).
- Discuss the potential links between the Water Framework Directive and the ecosystem services framework, where specific emphasis is given to how the ecosystem services concepts and framework can be used to support the main economic tasks in the water management policies, and how the ecosystem services framework might be helpful in situations where economic benefits and costs need to be assessed (section 2.3).
- Discuss how the ecosystem services framework might be linked to economic instruments for locally targeted measures (PES and water quality trading) (section 2.4).
- Discuss and conclude regarding findings and what we can learn from this chapter (section 2.5).

2.1 Introduction to the ES framework and the use of this framework in EU and the Nordic Countries

The term ecosystem services has been used since the early 1980s (see e.g. Ehrlich and Money, 1983; Erlich and Ehrlich 1987) to describe the relationship between nature (ecosystems) and goods and services that people appreciate and which are essential for our continued well-being and welfare (NOU 2013). The term had a revival in the Millennium Ecosystem Assessment (MA 2005) where the concept is central, and since then the term has been in much use. The Economics of Ecosystems and Biodiversity (TEEB) project has further spread the ecosystem services framework to a broader use during the last few years. TEEB emphasises the importance of asking questions like "which ecosystem services are central to my local/regional society and economy? Who depends on these services? Which services are at risk? How will a policy action affect these services?" (TEEB 2012, p. 5).

These questions are also important in the context of Nordic freshwater ecosystems, with numerous different users, services and policies influencing the quality and use of them – on the one hand the Water Framework Directive and national water policies, and on the other hand the Common Agricultural Policy (the CAP) and national agricultural policies. We will take these potential conflicts into consideration in the analysis of the provision and management of freshwater ecosystem services in the forthcoming chapters in this report. For further general and basic description and definition of ecosystem services we refer to the MA and TEEB publications (see e.g. Millennium Ecosystem Assessment 2005; TEEB 2012), and for a general description of ecosystem services in a watershed framework in a Nordic context we refer to the "VALUESHEDS report" (Barton *et al*., 2012).

Further refinement of the relationship between ecosystems and the socio-economic systems has been carried out for instance as part of the Analytical Framework for Mapping and Assessment of Ecosystems and their Services developed under MAES (Mapping and Assessment of Ecosystems and their Services (COM 2013, further developed in COM 2014.)

The MAES group defines the ES framework as an analytical framework where:

> "Ecosystem functions are defined as the capacity or the potential to deliver ecosystem services. Ecosystem services are, in turn, derived from ecosystem functions and represent the realized flow of services for which there is demand. For the purpose of this framework, ecosystem services also encompass the goods derived from ecosystems18. People benefit from ecosystem (goods

and) services. These benefits are, among others, nutrition, access to clean air and water, health, safety, and enjoyment and they affect (increase) human wellbeing which is the key target of managing the socio-economic systems."

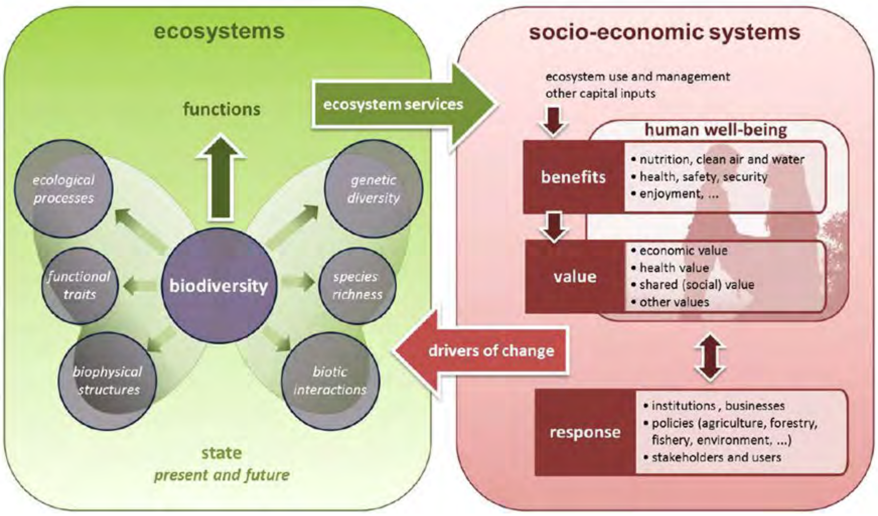
European Union, 2013, p. 16

This is also how we will use the term "ES framework" in this report.

MAES' framework figure for ecosystem services is used here as an illustration of the relationship between ecosystems and their functions and the ecosystem services these ecosystems provide for the socio-economic systems (European Union, 2013).

Important to notice in the figure is that the ecosystems provide services to the socio-economic system. But it is use and management that change these ecosystem services into benefits for people and contribute to human well-being and welfare. Another important issue to note from the figure is that most often capital inputs and labour are needed in addition to the ecosystem services in order to make the ecosystem services useful for us. What we aim at valuing in this system are the benefits we receive, not the ecosystem services themselves. It is also noteworthy that the socio-economic systems in term influence the ecosystems.

Figure 2.1: MAES' framework for ecosystem services[5]



[5] http://biodiversity.europa.eu/maes

2.1.1 *Use of the ecosystem services framework in the Nordic Countries*

The TEEB project in particular has launched a considerable amount or work in many countries related to assessment of ecosystem services in countries, in regions, from specific ecosystems etc. Kettunen *et al.* (2013) surveyed Nordic ecosystem services, including ecosystem services from freshwater and suggest ways of doing this on this scale. Examples from this assessment as well as for the national assessments described below will be presented in the following chapters of the report.

A *Finnish study*, "TEEB Finland – National Assessment of the Economics of Ecosystem Services in Finland" has been launched in 2014 with the aim to "improve the knowledge and understanding of the concepts of ecosystem services, as well as the ways different benefits provided by ecosystems – including the underpinning functions of these benefits – can be measured and valued."[6] The description of the TEEB Finland emphasizes the need to expand the attention of different land-use related ecosystem services beyond the provisioning services. The study aims to identify key ecosystem services, methods to assess quality and economic importance, and to make them useful for national and local management and governance. The work measuring the economic importance will however be at a preliminary level. TEEB Finland also aims to support a number of ongoing national and regional policy processes, e.g. the development of a national framework for assessing and monitoring ecosystem services and developing indicators (e.g. the FESSI project producing national ecosystem service indicators); the development of green economy, sustainable energy production and consumption etc. by the use of national policies and policy instruments. The final results of TEEB Finland are foreseen to be published at the end of 2014.

The Norwegian study: The Norwegian government appointed an expert commission in October 2011 "to assess and study the value of ecosystem services." The Commission was asked, among other things, to describe the consequences for society of the degradation of ecosystem services, to identify how relevant knowledge can best be communicated to decision makers, and to make recommendations about how greater consideration can be given to ecosystem services in private and public decision making. On 29th August 2013, the Commission submitted its recommendations to the Minister of the Environment in the form of a

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[6] http://www.es-partnership.org/esp/81104/9/0/50

Norwegian Official Report entitled NOU 2013: 10 Natural benefits – on the values of ecosystem services *(Naturens goder – om verdier av økosystemtjenester)*. In September 2013 the report was distributed for a broad public consultation among affected stakeholders, including the authorities, business and industry, academic communities and NGOs. After this consultation, the Government will consider how to follow up the work.[7]

The Swedish Study: The Swedish Government decided 17th January 2013 to give a mandate to a special investigation in order to analyse interventions and suggest methods and efforts to improve the valuation of ecosystem services and to improve the knowledge about the ecosystem services value for society (Dir 2013:4). The report should also suggest interventions and measures suitable for raising the awareness in society of the importance of biodiversity and ecosystem services for integrating biodiversity and ecosystem services in decision making. This report, called "Demonstrating the values of Ecosystem Services – Measures to improved welfare through biodiversity and Ecosystem Services" (SOU 2013:68; Synliggöra värdet av ekosystemtjänster – *Åtgärder för välfärd genom biologisk mångfald och ekosystemtjänster)* was finished 15th October 2013.[8]

In addition, Statistics Sweden launched a report called "Mapping of data sources for quantifying ecosystem services" (MIR 2013:2). In this report the main ecosystems and their ecosystem services, including fresh water, are considered, and the possible methods and estimates for quantifying and valuing the different ecosystem services are assessed.

The Danish Study: The Danish Ministry of Environment has launched a short term study with the aim to describe and map Danish ecosystem services (Termansen *et al*. 2014). The background of the study is that important characteristics of environmental problems make the ecosystem services framework promising; e.g. the conflicting interests related to land-use decisions and the instruments used to regulate land-use. The aim of the project is to provide an overview of data sources, data and maps that can be used for ecosystem services mapping in Denmark, building upon existing and present mapping exercise of ecosystem services and biodiversity. The project will consider relevant indicators to ensure that present and future mapping is performed so as to ensure the possibility for valuation of the ecosystem services using the mapping

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[7] For more information about NOU 2013:10; the commission's mandate, recommendations and work see *www.regjeringen.no/okosystemtjenester*

[8] For more information about SOU 2013:68, see: www.regeringen.se/content/1/c6/22/61/92/97321dd6.pdf

exercise. A system of green, yellow and red lights will be used to indicate whether the ecosystem service is mapped (green light), whether it is not possible to use the mapping for valuing the ecosystem services (yellow light), and a red coloured light that indicate that the services cannot be mapped. The study will build on existing data and mapping exercises, but also on existing and previous projects relevant for the ecosystem services assessment.

2.1.2 Ecosystem Services Classification

Ecosystem services are usually categorised into provisioning, regulating, cultural and supporting, following the main classifications in the Millennium Ecosystem Assessment[9] (MA), while the classifications in The Economics of Ecosystems and Biodiversity[10] (TEEB), and Common International Classification of Ecosystem Services (CICES)[11] categorise the services into provisioning, regulating and cultural services. There are also numerous other classifications used in specific reports, for specific purposes etc. Most of these are slightly different, but closely related to the three mentioned above. We will not discuss different categorisations in detail here, as we believe the choice of classification is not crucial for ecosystem services considerations related to water management. We use the CICES categorisation (see table 2.1 in section 2.2.) as our point of departure in the general discussions and analyses throughout the report. However, since we will also discuss different examples from different countries, the ecosystem services categorisation will vary somewhat across examples.

2.2 Overview of Ecosystem Services in freshwater

Based on the general definition and categorisation of ecosystem services and the known ecosystems and ecosystem functions in freshwater/watersheds, one can derive the potential freshwater ES.

The illustration in Box 2.1 represents a listing of "typical» ecosystem services from freshwater in Nordic countries.

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[9] "The Millennium Ecosystem Assessment" (MA) from 2005 describes and classify Ecosystem Services and make an assessment of status and trends in the Ecosystems worldwide.

[10] The Economics of Ecosystems and Biodiversity (TEEB), was initiated in 2007 by the leaders of the G8-countries. TEEB's purpose is to increase the understanding for "the true economic value of the benefits we receive from nature."

[11] http://cices.eu

Box 2.1: Ecosystem Services in Nordic freshwater

Ecosystem services freshwater

Freshwater	Provisioning	Regulating	Cultural
Lakes	Fish, drinking water, cooling water, water for agriculture, transport	Retention and recircu-lation of nutrients, carbon sequestration	Recreation; bathing water, sailing, walking along the shoreline and on beaches, tourism, angling/recreational fisheries
Waterways, rivers	Fish, drinking water, cooling water, water for agriculture, transport	Retention and recircu-lation of nutrients, carbon sequestration	Recreation; bathing water, sailing, walking along the riverside, tourism, angling/ recreational fisheries
Wetlands	Can be used for cattle (grazing)	Retention and recircu-lation of nutrients, carbon sequestration	Wildlife/Bird watching, hunting, picking mush-rooms and berries, walking
Groundwater	Drinking water	Retention	No

COWI (2014) gives an overview of ecosystem services which are relevant for WFD using CICES as their underlying framework for listing the potential ecosystem services we receive from freshwater (cf. section 2.3 in this chapter for a further discussion of the WFD).

Table 2.1 shows a detailed version of the classification of ecosystem services in fresh water according to CICES for the biotic resources. All the ecosystem services in this table may potentially be relevant for assessing benefits from water status improvements according to WFD.

Table 2.1: Ecosystem services which may be relevant from water status improvements in freshwater – biotic

Section	Division	Group	Class
Provisioning	Nutrition	Biomass	Algae and their outputs
			Aquatic animals and their outputs
			Plants and algae from in-situ aquaculture
			Animals from in-situ aquaculture
		Water	Surface water for drinking
			Ground water for drinking
			Water for agriculture
			Process water for industry
	Materials	Biomass	Fibres and other materials from algae and animals for direct use or processing
			Materials from algae and seagrass for agricul-tural use
		Water	Surface water for non-drinking purposes
			Ground water for non-drinking purposes
	Energy	Biomass-based energy sources	Plant-based resources
Regulation & Maintenance	Mediation of waste, toxics and other nuisances	Mediation by biota	Bio-remediation by micro-organisms, algae, plants, and animals
			Filtration/sequestration/storage/accumulation by micro-organisms, algae, plants, and animals
		Mediation by ecosystems	Filtration/sequestration/storage/accumulation by ecosystems
			Dilution by atmosphere, freshwater and marine ecosystems
	Mediation of flows	Mass flows	Mass stabilisation and control of erosion rates
			Buffering and attenuation of mass flows
		Liquid flows	Hydrological cycle and water flow maintenance
			Flood protection
	Maintenance of physical, chemical, biological conditions	Lifecycle mainte-nance, habitat and gene pool protection	Maintaining nursery populations and habitats
		Sediment formation and composition	Decomposition and fixing processes
		Water conditions	Chemical condition of freshwaters

Section	Division	Group	Class
			Chemical condition of salt waters
		Atmospheric composition and climate regulation	Global climate regulation by reduction of greenhouse gas concentrations in the atmosphere
Cultural	Physical and intellectual interactions with biota, ecosystems, and land-/seascapes [environmental settings]	Physical and experiential interactions	Experiential use of aquatic plants and animals and land/seascapes in different environmental settings
			Physical use of land/seascapes in different environmental settings
		Intellectual and representative interactions	Scientific
			Educational
			Heritage, cultural
			Entertainment
			Aesthetic
	Spiritual, symbolic and other interactions with biota, ecosystems, and land-/seascapes [environmental settings]	Spiritual and/or emblematic	Symbolic
		Other cultural outputs	Existence
			Bequest

Source: Modified from COWI (2014).

2.3 The links between the Water Framework Directive (WFD) and the Ecosystem Services (ES) framework

The Water Framework Directive (WFD) is the main directive regulating the quality and the use of freshwater as well as coastal waters in EU-countries, and as mentioned the Nordic countries Norway and Iceland have adopted the requirement in the Directive as well. The ES framework can be used in the implementation of the WFD, and in this section the main claims for economic assessments in the WFD will be described along with a description of the ecosystem services framework and Payment for Ecosystem Services (PES), with the aim to propose how this framework can be utilised for the WFD implementation with specific focus on freshwater management.

This is not the first attempt to describe how the ecosystem services framework can be linked to the WFD – other studies are e.g. the ESAWADI project (Blancher *et al*. 2013) and the assessment made by COWI for the EU Commission (COWI 2014). Following Blancher *et al*. (2013) the use of the ecosystem services framework is of specific interest because of the requirement of stakeholder involvement in the WFD, and COWI (2014) also point at the ecosystem services framework for communication of the benefits of the directive. Furthermore COWI describes the ecosystem services framework's advantages for the selection of measures in the WFD as it allows for consistent assessments of the co-benefits delivered by a measure, and the ability to align the implementation of the WFD and the EU Biodiversity Strategy.

Following the recommendations from the MAES group (Maes *et al*. 2013; Maes *et al*. 2014) the primary suggestions from the present assessment is that the use of the ecosystem services framework can be very helpful to assess and illustrate trade-offs between different goods and services, i.e. how different implementation strategies might lead to different results for the provision of ecosystem services, and hence also illustrate differences between implementation strategies when it comes to the total benefits (see box 2.2 defining the total benefits or the total value) of a strategy but also for the distribution of benefits between different users or beneficiaries.

Box 2.2: Total Economic Value of an environmental change consists of several parts

Total Economic Value (TEV) include the following parts:

- *Use values* include the value of using goods and services, and the use values can be divided into direct, indirect and option values.
  - *The direct use values* we can derive from freshwater ecosystems comprise e.g. the value of fisheries and the fish resources, and other species with commercial value. The direct use values also include recreation services; e.g. bathing waters etc.
  - *The indirect use* values include the utility related to e.g. the knowledge and ability to see a river basin in good conditions with healthy functions and ecosystems.
  - *The option value* is the value of having the possibility to use the services and goods in the future.
- *Non-use values* is the value of an ecosystem good and service that is not used – i.e. the value of knowing that the goods and services are protected and preserved (existence value). The value can also be altruistic, i.e. the value of knowing that other persons can obtain utility from these goods and services. The value for future generations can also be important (testamentary value).

2.3.1 Introduction to WFD and the ecosystem services framework

Following Article 1 of the WFD the aim of the WFD is "maintaining and improving the aquatic environment in the Community. This purpose is primarily concerned with the quality of the waters concerned. Control of quantity is an ancillary element in securing good water quality and therefore measures on quantity, serving the objective of ensuring good quality, should also be established." The aim of the WFD is therefore to maintain and improve the aquatic environment in the EU, with specific emphasis on the *quality* of the waters concerned. The general objective of the WFD is to achieve "good status" for all surface waters by 2015, where "good status" means both "good ecological status" and "good chemical status".[12]

Another aim of the WFD is to integrate water policies, and also to integrate water policies with other policies. In the Article 1 the following is

[12] http://ec.europa.eu/environment/water/water-framework/objectives/status_en.htm

mentioned: "Further integration of protection and sustainable management of water into other Community policy areas such as energy, transport, agriculture, fisheries, regional policy and tourism is necessary. This Directive should provide a basis for a continued dialogue and for the development of strategies towards a further integration of policy areas. This Directive can also make an important contribution to other areas of cooperation between Member States, inter alia, the European spatial development perspective (ESDP). Utilization of the ecosystem services framework is helpful for the assessments of trade-offs and barriers between freshwater ecosystem services provision and other ecosystem services.

"Water services" is an important notion in the WFD, as well as for the interpretation of how the ecosystem services framework potentially can be used in the implementation of the WFD. "Water services" in the WFD are defined as part of Article 2(38) ("Definitions"):

> "Water services means all services which provide, for households, public institutions or any economic activity: (a) abstraction, impoundment, storage, treatment and distribution of surface water or groundwater, (b) waste-water collection and treatment facilities which subsequently discharge into surface water."
>
> EU commission, 2000

According to the Article 9, member states shall account for the recovery of the costs of these water services. Article 9.1.states, that member states shall "take account of the principle of recovery of the costs of water services, including environmental and resource costs", and in 9.4 the Directive states, that "member states shall not be in breach of this Directive if they decide, in accordance with established practices, not to apply the provisions of paragraph 1 (….) where this does not compromise the purposes and the achievement of the objectives of this Directive."

From these citations from the WFD it is clear that the ecosystem services concept and framework has a broader definition of ecosystem services than the WFD. The ecosystem services framework can however be used in analyses which are part of the implementation of the WFD where the WFD incorporates economic principles and economic tools into water management and water policy.

This overview illustrates that the WFD incorporation of economic principles and a number of economic tools into water management and water policy (cf. Martin-Ortega and Balana 2012) is important for the linkages between the WFD and the ecosystem services framework. The main areas where economic analysis in the WFD can be linked to the ecosystem services framework are the required river basin characterization in the WFD (Article 5), the use of water pricing and cost recovery

(Article 9), the assessment of disproportionate costs (Article 4), and finally the requirement for identification and implementation of cost-effective Program of Measures (PoMs) (Article 11), see table 2.2.

The ecosystem services framework can also be valuable for the non-economic parts of the WFD, as description, quantification and spatial mapping of the freshwater ecosystem services, as well as the assessment and mapping of the status of these services might be used for the definition of good ecological status as well as for the monitoring of the status.

Table 2.2: Economic requirements of the WFD and the use of the ecosystem services framework

WFD Article	Requirement
Article 4: Environmental objectives	The Member States shall implement the necessary measures to prevent deterioration of the status of all bodies of surface water to achieve Good Ecological Status (GES) of water bodies in EU countries as well as in Norway and Iceland preferably by 2015 and no later than 2027. Following the EU Commission (2013) the inter calibration exercise is used as a harmonised framework to define GES. The inter calibration[13] process involves harmonisation of the monitoring results from different countries so that similar ecological status of water bodies in different countries leads to an equal environmental quality evaluation for these bodies (Møller *et al*. 2014). The Member States are organised in Geographical Inter-calibration Groups consisting of Member States sharing particular surface water body types, making the national results comparable. The common Environmental Quality Ratio (EQR) is used for the definition of the GES. Paragraph 4.4 of the WFD opens for exemptions from the GES target, extended deadlines, or less stringent environmental objectives if achieving GES are considered disproportionately costly. The concept of disproportionate costs is only vaguely defined in the WFD. Two examples of interpretations are the welfare economic interpretation, where costs can be defined as disproportionate when they exceed the environmental benefits. General guidelines on how to perform the disproportionate cost analysis are available (Wateco 2003; European Commission 2009), and even though these guidelines are not very detailed and they do not suggest a practical procedure by which a country can carry out this analysis, they suggest that judgment of disproportionate costs could be based on an economic analysis of the costs and benefits of achieving GES (European commission 2009, Wateco 2003). Some studies have investigated how welfare economic cost-benefit analysis can be used for the assessment of disproportionate cost (e.g. Bateman *et al*. 2006; Hanley and Black 2006; De Nocker *et al*. 2007; Lago *et al*. 2010; Molinos-Senante *et al*. 2011; Kinnel *et al*. 2012; Vinten *et al*. 2012). Examples in a Nordic context are Jensen *et al*. 2013; Holen and Magnussen 2011; Magnussen and Holen 2011).
Article 5: Characteristics of the river basin district, review of the environmental impact of human activity and economic analysis of water use	Water quality and status depend on several water characteristics – i.e. chemical, physical, hydro morphological and biological conditions. This means that measurement of water quality and status is directed against different pollutants and conditions depending on the water body observed. It also means that the relevant measure of quality varies between different types of water bodies. The typology developed for the WFD is useful, as the water bodies are classified in terms of quality and status on a 5 step scale from High to Bad (High, good, moderate, poor, bad) where this classification can be tied back to the status of the specific physical and quality conditions of the specific water body.

[13] This is however not the case for heavily modified water bodies.

WFD Article	Requirement
Article 9: Recovery of costs for water services	Each member state shall take account of the principle of recovery of the costs of water services, including environmental and resource costs. The water services include all services (public or private) of abstraction, impoundment, storage, treatment and distribution of surface water or groundwater, along with wastewater collection and treatment facilities. Economic analysis of the environmental and resource costs should be made and cost recovery should be in accordance with the polluter pays principle.
Article 11: Programme of measures	The aim of article 11 in the WFD is to identify cost-effective programmes of measures (PoMs). Each member state shall ensure the establishment for each river basin district, or for the part of an international river basin district within its territory, of a PoMs, taking account of the results of the analyses required under the above described Articles 4, 5 and 9. A central requirement is that the selection of PoMs should be based on a cost-effectiveness analysis (CEA) of abatement and mitigation measures. CEA aims at finding the combination of the least costly measures at river basin level that reach the goal of the WFD; these are then to be included in the PoMs in local river basin management plans. The measures should also safeguard water quality in order to reduce the level of purification treatment required for the production of drinking water; i.e. safeguard this provisioning service. A large number of studies have assessed cost-effectiveness of nutrient reduction measures, including WFD measures. A few examples relevant in a Nordic context comprise Barton *et al*. 2005; Barton *et al*. 2008; Brady 2003; Jacobsen 2007; Hasler 1998; Elofsson 2012; Iho 2005.

In the next section we present and discuss how the ecosystem services framework can be used for these tasks in the WFD, and vice versa – how the WFD implementation activities can be used for the assessment of ecosystem services.

2.3.2 The use of the ecosystem services framework for the different steps in WFD

We have described that the ecosystem services framework may be of use in several tasks connected to water management. The ecosystem services framework is useful as a tool to capture and describe benefits and possible co-benefits of achieving the objectives of the WFD, and thereby support the implementation of the WFD.

It is clear that the ecosystem services framework can be used in relation to the assessment of disproportionality of costs of implementing the WFD objectives, as the ecosystem services approach can be used to include the full range of benefits of water quality changes and of the measures implemented to obtain these, and also, as mentioned be used to describe and include non-quantifiable benefits which is described as part of the assessment. The benefits to people from environmental improvements include use and non-use values, see Box 2.2, and the listing of ecosystem services can be used as a way to identify benefits to different groups of people, both use and non-use values. However, in order to be useful, we need to carefully identify the ecosystem services that will be affected and the benefits they give to people, quantify them, and if possible value them.

This benefit assessment may be performed at different scales – on a national level, on a river basin level, and on a water body level. And it may be used as a screening procedure (as in Jensen *et al*. 2013), or in a more detailed benefit cost analysis on water body level. Jensen *et al*. (2013) use Danish data to propose such a CBA-based, river basin level screening procedure as a first step to identify the river basins in a country, here Denmark, where disproportionate costs are likely to occur. Jensen *et al*. (2013) propose that this screening can be used to identify where more comprehensive and costly assessments, e.g. of the full range of ecosystem services as inputs for more comprehensive CBAs, should be undertaken in order to assess disproportionality. Jensen *et al*. (2013) use an existing valuation study from Odense River Basin (Hasler *et al*. 2010; Jørgensen *et al*. 2013; Bateman *et al*. 2011) and benefit transfer of these results to the other river basins to estimate the benefits of the WFD, and on the other hand existing cost assessments (Jacobsen 2013) are used. Their conclusion is that if the welfare gain is clearly positive for the CBA in a given area it is likely that the costs do not exceed the benefits and achieving GES should not be claimed disproportionate, but if the welfare gain is not clearly positive the potential for disproportionality between costs and benefits should be further investigated. Jensen *et al*. (2013) do not explicitly discuss how to use the potential of using the ecosystem services framework for extending such CBAs.

Another issue that needs to be dealt with in estimating the benefits from water quality improvements are trade-offs between different ecosystem services. As discussed in TEEB (2010) and in Barton *et al*. (2012) the need for detail and carefulness with respect to scale, trade-offs, and the value of other inputs, differ with the intended use of the benefit estimates. If the purpose is to demonstrate the values we receive from rivers, a more general, not so detailed assessment of the ecosystem services and their benefits to people might be appropriate. But if CBA is performed in order to consider whether the project is beneficial to society, or if costs are disproportionate, a more careful and detailed assessment is needed. If the value (priced or unpriced) of affected ecosystem services are to be part of a CBA we need the net value from ecosystem services improvements, and it means that we need to identify any other inputs (capital, labour etc.) used to measure the final benefits.

The discussion above reveals that the ecosystem services framework can be used in many ways to guide the discussion and measurement of disproportionality.

The ecosystem services framework can be used for defining good ecological status according to WFD if knowledge of the relationships between the good ecological status and the services can be established. I.e. what are the ecosystem services of the different indicators for good ecological status of the water? The ecosystem services framework can also be used for setting targets and objectives by integrating indicators for good quality of the goods and services and the ecosystem services framework.

The ecosystem services framework can further be used for the choice of measures for the PoMs. Through the integration of the ecosystem services framework into the assessment of the PoMs, the additional benefits can be illustrated and taken into account in the choice of measures. The ecosystem services framework can be applied with and without valuation of the services so that non-quantifiable services can be included in the assessment and choice of PoMs.

The implementation of some of the measures in the PoMS can provide additional ecosystem goods and services, while others don't. Examples are creation of wetlands and buffer strips that, beside the regulating service of increasing the nutrient retention and the transformation of nitrogen (N) to harmless nitrogen compounds (NO_2), wetlands and buffer zones can also be habitats for wild animals, flora and insects, and therefore improve biodiversity compared to agricultural fields. Furthermore wetlands might be a measure to reduce floods in the cities by regulating the water flows. The conversion of arable land to permanent grasslands is a measure which can increase the carbon storage, i.e. imply a regulating service affecting climate change. In comparison a measure like nitrogen fertilizer reductions cause few indirect ecosystem services beyond the effects on the aquatic environment, and some reductions in energy use which might be relevant for the use of fossil energy.

The objective of good ecological status can also be obtained by hydrological and other technical measures as planned in the current River Basin Management Plans (RBMP). One measure aims at changed maintenance of water ways by reduced removal of plant biomass from the bottom and edges. This measure will affect the retention of nutrients and improve the conditions for flora and fauna in the waterways. Another measure is changing of the hydrological conditions in the creeks and waterways, by adding stone and gravel to the bottom that will improve the oxygen conditions at the bottom, and also improve the habitat value for juveniles and fish. Closed, excavated watercourses can be reopened and barriers for fish can be removed as measures to improve the quality of these habitats, e.g. increase the recreational value of angling.

An example is that for lakes the present indicator of good ecological status is the content of chlorophyll. The objective of good ecological status can be obtained by hydrological and technical measures within the lakes, and by reducing nutrient loads to the lakes, especially phosphorus. The phosphorus loads to lakes can be reduced by measures at fields under risk for phosphorus losses, and by restoration and construction of wetlands with the aim to retain phosphorus in the drainage areas to lakes. Buffer zones along rivers, streams and lakes is another measure aiming at retaining phosphorus, and as mentioned above both wetlands and buffer zones might improve additional ecosystem services beyond their effect on the nutrient retention.

COWI (2014) also discusses the use of the ecosystem services framework with the PoMs, also connected to communication. This is one of the major motivations for the TEEB project for example, while the WFD in many cases seem to under-estimate the need for communicating benefits to people.

2.4 Economic instruments for locally targeted measures – PES and water quality trading

The ecosystem services framework is one of the cornerstones in a number of economic policy instruments relating to water quality pollution. The voluntary policy instrument PES is based on a payment made for the delivery of ecosystem services and the mandatory water quality cap-and-trading instrument is based on understanding and counteracting the effects of deteriorated ecosystem functions on ecosystem services. PES schemes that target water quality pollution are found already in the Nordic countries and Europe (See Chapter 5). These PES schemes are not initiated because of the WFD, but are typically firmly established in the Common Agricultural Policy (CAP) or target drinking water protection and aquifer replenishment. Nevertheless, these policy instruments contribute significantly to meeting the obligations under the WFD and may potentially play a larger role for the WFD than today. Water quality cap-and-trading is primarily found in the US with a few cases in Canada and New Zealand. Common for policy instruments aiming at improving water quality is the growing recognition that instruments need to be adapted to local conditions.

2.4.1 Payment for Ecosystem Services – PES

PES works as a conditional performance contract where the provider of ecosystem services is contractually obliged to create, enhance or protect a specific ecosystem service or a bundle of ecosystem services. The beneficiaries of the ecosystem services pay conditionally upon the delivery of the ecosystem services. Beneficiaries can be the state on behalf of society, an enterprise in order to protect its production, an NGO or municipality on behalf of a community or landowners to offset own obligations. Ideally, ecosystem services in PES schemes should be well-defined or a land-use be clearly identified that is likely to secure the ecosystem services.

An argument in favour of paying for ecosystem services as compared to command and control regulation is that PES may offer a more cost-efficient way to ensure that nature and landscape is taken care of and improved in the agricultural areas and by offering a voluntary scheme, resistance from land owners can be minimised. Some of the challenges in the use of PES, however, is obtaining the appropriate spatial coverage (since the instrument is voluntary) and securing financing of the contracts and keeping transaction costs down when moving towards local PES schemes. Chapter 5.5 presents two cases where it has been possible to obtain an almost complete spatial coverage at a catchment area scale. In one case, sufficient financial incentives combined with a clear outreach and communication made the uptake cover 80% of the intended area, while in the second case, a combination of carrot and stick ensured that within five years of the scheme, 93% of farms had signed up (if 85% of land owners had not enrolled within five years, regulations would replace financial incentives).

PES schemes come in a variety of forms and set-ups; there are necessary conditions along with desirable/useful conditions; there are different elicitation methods for closing contracts. Most PES schemes, however, are paid based on an activity as a proxy for delivering a specific service as opposed to an outcome based payment. In practice, PES is rarely contracted directly for the delivery of an ecosystem service, but indirectly through the agreement to undertake a change in land management or land use. PES can be spatially targeted or non-targeted; they can be set up to accommodate local conditions or they can be designed as uniform contracts with uniform payments. Neither subsidies nor other compensating mechanisms like PES function according to the Polluter Pays Principle. However, if PES is a payment for a service or a bundle of services that lead to positive external effects, this would be according to economic theory. Zandersen *et al*. (2009) provide a more in-depth analysis of ecosystem services classification and PES in a Nordic context.

As a policy instrument, PES belongs to the group of market-based voluntary incentives. Box 2.3 gives an overview of the different types of policy instruments that are in use to deal with the problem of water quality pollution. These policy instruments are most often mixed to better deal with complex environmental issues.[14]

[14] There is increasing attention paid to policy or instrument mixes in the EU and worldwide. See for example the ongoing EC funded project POLICYMIX: http://policymix.nina.no/

Box 2.3: Overview of different types of policy instruments

Command & control

Market-based compulsory incentives

Charges | Tradable permits | Market friction reductions

Market-based voluntary incentives

Land acquisition Pre-acquisition | Easements | Subsidies | Conditional performance contracting

Other policy instruments

Ecolabelling | Information campaigns | Voluntary agreements

An example of mixing policy instruments from Denmark for the same farm is:

- the requirement to respect a fixed not tradable quota for the application of fertiliser which is 10% below the economic optimum (command and control instrument)
- the requirement to apply catch crops on 14% of the fields grown with crops where catch crops are required (command and control instrument) (only farms larger than 10 ha). If the farmer chooses not to establish catch crops he can establish energy crops, in-between crops ("mellemafgrøder"), burn the fiber fraction of animal manure, or transfer the obligatory catch crop area to another farmer or between years so that surplus catch crops areas from former years counts as catch crops this year. If the farmer does not adopt any of these actions, the nitrogen quota will be reduced. In other words the catch crop requirement can be traded between farmers, between years as well as between measures, so that too low implementation of catch crops may be compensated by a lower N-quota
- a tax on phosphorus in fodder, as well as a pesticide tax (a market-based compulsory incentive)
- a possibility to receive a one-off payment and yearly management fee for the establishment of wetlands (a voluntary performance contract).

Sources: Adapted from Jack *et al*. (2008) and Ferraro (2009).

Agri-environmental policies are examples of Payments for Ecosystem Services that pay farmers to reduce the negative externalities of agricultural production (Baylis *et al.*, 2008). In agri-environmental schemes, governments represent the wider society as "buyers" of the delivery of one or multiple ecosystem services. Chapter 5.1 provides an overview of the greening of the EU Common Agricultural Policy and Chapter 5.4 gives examples of Nordic PES schemes under the EU CAP that are relevant to the WFD.

2.4.2 Water quality trading

The ecosystem services framework is also central to water quality trading, a market-based compulsory incentive. Water quality trading offers a possibility to increase flexibility and reduce costs when aquatic quality standards are established or tightened as under the WFD. It allows emitters with new obligations to either adapt their own facilities and land use practices or finance comparable emission reductions by others. Trading makes it profitable for sources with low treatment costs to reduce their own effluents beyond legal requirements, to generate an emission reduction credit and to sell these credits to emitters with higher treatment costs (Faeth, 2000). Trading necessitates clear emission ceilings that are mandatory for all emitters in the defined market. By allowing trading, it's possible to obtain a less expensive (more cost effective) outcome overall while achieving – or in some cases going beyond – the mandated environmental target. Water quality trading is not necessarily a stand-alone policy instrument but typically enters a mix of instruments combining regulations, taxes and subsidies. Trading per se is voluntary for emitters while the emission ceiling is mandatory. Examples of water quality trading in practice are presented in Chapter 5.6.

2.4.3 Locally adapted policy instruments

River basins vary considerably with respect to the natural state and the current quality. Different factors affect different river basins and the same is the case for sources of pollution. Non-point source pollution such as arable nitrogen emissions to water bodies are in practice very difficult to control because of the prohibitive costs of measuring the contribution of individual farms to ambient pollution levels. In practice therefore, regulation of non-point pollution has been based on factors that *indirectly* determine pollution levels and are possible to monitor at a reasonable cost. This includes taxes on commercial inputs (e.g. fertiliser), changes in crop management practices (e.g. catch crops during winter), or land use changes (e.g. construction of wetlands or riparian buffer zones).

However, socio-economic, physical and geo-chemical conditions differ across space. Some farm systems have higher opportunity costs and/or production costs than other farm systems; within farm systems, farmers vary in their ability to run a farm profitably; soil conditions and retention capacities can vary within a fairly small area; and water bodies have different quality levels and different capacities to cope with additional loads. Given this complexity of heterogeneous conditions across space, uniform or undifferentiated policies are unlikely to i) meet the ambient water quality targets and ii) ensure an implementation at least-cost or relatively low cost. This large variation makes it difficult to establish nationwide (economic) instruments that will be economically optimal (or even work well) in all river basins across a country. In practice, however, uniform or undifferentiated policy instruments dominate in the Nordic countries as well as in Europe.

Dealing with non-point source pollution at a catchment scale as under the WFD represents a "wicked" problem for which there can be no single instrument or measure; rather there is the need for a *mix* of instruments, a mix of measures as well as a mix of stakeholder involvement. This *mix* of instruments can be more efficient when adapted to local conditions.

Wicked problems are characterised as complex, dynamic, uncertain with diverse legitimate values and interests. There is no definite problem formulation because there are many externalities, beneficiaries and multiple trade-offs (Smith *et al.* 2011). For instance, on the issue of *localizing* land use measures optimally in terms of cost efficiency, it is challenging because i) there are multiple conservation practices with different effectiveness and costs; ii) there are multiple water quality endpoints (nitrogen, phosphorous, sediments, etc.); and iii) water quality effects from one field may be affected by choices on other fields, implying the need for cooperation among polluters.

The provision, quality and value of ecosystems services are very site specific – the provision of groundwater for drinking water is for instance dependent on the soil's ability to purify the water so that polluting compounds are not above the limit values, bathing water quality depends not only on the nutrient loads but also geomorphological conditions, and wetlands' ability to retain nutrients depends on site specific hydrological conditions, among other factors. Instruments with an ability to target these local conditions will be preferable, and a mix between local and general instruments (e.g. PES schemes and taxes/transferable quotas) can be a cost-effective mix. This is also important since not only recipients have different characteristics, also polluters and other actors have different characteristics, objectives and behaviour, and they will therefore react differently to different incentives. This suggest that one should think more varied when it comes to the structure and choice of instruments in a second best option – as first best are seldom achievable because of the diffuse character of nutrient emissions.

Targeting pollution control programmes means abating more where it will be most effective and least costly (Braden and Segerson 1989). It requires combined ecological-economic models that identify the best methods and locations for reducing and containing discharges (See Chapter 5.2). Literature on this issue agrees that targeting areas and adapting measures locally will be cheaper, less disruptive (for the farming industry) and more ambitious targets can be imposed than if all farmers are met by the same abatement standards (Braden and Segerson 1989; Brady 2003).

There are good reasons to consider combining different instruments and establishing mixes of instruments for each river basin/water region given the WFD required region based management, variation in appropriate instruments across sectors and the heterogeneity of agents within sectors. This has been suggested for example in Norway in an evaluation of increased use of water pricing in Norwegian water management (Magnussen and Holen 2011). Several countries have discussed and tried such models, e.g. this is the case in Morsa which is one of the example river basins in VALUESHEDS as well as in Denmark where the Commission of Nature and Agriculture have proposed more targeted measures in a mix of general and local instruments for nutrient abatement and mitigation (See Chapter 5.2).

2.5 Main findings in this chapter

The ecosystem services framework can be used to map and measure the value of the changes in provisioning, regulating and cultural services, and the trade-offs between these when the policy targets of the WFD are modelled.

The ecosystem services framework offers a more thorough assessment of benefits of positive environmental changes in a complex ecological system. It can help improve the evaluation methodology of disproportionate costs. This is exemplified in Jensen *et al*. (2013) who use information on ecosystem services and economic valuation of water quality and ecological improvements in Odense river basin in a benefit transfer to other Danish water bodies. The benefit transfer results by river basins are subsequently used for a cost-benefit analysis for the WFD implementation in Denmark. The CBA is used as a conservative screening of where costs appear to be disproportionate, i.e. exceed the benefits.

Furthermore the ecosystem services framework can be used to assist the analysis of the Programme of Measures and the cost-effectiveness of the measures. Examples hereof are presented in the next chapters.

This chapter also introduces economic instruments for locally targeted measures, PES and water quality trading and show how the ecosystem services framework is one of the cornerstones in such policies. The voluntary policy instrument PES is based on payment made for the delivery of ecosystem service. The mandatory water quality cap-and-trading instrument on the other hand is based on understanding and counteracting the effects of deteriorated ecosystem functions on ecosystem services.

3. Use of the ES framework to describe and value benefits of improved ecological status in water

In this chapter we:

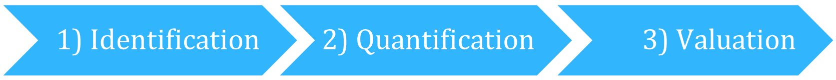
- Discuss the necessary steps for benefit assessment of water status improvements based on the ecosystem services framework: identification, quantification and valuation (section 3.1).
- Discuss identified important issues to consider in ecosystem services based benefit assessment: the purpose of the study, the scale (local, regional, national, etc.), the need to consider trade-offs, the ecosystem services' share of final benefits (value added from ecosystem services), and the issue of double counting (section 3.2).
- Provide examples from the Nordic countries (and a few others) in order to illustrate how ecosystem services can be identified and mapped (section 3.3) and quantified and valued (section 3.4).
- Discuss and conclude regarding findings and what we can learn from this chapter (section 3.5).

3.1 Benefit assessment based on ecosystem services

> "The ecosystem services framework does not necessarily imply something radically different in terms of the application of the valuation techniques themselves, but it does imply the development of valuation scenarios more solidly rooted in the biophysical underpinning of ecosystem functions, service delivery and stakeholder engagement processes, and can help to address issues such as at which scale costs and benefits are to be measured."
>
> Martin-Ortega (2012; page 87)

This way of seeing the role of ecosystem services for benefit estimation connected to WFD is the approach taken in this report.

Following traditional welfare economic analysis (Cf. e.g. Ministry of Environment, 2010) an assessment of ecosystem services should follow these three steps:



In the following sections, we will describe each of these steps, respectively.

3.1.1 Identification of ecosystem services – which ones will be affected

In order to assess the benefits of ecosystem services from improved water status, we need to identify exactly which benefits are affected by the change. In chapter 2 we described the different categories of ecosystem services, which is the basis for identification. Furthermore, there are descriptions and listings of which ecosystem services are potentially important in rivers and watersheds which may be used.

If a recognition of ecosystem services in freshwater is the main purpose, a general identification and listing of the ecosystem services that will be improved by improved water status may be all that is needed – in order to demonstrate and communicate the importance of improved water environment by reaching the environmental goals of the WFD – good ecological status (GES). On the other hand, if the purpose is to use the ecosystem services framework to estimate the benefits as input to a CBA of the benefits and costs of achieving GES, we need to be more specific about which ecosystem services are really being of relevance *and affected* by the proposed measures which will be implemented in order to improve the water environment. And if the purpose of applying the ecosystem services framework is to enable assessment of how different ecosystem services are affected by a policy change, the ecosystem services also have to be separated and not merged into a general description.

The first step in all cases will be to identify the ecosystem services of importance in the river basin or water body of the study. In the second step we need to be more specific about whether the proposed measures will change the quantity or quality of these ES.

This identification of affected ecosystem services *could* be carried out for each measure or intervention in a cost-effectiveness analysis. However, it is a quite demanding process to do this for each measure. If the purpose is to estimate the total benefits of the improved water status, therefore assessing the benefits of the total package of measures can be an alternative. However, as mentioned in chapter 2, different kinds of measures may affect different ecosystem services (Magnussen *et al*. 2014; Termansen *et al*. 2014). Our description of measures used for the implementation of the WFD also show this as different measures provide different additional ecosystem goods and services, beyond the water quality improvement, such as e.g. flood protection (wetlands), carbon storage (grasslands/set aside). One could make benefit assessment for different packages of measures – which affect different ecosystem services and hence result in different total benefits. Programming (optimisation) models can also be used to analyse and quantify trade-offs between different ecosystem services and the share of each ecosystem services of the total benefits of an optimal policy. Such models are developed for e.g. nutrient load reductions (e.g. Hasler *et al*. 2014), and these models can be extended to include e.g. the effect on climate regulation, biodiversity etc., and used to assess the specific value of e.g. retention as a regulating ecosystem services. In real life studies, we often find that it is not so easy to identify precisely which ecosystem services are affected. This may be due to lack of ecological knowledge regarding the effects of different measures. It may also be difficult to quantify and value the benefits.

3.1.2 Quantification

The next step is to quantify in physical units the identified, affected ecosystem services. This is often challenging. It can be difficult enough to tell that the water will be more suitable for fishing or swimming if the water status is improved. In this step, however, we want to quantify "how much better" it is suitable for swimming, how many people benefit from this improvement, how much will breeding conditions improve, and how much will the living conditions improve – and how many, and which fish species, will be available for anglers to catch?

The quantification can be based on existing sources, such as monitoring data and maps, statistical data on fish catches, the number of visitors etc. It may be necessary to combine different existing sources. For example, the number of people who will benefit from water improvement in a specific river or lake may be important information. We then need to identify how far from the water string people are affected. This may differ for different ecosystem services. Then combining maps and statistics of population, one can estimate the number of people affected. This is illustrated in the example from urban Oslo's water bodies (see section 3.4.3) where the local community's GIS-office could give exact numbers of inhabitants living less than 100, 300 and 1,000 meters from the water bodies in question, respectively. Similarly the respondents of the Aquamoney study in Odense were asked about where they live and to indicate their address either on a map or to provide information about the road-name and house number (within an interval to avoid drop outs because of lack of anonymity), and also to click on a map to indicate the area along the coast, fjord, river or lake where they went to for their last visit. The researchers therefore obtained information about the distances from where people live to the places they go for recreational visits, and in the same survey the respondents were also asked about their willingness to pay for water quality improvements. But the missing link is the connection between the water quality and their preferences for particular services, such as swimming and angling, as they are not specified in the study. We return to that question in the next section.

3.1.3 Valuation

In conducting a cost-benefit analysis in order to assess the benefits compared to costs, we aim at monetizing all effects that may be meaningfully monetized. The effects that cannot be meaningfully monetized should also be included, as so-called un-priced effects, which are treated in the analysis in quantified, physical terms or qualitatively described. In all cases, we should start with identifying the effects (identify ES), quantify them as far as possible in physical terms (quantification of ES) and value in monetary terms as far as possible.

In practice we are not always able to value all benefits from water quality and water ecosystem services improvements. This may be due to several reasons. We discussed in 3.1.1 and 3.1.2 difficulties of identifying and quantifying ecosystem services from water status improvements. Furthermore, some ecosystem services are difficult to value due to shortcomings in the methodology for valuation, because we do not know enough about the ecological effects or because laymen being asked about their willingness to pay don't have knowledge of the ES. For most or many ecosystem services there are no market prices. Therefore we need to estimate prices using methods developed to value non-market (environmental) goods.

There are several methods available and applicable to value environmental goods, both use and non-use values,[15] see box 3.1 for a brief overview. We will not discuss valuation methods further here, as a description of such methods is provided many other places, for instance in Barton *et al*. 2012 with reference to fresh water management.

Sometimes it is deemed too time consuming and/or costly to carry out new primary studies. In some cases, we can find valuation studies and "prices" for similar ecosystem services in other river basins or water bodies, and then benefit transfer is an option. The concept and methods for benefit transfer are presented in box 3.2. Valuation results will be more uncertain when using benefit transfer instead of collecting new site-specific information for valuation. The more similar the change in ecosystem services to be valued, the context of the ES, and the affected population are, the better one can expect the benefit transfer value to be. However, in most of the Nordic countries, the largest problem with benefit transfer is the lack of available, original studies to transfer from. Still, this is often the option used in practice.

Box 3.1: Valuation methods for environmental goods

Market	Approach	Type of value elicited	Common valuation methods
Existing markets	Market based	Use values	Market prices, Production Function Methods, Preventive costs, Mitigation costs, Replacement costs
Parallel market	Revealed Preferences	Use values	Travel Cost Method, Hedonic Price Method
Hypothetical Markets	Stated preferences	Use and non-use values	Contigent Valuation Method, Choice Experiments

[15] Use and non-use values are described in box 2.2 in chapter 2.

Box 3.2: Benefit transfer

Benefit transfer (BT) involves transferring an economic value of a public good estimated from a study site (source site; primary valuation study) to a policy site (target site). Both benefits and costs can be transferred, and the term "Value Transfer" (VT) is also used to cover both.

There are three basic requirements for value transfer:

- Database with primary valuation studies.
- Criteria for assessment of the quality of primary valuation studies.
- Methods for value transfer.

There are different approaches to value transfer (and different ways of categorizing the approaches; the listing below builds on Navrud 2008). So far, there is no single universally adopted methodology used for BT (VT).

- Unit value transfer: the unit value at the study site is assumed to be representative for the policy site, with or without adjustments for differences in income levels etc. between the two sites.
- Value function transfer: a valuation function is estimated at the study site and transferred to the policy site.
- Meta analytic transfer: A valuation function is estimated from several study sites using meta-analysis.

Transfer of the valuation results from Odense to other water bodies has been tested between Danish water bodies (Odense and Roskilde fjords) and between Odense and other North European rivers, among them Morsa in Norway. The tests showed that the benefit transfers from this study, where the ecosystem services are presented and valued in a holistic manner, resulted in relatively low transfer errors (Bateman *et al*. 2011; Källström *et al*. 2010).

From Sweden, we present a benefit transfer exercise where value estimates from river basins in Norway (Morsa) and Denmark (Odense) are used to estimate values in nearly all Swedish rivers, see chapter 3.4 (Hasselstrøm *et al*. 2014).

Another issue, raised by the valuation study in Odense, is that the measurement of water quality according to the WFD does not necessarily conform to laymen's perception of good water quality (cf. Kataria *et al*. 2012). When valuation studies use the monitoring results to characterise the water quality this might differ from laymen's perception of the quality and also differ in terms of their preferences for when they will use the water body for recreational purposes. Therefore, the recreational services, as perceived by the users, might not be directly linked to the water quality indicators in the WFD.

3.2 Some issues to consider in ecosystem services benefit assessment of improved water status

In most cases, where we want to assess the benefits received from water quality improvements, the above mentioned steps to identify, quantify and value the ecosystem services affected will be necessary. Still, although the steps are the same, different purposes may influence the way these steps are carried out; and it may influence which scale is appropriate and whether or not trade-offs or double counting need to be considered.

3.2.1 Purpose

The purpose of the study will be important for how the ecosystem services assessment is carried out. For instance, if the purpose is to demonstrate the values of good ecological status, the demand for detail and precision may not be so high, and it may or may not be appropriate or necessary to monetize the values. On the other hand, if the purpose is to compare benefits and costs, and assess whether the costs are disproportionate to analyse if exemptions from the general environmental goal of WFD are justified, the demand for detail and precision is much higher. We will show examples of different purposes and detail in sections 3.3 and 3.4.

3.2.2 Scale

Ecosystem services may be identified on different geographical scales, depending on the scale and purpose of the study to be carried out. One of the first, and still most famous (though much disputed) papers on the value of ecosystem services, was Costanza *et al.*'s paper in "Nature" on the value of the world's ecosystem services and natural capital (Costanza *et al*. 1997). There are also studies on the European scale, for instance on the value of climate change induced losses of wetlands in Europe (Brander *et al*. 2012), and Kettunen *et al.*'s (2013) assessment of Nordic values of different ecosystem services, although they do not value ecosystem services from fresh water as such. We will come back to this publication in section 3.4. Another recent publication of interest in this respect is TEEB for water and wetlands (Russi *et al*. 2013) which emphasise the importance and values of water and wetland. Neither of these publications discusses freshwater ecosystem services related to WFD. However, the general framework for identification, quantification and valuation of freshwater ecosystem services is much the same.

In order to assess the benefits and compare to the costs in a CBA, this may potentially be useful at different scales and at different stages in the WFD implementation.

An interesting example of using CBA on a national scale is Jensen *et al*. (2013) who use CBA, including valuation of good ecological status according to the WFD as a national screening procedure, in order to identify the rivers/river basins where the costs of measures may be too high compared to the benefits achieved, in which case they recommend that further CBA analysis should be carried out. This example is described in 4.3, but worth noticing here is that for this overall screening the value of good ecological status is satisfactory, but in areas where this coarse framework indicate that the costs are disproportionate there is a need to apply the precautionary principle by assessment and valuation of the detailed, partial ecosystem services more in depth – such as the recreational, provisional and regulating services.

Similar procedures could be used on a river basin scale – screening water areas or water bodies which need further and deeper analysis in order to estimate benefits and (disproportionate) costs. One way to deepen the analysis could be to include the ecosystem services of the measures in the Programs of Measures (PoMs) on a water region scale, as described in chapter 2. We will give some examples of this in the following (section 3.4.).

The water body level is also the correct scale for assessing goals and exemptions for heavily modified water bodies (HMWB). Assessment on this scale for HMWBs is mentioned in the example from river Alna and Hovinbekken in urban Oslo, Norway (section 4.4).

3.2.3 Trade-offs

In many studies in which the purpose is to demonstrate or illustrate the benefits we receive from improved water status, it is not (so) important to consider the potential need for trade-offs between different ecosystem services. In cost-benefit assessments however, including CBA for assessment of disproportionate costs, this may be of importance. There are four main types of trade-offs considered central (Magnussen *et al*. 2013):

- *Between goods and services*: Use and management of water resources may enhance one or a few services or use areas, at the expense of others.
- *Over time*: Management may give benefits in the short run, but negative impacts and costs at a later point in time. For management and use of natural resources the long-term perspective is normally of great importance.
- *Between interest groups*: Prioritizing of use areas which are important for some interest groups compared to others. If such trade-offs have to be made, it may be that some groups mainly receive the benefits while others bear the costs.
- *Spatial*: Different kinds of regulations or management regimes may give benefits and costs that are spatially differentiated.

These trade-offs may be more or less inter-related. However, we will discuss each of them in turn. Trade-offs between goods and services and over time are traditionally handled in welfare economic analysis while this is not the case for the latter two because these aspects are considered to be distributional/equity effects. However, these issues may be of great importance in practice, because they relate to who should pay for the improvements. In rivers, for example, the maximum benefits to society could be to reduce pollution in the water bodies upstream, because all the downstream water bodies then will reach improved water status. However, people living in the upstream water bodies may not receive so much of the benefits, while they may have to pay for most of the measures.

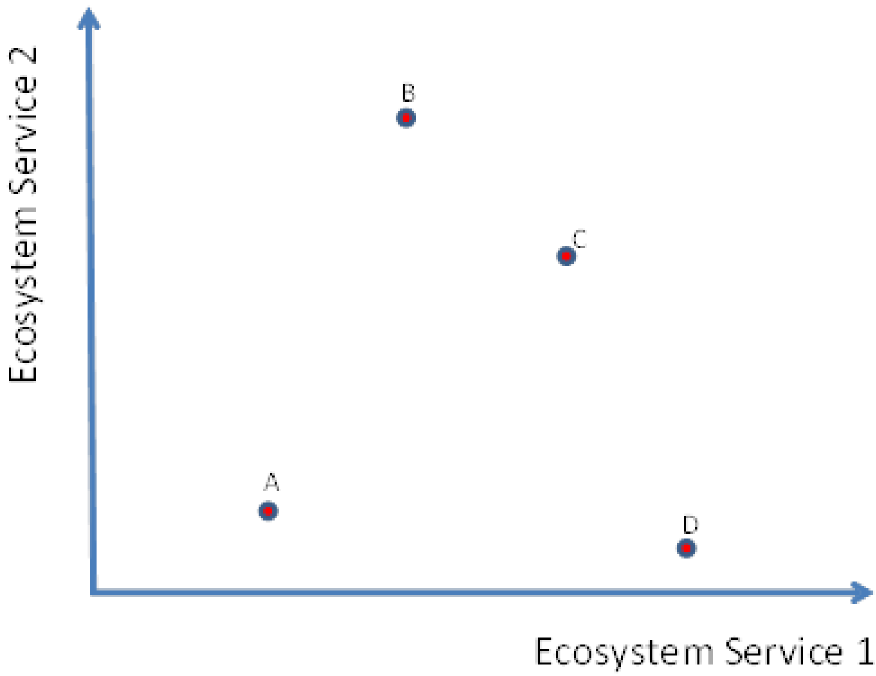
Trade-offs between goods and services

Much of the literature on trade-offs between ES, analyses ecosystem services measured in physical units, e.g. water quality and quantity, fish production, etc. (Kareiva *et al*. 2011). Obviously, there may be purely biological trade-offs: More of one ecosystem services result in less of one or several others. For example increased fishing of one species can reduce the fish catch of another species or of other services in the river. There are complicated and often not well-known ecological relationships between different provisioning, regulating and cultural services. How these relationships are in the watersheds will be important for how different uses affect the functioning of ecosystems.

One central question in an economic analysis is what someone has to give up of one service in order for someone (else) to receive more of another. This trade-off is dependent on the underlying physical and ecological relations, but also on how the different ecosystem services are valued on the margin. In a situation where we have much of one ecosystem service, for example much trout in a river, increasing the number of trout may not be highly valued. This fact implies that trade-offs between more trout and less water for irrigation will depend on the current flow of services and the value of each of them.

Figure 3.1 can illustrate this point (based on Polasky *et al*. 2011). Suppose that there are four political choices, regulations or measures that are considered for a particular river basin (points A, B, C and D in the figure) and that the costs of these measures are equal. Suppose further that there are two ecosystem services or potential uses only, for example using water for irrigation or for angling. These two services are marked as Ecosystem Service 1 and 2 respectively on the two axes in the figure. Each of the four alternatives gives different combinations of the two ecosystem services. We can interpret the axes as showing physical units or economic units (Euro).

Figure 3.1: Simplified example of combinations of level on two different ecosystem services under four hypothetic management regimes



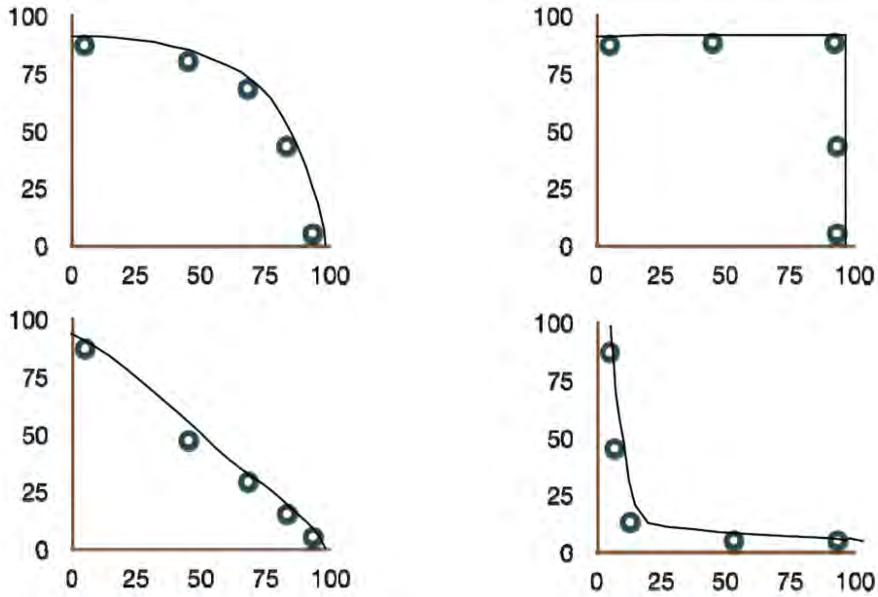
Source: Based on Polasky *et al*. (2011); here reproduced from Magnussen *et al*. 2013.

It is easily seen that alternative B and C are preferred to A, because in B and C we can have more of both ecosystem services than in A. Choosing between B, C or D (or between A and D) however, includes trade-offs: Each alternative gives more of one ecosystem services and less of the other.

If we were in situation A, in our illustration this would not make it necessary to make trade-offs to choose alternative B or C, since these alternatives give more of both services.

Considering one set of management alternatives that gives the most efficient combination of the two ecosystem services, we can draw a line between these alternatives that define an "efficiency frontier" for the ecosystem services. These management alternatives are defined where it is not possible to achieve (produce) more of one service without reducing (producing less of) the other, that is one has to make trade-offs. Figure 3.2 illustrates four different possible connections between pairs of ecosystem services. We may imagine the service measured in physical units (number of recreation days, fish in tons, etc.) or in Euros.

Figure 3.2: Trade-offs between pairs of ecosystem services depending on how they affect each other



Source: Barth *et al*. (undated); here reproduced from Magnussen *et al*. 2013.

In the top left diagram a relatively typical connection between pairs of services (and production of goods) is shown.[16] If we are in a situation with much of service 1, reducing this a little, we can have much more of service 2. The "exchange rate" changes as we have less of service 1 and more of service 2. This has analogies in reality for many kinds of ecosystem services and commercial and non-commercial uses of natural resources. Polasky *et al*. (2011) modelled for a specific forest area a connection with this shape for expected number of protected species (y-axis) and potential economic result (x-axis) from different use of the same forest area. Their calculations show that by reducing the demand to yield a *little*, the number of species protected increased significantly.

This trade-off pattern assumes the possibilities for substitution, and may be relevant for trade-offs between resources/services where none of the actual alternatives are critical to the ecosystems. On a micro level we will relatively seldom deal with resources /services that are "critical" from an economic perspective. However, the sum of many micro decisions may cross critical limits. This is a basic problem that can imply a need for more overarching frameworks.

The top right illustration in Figure 3.2 shows a situation where it is not possible to make a trade-off between the two services, within a reasonable interval (up to 100 points in the diagram). This means that we can increase our use of one ecosystem service without decreasing the use of the other. The analogy in practical management is that the provision of many services is not, or to a minor degree, directly connected. This can be interpreted as a case where we can increase the use based on one resource in a large interval, without affecting other services. If we use/take out more than 100 units in the diagram, the other service is nearly totally lost.

The illustration at bottom right in the same figure shows nearly exactly the opposite, the choice is roughly to choose between one use or the other. In other words, prioritizing one service (nearly) completely excludes prioritizing the other. In some cases we can meet these kinds of trade-offs in practice, for example by reserving one area to one kind of use which excludes other uses or services.

The final illustration in Figure 3.2 (bottom, left) shows a more or less linear relationship: "the exchange rate" between services is nearly the

[16] This line can be interpreted as typically for the "production possibility frontier"-curve, often presented in textbooks in economics.

same independent of how much we have of one or the other. This is not a typical connection between services.

Our discussion so far has used trade-offs between pairs of ecosystem services as examples. These kinds of trade-offs do not only exist between pairs of ecosystem services. They also exist between ecosystem services and other goods in society. Depending on how wide the definition of ecosystem services is made, we may consider trade-offs between ecosystem services or between ecosystem services and other goods and services (including abiotic natural resources).

Trade-offs over time

So far, we have discussed trade-offs between services and use of natural resources at the same point in time. However, different management strategies can give benefits and costs that arise at different points in time. Two aspects are of particular relevance in this discussion:

- *Discounting*: How should we discount the value of a benefit or cost that arises in the future, so that all costs and benefits that accrue in the near or distant future can be compared? And of particular relevance for management of nature and ecosystem services: How can this best be done when the effects potentially will arise in a distant future?
- *Real price adjustment*: The value of goods and services can develop differently over time for many reasons. For example, it is reason to believe that some environmental goods may increase their relative value in the future, due to increased scarcity. Also, people's preferences may change over time, influencing the relative prices. This factor may work both ways for ecosystem services, and the values for some ecosystem services may increase relative to other prices (and indexes for such goods, such as the Consumer Price Index) due to scarcity and changes in preferences while others may not.

We will not discuss the issue of trade-offs over time much further here, as they should be treated according to the guidelines for CBA the Nordic countries have developed, and which include guidelines for discount rates and real price adjustments. Also the need for real price adjustments are discussed in CBA guidelines in several countries, and the countries should check their own rules. This issue is first and foremost a case when using the ecosystem services framework in economic analysis where the value of ecosystem services over time is a question.

The issue is often less clear, also in CBA guidelines, for the so-called unpriced effects, for which ordinary discounting rules cannot be used. However, this is an issue beyond the scope of this report.

Trade-offs between interest groups

There may be several ways of dividing interest groups/interested parties/stake-holders which carry the costs or receive the benefits of a project or a management decision:

- Administrative (local communities, counties, region, country).
- Private and public sector.
- Economic sectors (fisheries, tourism, other sectors).
- Groups of people (for example based on sex, age, income, educations).
- Geographic (spatial).

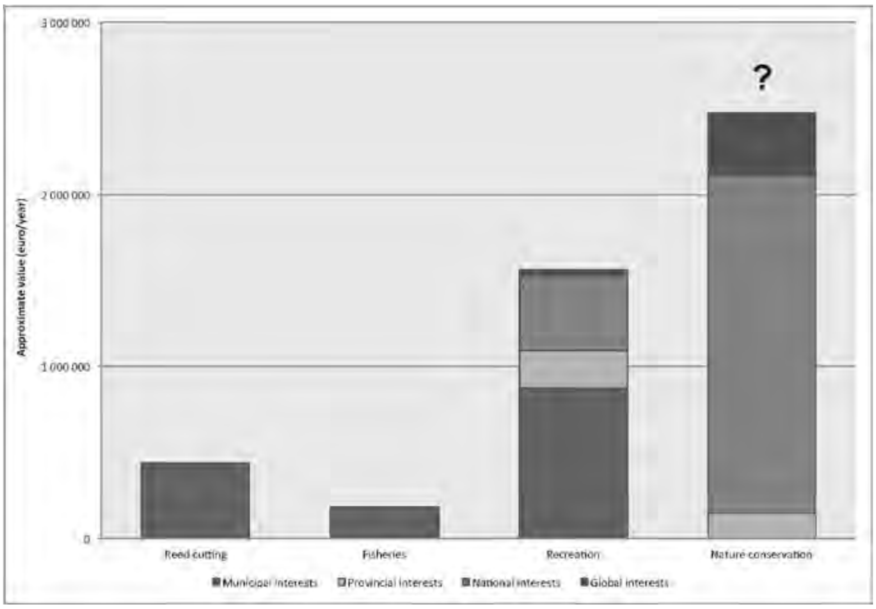
Distributional effects can be analysed and illustrated in many ways. The most advanced analyses use economic model tools to calculate how the effects are distributed in the economy for different groups. In many cases it is more realistic to make rougher assessments of which main groups that are affected.

It can be difficult to decide to what extent different groups are affected. An alternative framework can be to make a simple list of "winners" and "losers", as suggested for example in NOU (2012).

Hein *et al*. (2006) suggest a way of showing distributional effects for local, county, national and global interests. This example is shown in Figure 3.3 below. The example is about assessment of the values of the flow of ecosystem services from a wetland in the Netherlands.

The diagram shows relative value per year in Euros for different kinds of services (straw harvest, fisheries, recreation and nature preservation) and how these values are distributed between institutional levels. We can see that nature preservation mainly has a national value (and possibly global) while the provisioning services straw harvesting and fisheries benefit local groups.

Figure 3.3: The relationship between institutional scale and the value of ecosystem services



Source: Hein *et al*. (2006).

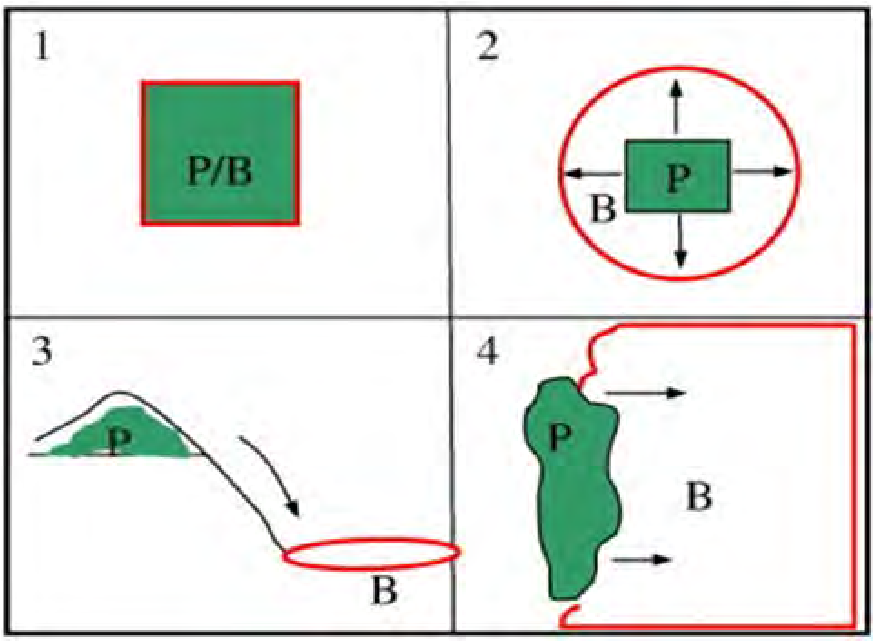
Spatial trade-offs and relationship between production and use of ecosystem services

A much used illustration for spatial relations between the place where an ecosystem service is produced and where it is used or exploited in shown in Figure 3.4 below. The place of production is indicated with "P" in the figure, while the place of ecosystem service use is indicated with "B" ("beneficiary"). The figure shows four possible spatial relationships. The first on top left in the figure indicates that a service is produced and used within the same, limited area. This will be the case where all the effects are local, for instance in a small lake. The other kind of relationship shows that the service is used by people outside the area in which the service is produced. This is typical for river basins or larger rivers, and for management decisions which affect ecosystem services of regional or national importance.

The two last illustrations (bottom of the figure) show that the benefits of services are provided to individuals outside and in a particular geographic direction from the production area. A typical example of situation 3 is when a wetland purifies water so that water quality is improved for the population downstream.

Spatial relationships between producers and consumers (beneficiaries) areas (which is analogous to assess spatial distribution of costs and benefits in a cost-benefit analysis) is an important part of the understanding of trade-offs or conflicts of interest. For example it may be more difficult to solve a conflict where the costs of production of improved services are carried locally or by a particular economic sector while the benefits are harvested somewhere else or of other groups of people or sectors.

Figure 3.4: Possible spatial relationships between production and use of ecosystem services



Source: Fisher *et al*. (2009).

3.2.4 The ecosystem service's share of final benefits and the need for other inputs

Sometimes we cannot use the ecosystem services directly, we have to add other inputs before the goods and services from nature are perceived as benefits to people. The clearest example of this may be agricultural production, where the soil is essential for production, but where lots of other inputs are added before we get the steak that gives us welfare.

Also ecosystem services from freshwater may need other inputs, for instance fishing equipment and man time, water treatment plants before we can drink the water from the tap etc. If the purpose is to demonstrate the values we receive from ecosystems, the presence and amount of these other inputs are not so important.

However, in assessing costs and benefits in a CBA, for example in order to assess disproportional costs, we need to be more precise. If we include the total value of fish as a benefit we receive from ecosystems, we overestimate the role of ecosystems. For example for benefits such as drinking water, flood control etc., which we receive from fresh water, a considerable amount of "other inputs" may be necessary in order to give us the benefits. For other ecosystem services from freshwater the benefits are received more or less without any other inputs. In CBA the net benefits of the ecosystem services should be included implying that the total value of the ecosystem services is the value of the final good minus the value of the other inputs to produce the final good (like labour/time and equipment/capital).

3.2.5 Double counting

Related to the question above, is the issue of double counting. One topic is that economists prefer to value the final goods only, in order to avoid double counting if both intermediate and final goods are valued. That is, we should be careful not to value and include both the functions of the ecosystems and the final services. It is worth remembering that this question depends on whether we want to analyse the trade-offs between ES's or the total value of the final goods.

Another concern is that because our valuation methods are less than perfect, we may not value exactly the ecosystem service we aim at valuing, but rather a "bundle" of ecosystem services, or even indicators. This is of particular concern if we aggregate value estimates for several ecosystem services one by one. It is also relevant if we have values for some ecosystem services, for example from a contingent valuation study of improved water quality, and then add values for other services which were not directly included in the Contingent valuation study, like regulating services, provision of drinking water etc. Then we should take care that we do not include the value of the same ecosystem services twice (totally or partially) and add up.

The double counting issue has been discussed to some extent in the economic literature (e.g. Boyde and Banzhaf 2007). We should also note that many (perhaps most) of the ecosystem services are in many cases not valued at all, so that the danger of "zero-counting" is also present. However, particularly, when value estimates are used in CBA, for assessing disproportionate costs, or for PES and other economic instruments, we should take care to avoid double counting.

3.3 Examples of identification and mapping

3.3.1 Introducing the examples

In order to assess the benefits we receive from improved water status in fresh water, we need to identify the ecosystem services which are supplied by freshwater in a country, in a region, river basin, river, or in a specific water body area. Sometimes the purpose may be to recognize or demonstrate all the goods and benefits we receive. More often, this identification is used as a point of departure for more quantification and valuation, or for demonstration of trade-offs or implications of policy options.

In this section we will show some examples where the main purpose has been to illustrate and map which ecosystem services we receive from fresh water. Relating to the steps in section 3.1., these examples focus on identifying which ecosystem services are found or are improved due to improved water status, on different scales. The level of detail can be different, depending on the purpose, the scale and the present knowledge.

As an introduction we present two examples which map ecosystem services on a broader scale than just ecosystem services from fresh water. These are included because we see them as an elegant way to present how ecosystem services are distributed in the landscape, and because we believe a similar assessment could be carried out for river basins (or other scales relevant for water management).

In the next section we present an interesting Finnish example that use some of the same techniques in order to map the linkage between biodiversity information with the landscape's capacities to provide ecosystem services, the example we include shows examples for water provisioning. This kind of mapping is not so common yet, but much work is expected in this area the coming years.

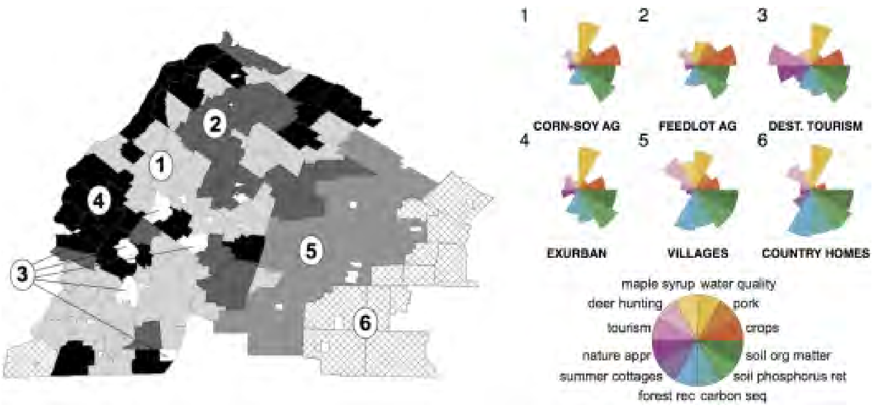
The more factual examples are from Finland and Norway, illustrating different scale and detail. The first one is from a Finnish report (Alahuhta *et al*. 2013), in which the authors discuss the general categories of ecosystem services and which ones are relevant for freshwater in Finland on a national scale. They also illustrate the richness of waters and the water regions in Finland as background for the categorisation. The last example is from a study on a local scale (water body/river) in the urban Oslo area in Norway where the potentially affected ecosystem services from measures to improve water status according to WFD was investigated (Magnussen *et al*. 2014). They used a listing of ecosystem services in fresh water as a starting point for identifying which ecosystem services would actually be affected by the planned measures in these rivers according to the WFD.

3.3.2 Examples

Illustration of mapped flow of ecosystem services

Below we show an illustration of mapped assessment of average values of ecosystem services per unit of area for a landscape (Figure 3.5). First, the ecosystem services are divided into “service bundles”, with a content of hunting, carbon sequestration, tourism, etc., which is indicated in the circle at the bottom right of the figure. Then the areas are classified in six different kinds at the right side in the figure, depending on which of the services dominate (as indicated in each of the six circles). These six kinds of areas can be retrieved on the map on the left with their specific number. This is an elegant way to illustrate how ecosystem services are distributed in the landscape, by this kind of management which gives a very heterogeneous exploitation. A similar assessment could be carried out for river basins. However, we would expect that the use would not show the same heterogeneity or diverseness of services.

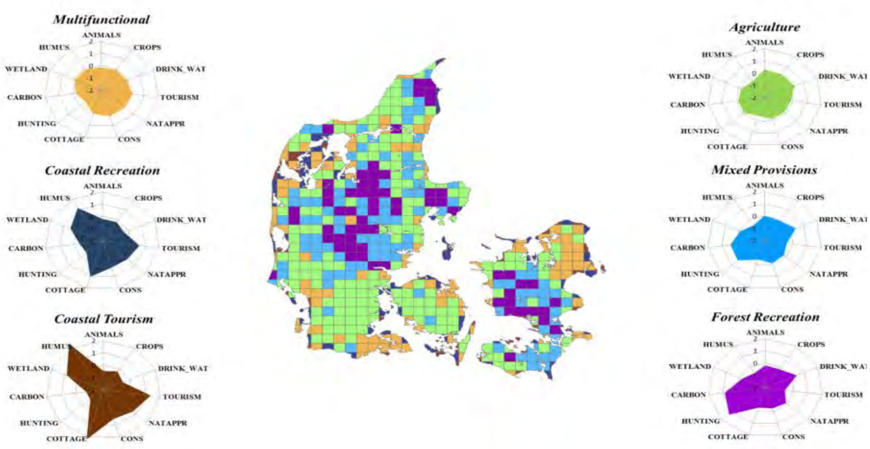
Figure 3.5: Average values of bundles of ecosystem services in a landscape



Source: Raudsepp-Hearne (2010).

The same type of analysis bundling ecosystem services in Danish landscapes is performed by Turner *et al.* (2014). Figure 3.6 illustrates Turner *et al.*’s (2014) assessment of 11 ecosystem services that they mapped at 10 km × 10 km grid scale, while covering most of Denmark. Their aim is to describe the spatial distribution as well as the interactions between the ecosystem services. They identified trade-offs between regulatory and cultural services on one hand and provisioning services on the other hand. The figure shows the identification of six ecosystem service bundle types which indicate interactions at landscape level. The analysis reveals, taking the underlying data and assumptions for granted, that there is a large potential for recreation at the coast, indicated by summer cottages and tourism, while there are more multifunctional mixed-use bundle types around urban areas. As mentioned by the authors the bundling results are sensitive to the input data, the indicators, available to define the services.

Figure 3.6: Ecosystem service bundle types

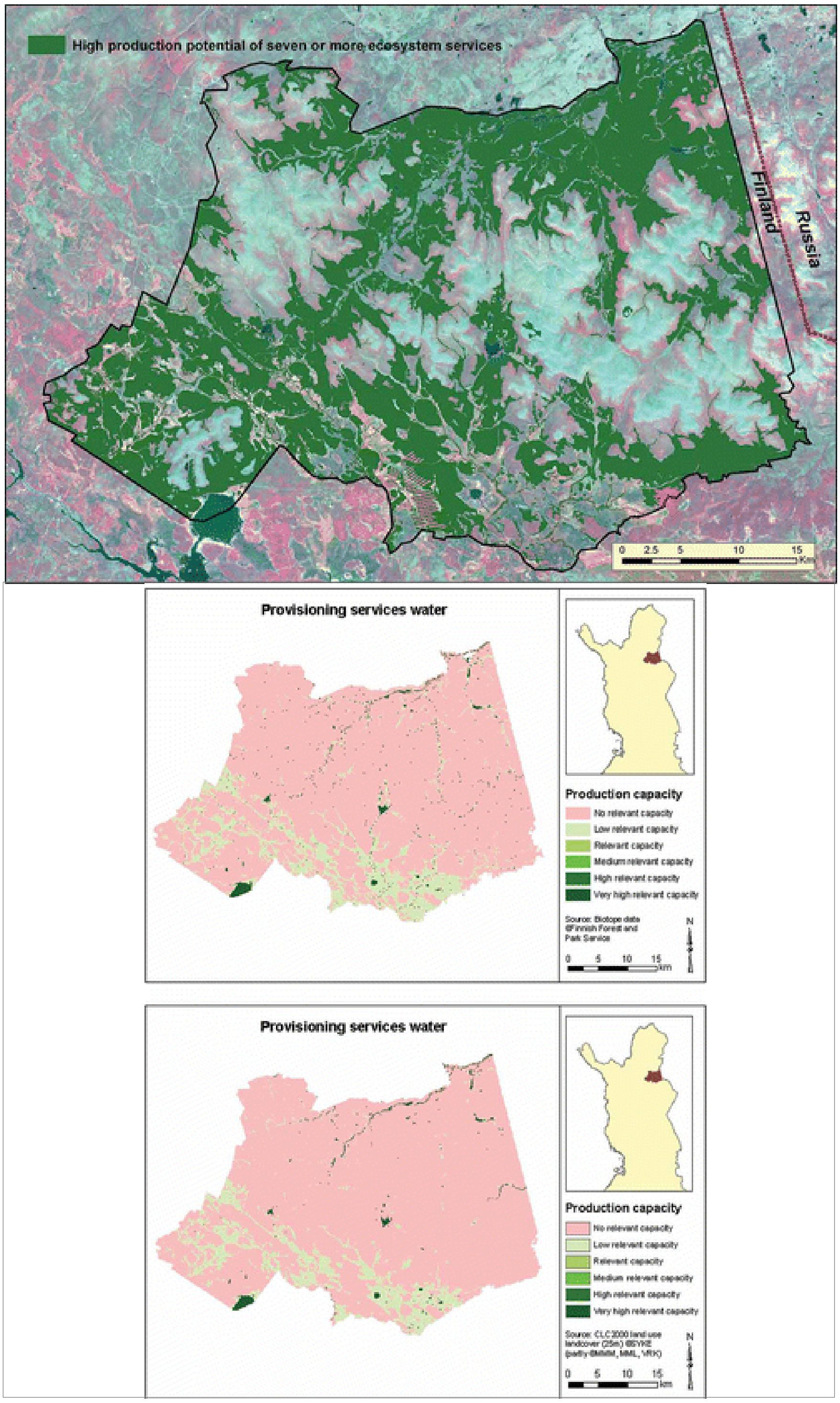


Source: Turner *et al*. 2014, page 96.

Mapping of ecosystem services in natural protection areas in northern Finland

Vihervaaraa *et al*. (2012) used detailed tools such as aerial photographs and field surveys to produce high-quality biotope data to map ecosystem services in natural protection areas in northern Finland. The use of detailed biotope data supports the linkage of biodiversity information with landscapes' capacities to provide ecosystem services. Figure 3.7. shows examples for water provisioning in the bottom, and the mapping of six different services at the top (the green map), as an example of how trade-offs and conflicts between ecosystem services can be mapped.

Figure 3.7: Mapping of ecosystem services in natural protection areas in Northern Finland

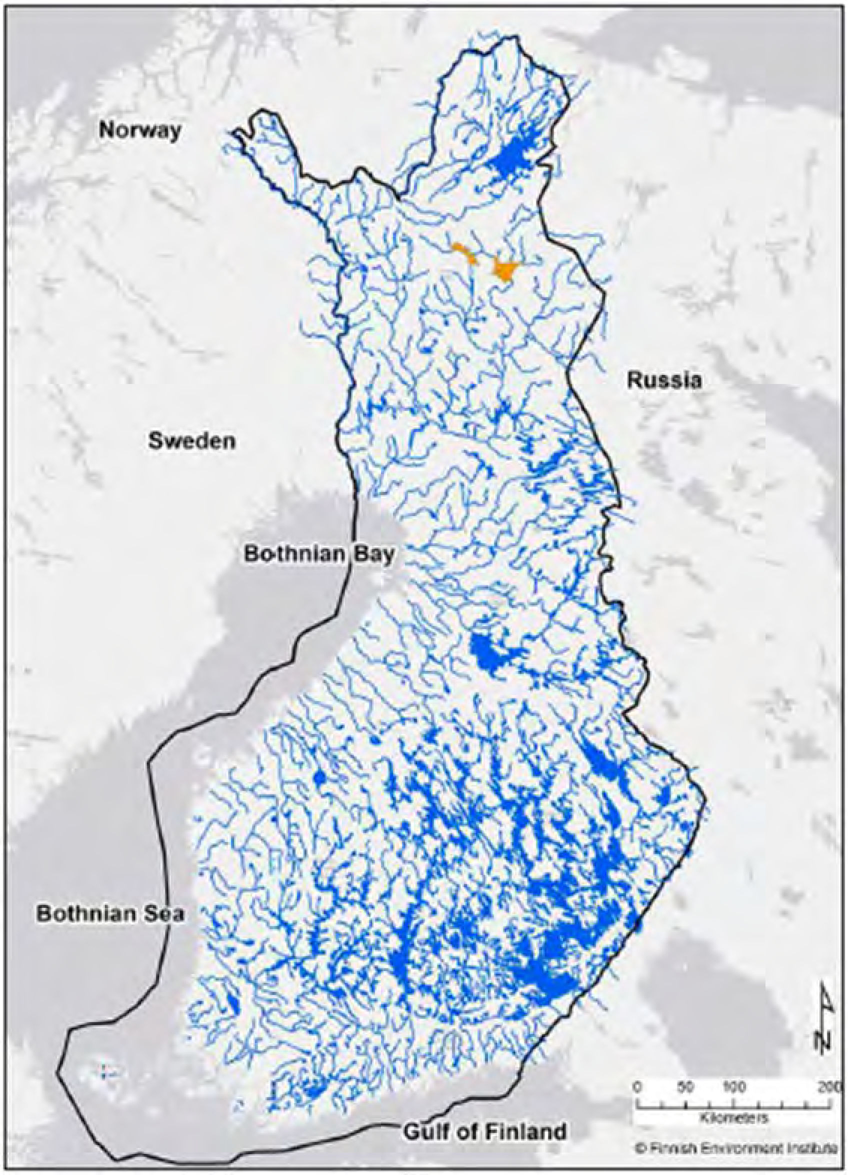


Source: Vihervaaraaa *et al*. 2012.

Identification of freshwater ecosystem services on country scale in Finland

In a report from the Finnish Environment Institue (Alahuhta *et al*. 2013) the authors compile the first systematic suggestion for the classification of the boreal freshwater ecosystem services in Finland. Figure 3.8 below illustrates the richness of inland waters of Finland.

Figure 3.8: The inland waters of Finland. Two major, artificial reservoirs are marked in orange



Source: Alahuhta *et al*. 2013

The report discusses the different categories of freshwater ES, and suggest the following ecosystem services categories to be used, as shown in table 3.1.

Table 3.1: Suggested freshwater ecosystem services in Finland

Freshwater ecosystem services			
Provisioning	**Regulating**	**Cultural**	**Supporting**
Food	Macro-climate regulation (!)	Recreation	Nutrient cycling
Clean water (!)	Micro-climate regulation	Aesthetic value (!)	Soil formation (!)
Energy	Air quality regulation	Cultural heritage (!)	Food web dynamics (!)
Transportation	Water flow regulation	Science and education (!)	Habitat (!)
Biochemical resources (!)	Water purification	Inspirational value (!)	Primary production
Ornamental resources (!)	Invasion resistance (!)		Photosynthesis
Construction	Disease regulation (!)		Water cycling (!)
Genetic resources (!)	Seed dispersal and pollination (!)		
	Erosion regulation (!)		
	Natural hazard regulation		

The mark (!) in the columns means that more research is needed to confirm and understand the meaning of that particular service as a frehwater ecosystem service in boreal region, according to the source.
Source: Alahuhta *et al*. 2013.

Identification of freshwater ecosystem services on water body scale in local rivers in urban Oslo area, Norway

The Norwegian NOU (2013:10) on ecosystem services (see description in section 2.2.) does not make a listing of all the potential ecosystem services in freshwater, however it mentions the main ecosystem services we receive from different ecosystems. An example of practical use of the classification is a recent Norwegian case study which analyse how benefits and (disproportional) costs to society can be assessed in urban rivers (on the water body level) using the ecosystem services framework for benefit assessment (Magnussen *et al*. 2014). As part of the assessment, a list was first made of all potential ecosystem services that can be derived from rivers and the fjord recipients, using the Norwegian NOU-ecosystem services-categorisation. Magnussen *et al.* (2014) then make a shortlist of which ecosystem services will actually be affected by the water status improvement, given the measures suggested according to the program of measures developed for the area under the Water Framework Directive, see table 3.2.

Table 3.2: Identified ecosystem services potentially relevant/important for assessment of improved water status following program of measures according to WFD in Hovinbekken and Alna rivers in urban Oslo, Norway

Basic life processes (Supporting ES)	Potentially important	Relevant/important for assessment of measures?		
		Hovinbekken	Alna	The Oslofjord
Water cycle	Yes	Yes	Yes	Yes
Evolutionary processes and ecological interactions	Yes	?	?	?
Regulating ES				
Water flow regulation	Yes	Yes	Yes	
Erosion protection	Yes	Yes	Yes	Yes
Nature damage protection	Yes	Yes	Yes	Yes
Water cleaning and waste treatment	Yes	Yes	Yes	Yes
Regulation of diseases	Yes	Yes	Yes	Yes
Pest regulation and biological control	Yes	?	?	?
Provisioning services				
Fresh water	Yes	Yes	Yes	Yes
Genetic resources	Yes	?	?	?
Cultural services				
Recreation and nature based tourism	Yes	Yes	Yes	Yes
Wellbeing and aesthetic values	Yes	Yes	Yes	Yes
Local identity	Yes	Yes	Yes	Yes
Inspiration and symbolic perspectives	Yes	Yes	Yes	Yes
Knowledge and learning	Yes	Yes	Yes	Yes
Natural heritage	Yes	Yes	Yes	Yes

Source: Magnussen *et al*. (2014).

We come back to these following steps in section 3.4 and chapter 4 when we discuss quantification and valuation, and assessment of disproportional costs.

3.4 Examples of quantification and valuation

3.4.1 Introducing the examples

In this section, we will give some examples of how ecosystem services are, or could be, quantified and valued in the Nordic countries. We start with how ecosystem services from fresh water was summarized in the "Nordic TEEB", that is for a region including all the Nordic countries. The Nordic TEEB did not quantify or value all ecosystem services from Nordic fresh water. However, they suggest some indicators that could be used across the Nordic region in order to value these ES. This is seen as an interesting suggestion, as it is supposed to hold true for all the Nordic countries, and because, as this section will indicate; there is still a lot of work left to do when it comes to quantification and valuation of Nordic freshwater ES.

We then move on to how quantification and valuation can be done within a country, on a country scale, river basin scale, local scale or for one ecosystem services in particular. Although the focus of this report is how the ecosystem services framework can be used in water management, we will also include some recent studies which have not used the ecosystem services framework directly, but where the ecosystem services categories valued are recognizable, and where the valuation is relevant for water management according to WFD. We see these examples as highly relevant in our context because there are so few examples which follow the ecosystem services framework so far – due to its quite recent break-through. And even if the study does not mention ecosystem services as such, often improvements in a bundle of ecosystem services are assessed in these studies, and therefore one can go back to these studies and identify which ecosystem services were valued. Therefore, being able to exploit this information will be important in order to move forward in using the ecosystem services framework in water management. We have focused on including recent work in our examples; this is again because the VALUESHED report (Barton *et al*. 2012) gave an overview of Nordic studies of interest carried out until 2011.

In the next section we describe a valuation study of ecosystem services from the wetland area Store Åmose in Denmark. We should note that this study even included cultural heritage as an ecosystem services connected to wetland restoration. The different ecosystem services were valued using a Choice Experiment, so that the values of the different ecosystem services could be estimated. These benefit valuation results were also used in a CBA. This study is related to a wetland, but the goals are not directly connected to WFD, and the benefit and cost estimates are used in a traditional CBA, but not for assessment of potential disproportional costs. This is one of relatively few examples where ecosystem services for wetlands are valued and used in a CBA.

We then briefly discuss the case study in Odense river basin where the benefits of reaching GES was investigated. This example was described in some detail in Barton *et al*. 2012, and therefore not repeated here. However the study is briefly mentioned and the information used for the valuation survey is presented. This is because this case-study (together with the case study in Morsa, which was also presented in detail in Barton *et al*. 2012), are still some of the most comprehensive studies of the benefits of reaching GES according to WFD framework.

The Odense river study (and Morsa) did not use the ES framework per se, but the goods and services valued are easily recognisable in an ecosystem services terminology.

The Odense and Morsa valuation results are used for benefit transfer to nearly all Swedish rivers in a recent study, which is presented briefly. They are used to give estimates for the benefits of reaching the WFD goals of GES in Swedish rivers.

We then move on to two new valuation studies of reaching GES in Finnish water. These studies do not use the ecosystem services terminology directly in the valuation, but again the goods and services valued are easily recognisable in ecosystem services terminology. As far as we know, these studies are two of the very most recent valuation studies undertaken in the Nordic countries, and there of great interest event if they do not use the ecosystem services terminology directly. The valuation results in one of the Finnish studies are used to compare benefits and costs of carrying out programmes of measures, and the benefits are found to be much higher than the costs.

In the next example the scale is very local – down to the water body level in WFD terminology. This study does not value ecosystem services itself, but uses all steps identified in section 2.1 to identify, quantify and put a monetary value on the benefits, expressed as improvements in identified ecosystem services. This case study therefore is one of the few we have found that try to follow the ES framework in order to assess the benefits of water status improvements following the fulfilment of the WFD goals.

Most of the studies discussed so far in the report assess the bundle of ecosystem services that are affected by water status improvements. However, there are also some examples that focus on one ecosystem service only. Barton *et al*. (2012) and the national reports about the ES framework, include lists of valuation studies of separate ES, e.g. recreational fishing, and flow reduction. Many of these were carried out before the ES framework was much discussed, but are still relevant. We will not repeat these here. However, following the interest for ecosystem services, and following the foundations for the ES framework (see figure 2.1) there seem to be a growing interest to dig deeper into the biotic/ecosystem functions and combine with assessment of the value of the benefits these provide society. We included one example of mapping of water provision based on landscape and biodiversity in section 3.3. In this section we include one example which shows the linkage between mapping and estimating the scientific changes in retention due to different agricultural practices and the values that can be attached to these ecosystem functions.

3.4.2 *Examples*

The Nordic TEEB – suggested indicators for assessment of ecosystem services

The Nordic TEEB (Kettunen *et al*. 2013) presents the results of a synthesis on the socio-economic importance of biodiversity and ecosystem services in the Nordic countries. It also gives a synthesis of the (at the time) existing information on the status, trends and value of Nordic ecosystem services, identifies gaps in existing information and knowledge and develop recommendations for key future policy action on ecosystem services in the Nordic countries.

When it comes to ecosystem services from fresh water the report states that professional freshwater fishing in the Nordic countries is small compared to marine fishing, and mainly taking place in Finland and Sweden where fishing is concentrated to a few big lakes. There are some 56 fish species in Nordic countries' freshwater, of which 13 are introduced species. The main problem with fresh water fishing lie in water quality issues (eutrophication and acidification) and not som much in overfishing, according to Kettunen *et al*. (2013).

The other importan freshwater provisioning service mentioned is drinking water, of which the Nordic countries have abundant resources compared to other European countries.

The freshwater's importance for water purification and nutrient retention is acknowledged. Generelly speaking, it is said that in the Nordic countries the state and quality of the water ecosystems has been relatively good compared to Central and Southern Europe. However, water quality has decreased in many parts of the Nordic region due to agricultural loads, ditching of forests and drainage of mire either for forestry or peat production, or due to nutrient loads from point sources.
Recreation and supporting ecosystem services are not described in detail for freshwater ecosystem services in the Nordic TEEB.

In the table below we have listed some of the major fresh water ecosystem services identified in the Nordic TEEB, and the identified direct and proxy indicator suggested used to estimate the socio-economic value of Nordic ecosystem services.

Table 3.3: List of identified direct and proxy indicators to estimate socio-economic value of Nordic ecosystem services, suitable to be explored to be adopted at national level. Note that the table does not address the issue of double counting which needs to be considered when calculating aggretate values for multiple ecosystem services

Ecosystem service in fresh water	Identified direct indicator	Identified proxies
Fishing: fresh water	(Market) value/value added of catch (sustainable) Number of jobs/employment/ businesses/income	Size/value of catch (current amount or value) Number/% of fish and other species in commercial use
Aquaculture	(Market) value/value added of catch (sustainable) Number of jobs/employment/ businesses/income	Amount of cultured fish and other species (current)
Fresh water (provisioning of:) drinking and potable water, water for other human consumption	(Market) value/value added of (drinking) water, adjusted to reflect the real value (remove effects of any dirstorting subidies)	Population/business served by renewable water sources
Carbon sequestion and storage	Value of carbon sequestrion and storage (e.g. based on CO_2-markets)	Costs related to climate change (real or estimated), e.g. based on costs of climate induced natural disasters
Natural hazards: Mud flow/flood	Value of protective function Replacement/avoided costs	Economic losses associated with mud flow (real or estimated) Population living in areas depending (directly) on ecosystem based regulation
Water and water flow: drainage and stabilisation of water flow (non-flood related)	Difficult to find a reasonably independant indicator, mostly integrated into values below	Not identified
Water and water flow: Drought mitigation	Value of protective function Replacement/ avoided csots	Economic losses associated with droughts (real or estimated). Population living in areas depending (directly) on ecosystem based regulation (i.e. drought risk areas)
Water and water flow: Irrigation	Value of protective function Replacement/ avoided costs	Not identified
Water and water flow: Aquifer recharge	(Market) value of water originating from aquifers, adjusted to reflect the real value (remove effects of any distorting subsidies) Replacement/ avoided costs	Economic losses associated with lack of ground water (real or estimated). Population living in areas depending (directly) on ecosystem based regulation)
Water retention and purification and waste treatment	Value of regualting and protective function Replacement/ avoided cost	Economic losses associated with lack of water quality (real or estimated). Population living in areas depending (directly) on ecosystem based regulation

Ecosystem service in fresh water	Identified direct indicator	Identified proxies
Recreation and tourism related to fishing	Money/time invested in carrying out activities (e.g. travel costs, fishing licences, equipment) Number of recreation fishermen/ fishing licences	Value of service based on stated preference methods (e.g. willingness to pay derived via contingent valuation) and revealed preference methods (e.g. travel cost methods) General investment in the conservation /restoration of natural areas, e.g. local/regional/state budgets for maintenance of green areas, extension of national and nature parks/protected areas, afforestation etc.

Source: Modified from table 10.1. in Kettunen *et al*. 2013.

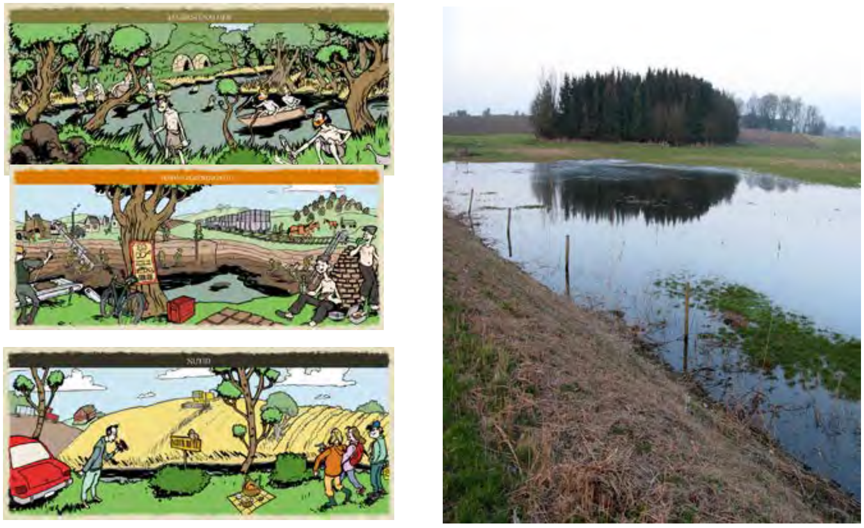
Valuation of ecosystem services from wetland area Store Åmose in Denmark – Valuation and CBA, but not WFD

The wetland area "Store Åmose" has delivered different services to humans throughout history, and therefore one of the services delivered by restoration of the wetlands in this area is the protection of cultural heritage and artefacts. In the stone-age the area was a good place to live for hunters, and there are a lot of artefacts buried in the top soil. These artefacts are protected by the wetland so that future generations can experience these artefacts if they are unearthed then.

In the 19th and 20th century the area, like many other wetlands, was used for energy production. After the 2nd world war the area was dried, a channel was built and the area was used for agricultural production. In the last 15 years parts of the area has been protected and some of the wetland has been restored , hereby protecting nature in the areas, protecting the cultural heritage and creating recreation possibilities for people. This historical development is illustrated in figure 3.9 below.

Figure 3.9: Store Åmose – uses and development over time

Danish valuation study of a wetland



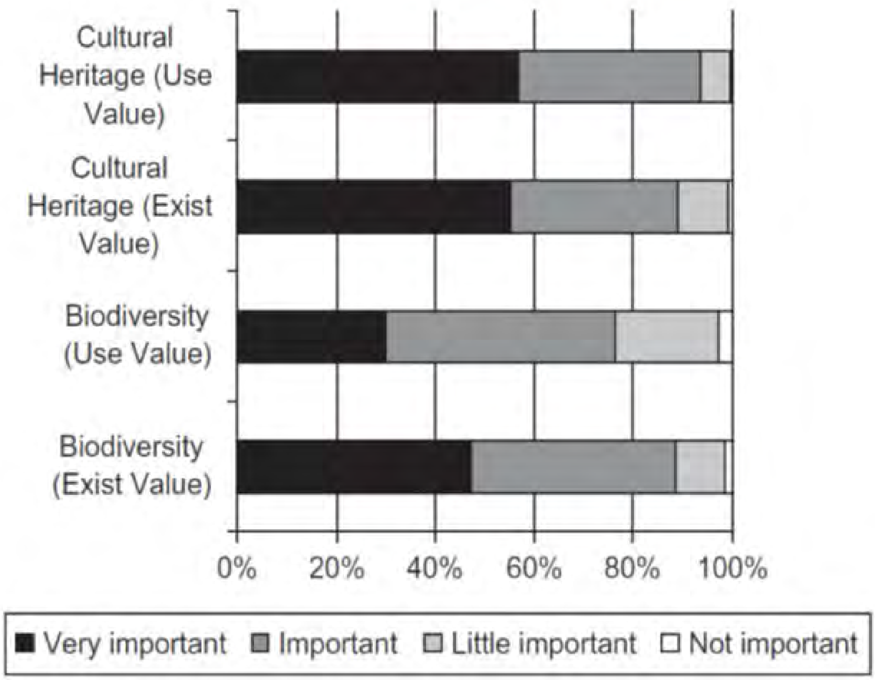
Source:http://www.aabne-samlinger.dk/svm/skoletjenesten/kulturkoerekort/pdf_aamosen/Laerervejledning_aamosen.pdf

Wetland restoration has been proposed for this area to protect biodiversity, the artefacts and also to improve the recreational opportunities in the area. This means that wetland restoration in this area have potentially a number of ecosystem services with value for society. If no wetland preservation takes place the artefacts and also the biodiversity in the area is threatened by destruction, because agricultural practices, with artificially low water levels and processing of the soil, cause oxygenation of the soil. The Nature Protection Agency anticipated in an action plan for the Åmose; “Åmose – the green heart of West-Zealand” in 2001 that the value of the recreational area as well as the nature protection would be high.

Three different scenarios were proposed, where these three scenarios differed in how large area would be protected. The protection should take place by restoring wetlands by raising the water level by cutting off drains and dikes and damming up watercourses. The recreational assets were anticipated from paths, information signs, and a visitors’ center. In 2005–2006 a valuation study was made by researchers at the former National Environmental Research Institute and the former AKF (the institute for Local Governmental research) (Lundhede *et al*. 2006; Lundhede *et al*. 2013). The researchers tried to elicit the value of the different ecosystem services of the wetland creation: the biodiversity protection and improvement, the protection of the artefacts and the improved recreational possibilities, taking departure in the three scenarios for the Åmose. The valuation method used was Choice Experiments, which allowed for trade-offs between the different services. The nutrient uptake by the wetland was not assessed in the analysis.

The results indicate that the respondents care much about protection of the artefacts, even if they cannot be seen and experienced now, and also for the biodiversity of the area. But the major proportion of the respondents did not care about improved access to the area; i.e. these cultural services did not have a value for the respondents. The sampling was the whole country, to explore the value of the ecosystem services of this area among the Danish population. These results from the valuation study are summarized in figure 3.10.

Figure 3.10 Restoring wetland Store Åmose: Results of a valuation study, showing how important the respondents meant different values were



Source: Lundhede *et al*. 2013.

As shown in figure 3.10 the value of the cultural service, i.e. the protection of the artefacts, is considered higher among the interviewees than protection of the biodiversity in the area.

The results from the valuation study have been used for a CBA. The CBA of this project shows how sensitivity analysis can be done to explore the certainty of the results. The CBA addressed the three different scenarios for the protection of the Store Åmose. The CBA indicate that each of the three Åmose scenarios gives a substantial welfare-economic surplus; the costs of the restoration are not very high, except for the lost agricultural production and the visitor center there are no costs and the benefits from protecting both artefacts and biodiversity was found to be high. The CBA showed that the overall welfare is improved significantly by completing each of the three scenarios, and the large welfare-economic surplus is primarily explained by the values connected to the improvement of the cultural historic and biological assets. Since these benefits exceeded the costs with a very high ratio the Ministry of Environment (2005) decided to base their social-economic analysis of the wetland project on a break-even price.

This assessment of the break-even price was done calculating all the other costs and benefits, and then the necessary value of improving the biological, cultural historic and recreational services were added in order for the project to result in a welfare economic surplus. In this way the Ministry of Environment made it possible to compare the break-even price with the results of the valuation study, and they also claim that this method can be used to assess "whether the uncertainty in the valuation is significant enough to affect the conclusion on the welfare-economic surplus from completing each of the three scenarios" (Ministry of the Environment 2005). The results indicate that the break-even price should be at least DKK 56 mill, DKK 85 mill, and DKK 59 mill, respectively, at present value for the three scenarios, if there is to be a welfare-economic surplus from completing the project. These numbers are much lower than the willingness-to-pay results from the valuation study (Lundhede *et al*. 2013). The Ministry therefore concluded that the welfare gain would be high.

In addition to this conclusion, it is of importance to notice that even if the valuation result might be overestimated because the sampling area was the whole Denmark and no substitute sites were mentioned, the trade-offs between the different services provide valuable information of which services of a wetland restoration project like this are most valuable.

Odense river basin – valuation according to GES, not ecosystem services per se – used for benefit transfer in Denmark for screening of disproportional costs and to Sweden for assessment of benefits of reaching WFD goals

The benefits of reaching good ecological status (GES) were investigated in Odense catchment as part of the Aquamoney project (2009). This case was presented in some detail in Barton *et al*. 2012, and we will not repeat the case study description here. However, we will present some important aspects of the study, as this is one of the first and most comprehensive studies used to value improved water status according to WFD. It did not use the ecosystem services notion per se. The goods and services valued, however, are easily recognizable in an ecosystem services terminology.

In the textbox below we introduce the information used for the valuation survey, referring to the water status classes in WFD, and how the water status classes were described in the Odense survey.

Box 3.5: Illustration of water status classes according to WFD in the Aquamoney survey in Odense

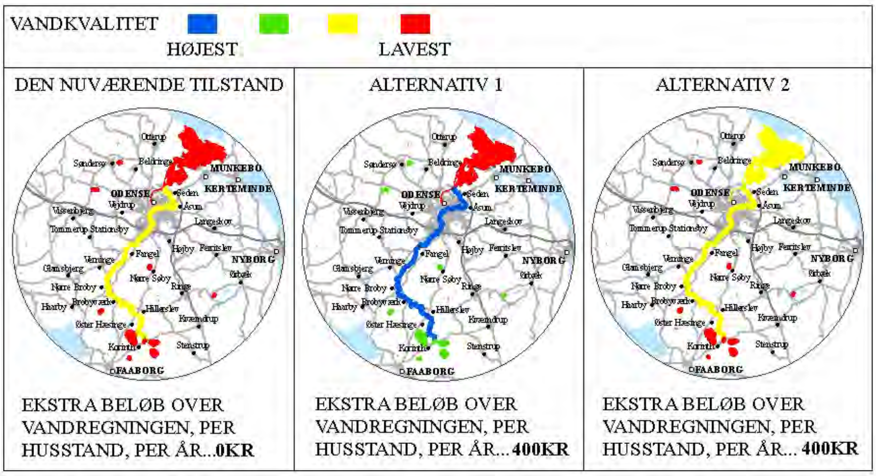
	Highest quality
This picture indicates a river in its highest quality. The water is suitable for boating, angling and swimming. The water is suitable for fish, plant and bird species being present under natural river conditions.	
This picture indicates a river where the water is suitble for boating, swimming and angling, even though the most pollution sensitive fish can be absent under these conditions. The diversity of birds and plants is somewhat lower than compared to the highest water quality.	
This picture indicates a river where the water is suitable for boating, but the possibilities for swimming and angling are more limited. Pollution sensitive fish species are present as they are artificially planted out. The presence of fish, birds and plant species are limited.	
This picture indicates a river where the water is not suitble for boating, swimming and angling. The presence of bird and plants are very limited, and there is few or no fish.	
	Lowest quality

Source: Hasler *et al*. 2010.

As can be seen the four different water quality classes (poor and very bad water quality was merged into one class) the water quality is described by the four pictures illustrating the water clarity and the biodiversity in the water as well as the landscape of the land close to the water, i.e. the river bank, or the coastal area. Furthermore the text related to the four pictures describe what the water is suitable for, highlighting both the use of the water for recreation as well as the fish composition in the recipient of this quality. The usability of the water is also indicated by the icons used for each of the four water quality classes.

The information was linked to monitoring results and specific targets for nutrient content etc. in the water bodies necessary to achieve good ecological status. The information about the present situation (quality) was used to characterise the status quo in maps, see figure 3.11. below. Then the present water quality was presented together with two other alternatives for water quality improvements of the water bodies in the catchment. Hereby the valuation design facilitated spatial assessments of the value of the improvements.

Figure 3.11: Presentation of the present situation and two alternative scenarios in the Aquamoney survey in Odense



Souce Hasler *et al*. 2010.

In this way the description of the water quality highlights the ecosystem services provided when the water has different status according to WFD, and hence this is an example of how the WFD water quality classification can be linked to ecosystem services assessments. The Odense study is further discussed in chapter 4 as the study results have been used for a cost benefit analysis and screening of disproportionate costs (cf. Jensen *et al*. 2013, presented in Chapter 4).

Assessment of the benefits of reaching good ecological status in Swedish river basins based on benefit transfer from Odense (Denmark) and Morsa (Norway)

This study is based on benefit transfer from Odense river basin in Denmark (see examples above) and Morsa in Norway (the two case-study-areas in Barton *et al*. 2012 ("VALUESHEDS") to most of the Swedish river basins ("åtgärdsområder"). Both the Morsa and the Odense study used both Choice Experiment (CE) and Contingent Valuation (CV), and the CE part was used for the transfer to the Swedish areas. Furthermore the differences between the Norwegian and the Danish results were used to construct an interval for the transfer. The purpose of the Swedish study was to estimate the value of good ecological status (and if possible also high ecological status) for as many as possible of Swedish river basins.

The authors underline that the estimates are not meant to be used for policy decisions for example to assess disproportionate costs. However, they may be used in order to point to the areas where further analyses are needed. The study naturally does not say more about ecosystem services than the Odense and Morsa studies do, and as mentioned even if different ecosystem services are described as part of the water quality description the valuation did not focus on ecosystem services per se.

The benefits of reaching good ecological status in local water course in Lake Vesijärvi, Finland – Valuation, not ecosystem services per se, but ecosystem services are recognizable, compare benefits and costs, but not disproportional costs

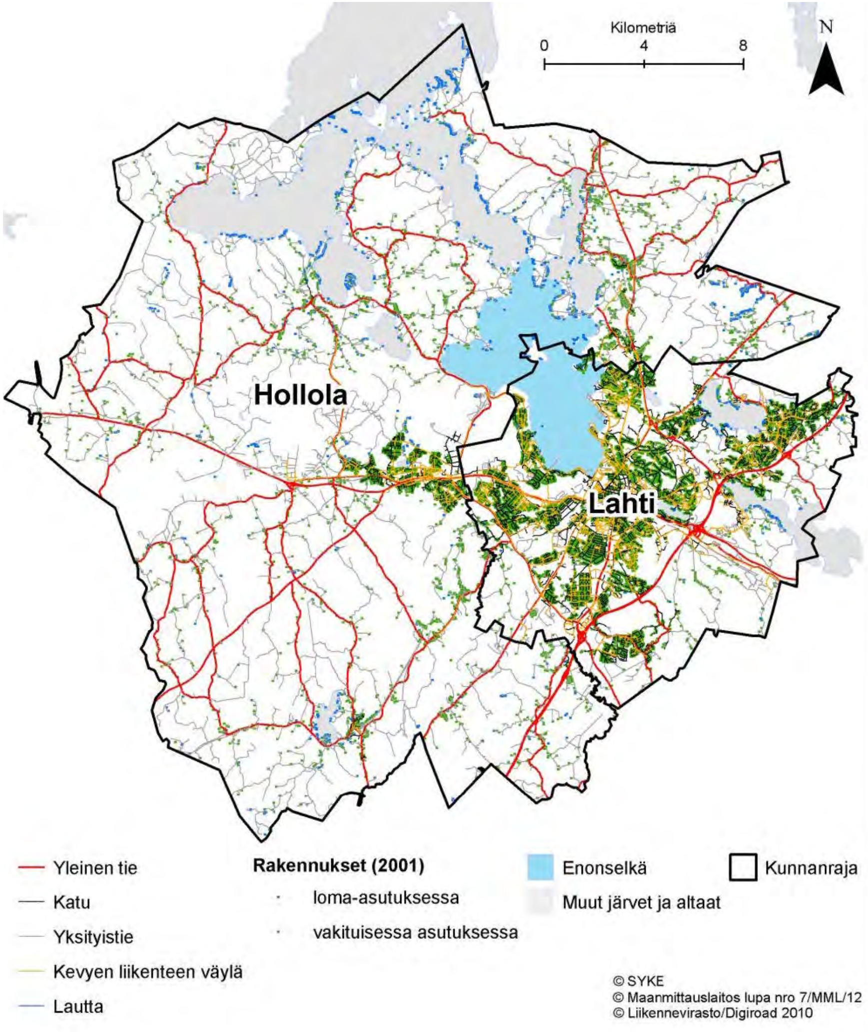
The purpose of this study was to estimate the welfare effects of water management in monetary units (Lehtoranta 2013). The demand for water management was surveyed in the city of Lahti and the local community Hollola in a situation where water status in Lake Vesijärvi would be improved from moderate to good. The study area is whown in figure 3.12. In the survey the Contingent Valuation (CV) method was applied. The households were asked about their willingness-to-pay (WTP) for improved water quality to gain increased recrational values (several forms of recreation uses, these were not specified in the survey). The survey was carried out as a mail survey sent to 2,550 random households in Lahtis and Hollola. Out of these, 1,362 (54% of the sample) answers were completed and could be analysed.

The probability that the households would pay for water quality improvement was investigated using regression analysis. The most important reason for people to pay was the wish to preserve water in a good status for the future generation.

The households were willing to pay the following five years an average of EUR 11 with certainty and EUR 23 with “quite high certainty”. The total benefits for improvement from moderate to good status was estimated to be more than 1.3–2.5 the yearly costs needed to reach the water quality improvments in Lake Vesijärvi. Hence the benefit and cost estimates were compared at water body level.

Households with higher income and housholds who believed in the payment mechanism used (payment to a water improvement foundation that would carry out the necessary water improvement measures) were willing to pay more than households with smaller incomes and who doubted the foundation’s work. The belief that the suggested restauration methods will work also increase people’s willingness to pay.

Figure 3.12: Study area: Enonselkä basin is the largest basin of Lake Vesijärvi



Source: Lehtoranta 2013.

The benefits of improved water status in local water course in northern Finland – Valuation, not ecosystem services per se, but ecosystem services are recognisable

This survey responded to a need by the Kello village association and the Kiiminki-Jääli water management association to consult local residents on the goals of, and the willingness to participate in water management in the river Kalmenjoki catchment area (Lehtoranta *et al*. 2013), see figure 3.13. The scope level was the river catchment area. As a research method, the questionnaire-based contingent valuation method, the most common framework taken to economic valuation on non-market resources, was employed. Two versions of the questionnarie were created and were sent to 1,632 households in the catchment area. Half of the households were asked about their hypothetical willingness-to-pay (WTP) to the water management association while the other half were presented with an actual option to pay for water management. The benefits gained by way of implementing the restoration plan were explained in the questionnaire scenario. The catchment area would provide for example increased recreational and aesthetic values, habitats for fauna and flora and local pride. There was a response rate of ca. 31%.

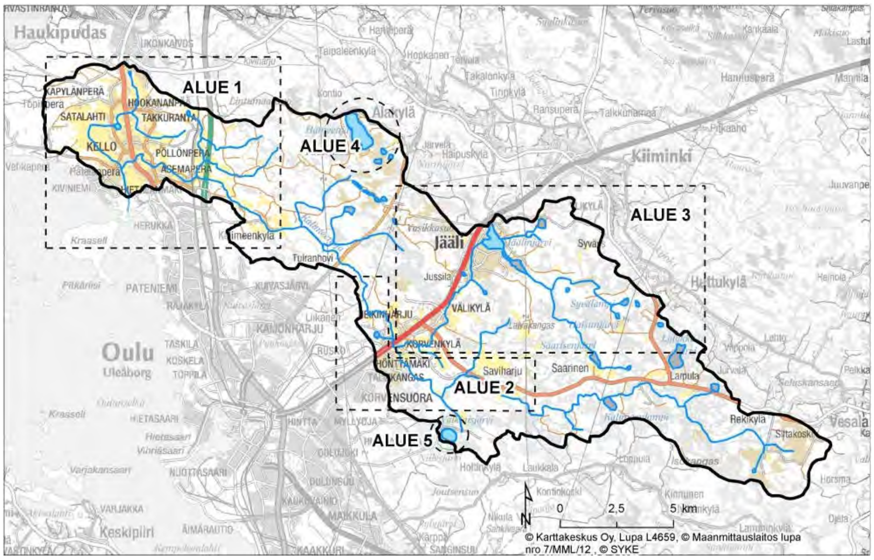
The respondents' unanimity about the major impact of the river basin on the residential environment's attractiveness demonstrated that both the river landsapce and water quality are important to residents of the Kalimenjoki river basin. For most of the respondents, it is extremely important that minor water bodies in the Oulu Region be nurtured and restored. Almost a tenth had participated in pracitcal restoration activities. However, despite regarding the river basin and its restoration important, respondents felt that responsibility for improving the river basin's status was held by those contaminating it and society. Only ten per cent were uninterested in the river basin.

Most respondents were unwilling to pay "a water management donation", in most cases because they believed that the polluter and society should pay. A total of 150 named the maximum management donation that they would be willing to pay. This fee ranged from euro 18.70–25.70 per household. Those asked about the hypothetical payment option were willing to pay around 1.9–2.5 times more than recipients of the actual option to pay the association. At its most powerful, WTP was based on the wish to use the river basin for leisure activities. A third of the respondents willing to pay wanted to keep the river catchment area in good status for future generations.

The survey presented a new method of describing national water management goals and making them tangible for the general public. Its results provide valuable information in support of regional activities and decision-making, information on the survey was efficiently described by the local media. At the same time, the survey led to the widespread distribution of information on water management and river basin restoration based on almost half of all the households in the catchment basin. It also prompted a discussion of the benefits and costs of water management in the operating area. Surveys, based on the Kalmenjoki model could also be used in other areas, where there is a need to ascertain the views of catchment area residents on water management.

The aim of this study was to estimate the benefits of improved water status, and to compare hypothetical and real willingness-to-pay. It did not compare benefits to costs.

Figure 3.13: The River Kalimenjoki catchment area (number 84.114)



Source: Lehtoranta *et al*. 2013.

Hovinbekken and Alna local rivers in urban Oslo, Norway – ecosystem services quantification and valuation by benefit transfer – used to assess disproportional costs

In this project the aim was to assess and compare benefits, costs and disproportionate costs in water bodies in two urban rivers in Oslo. We will come back to the assessment of disproportionate costs in chapter 4, and have described the potential ecosystem services which are of interest in these rivers in section 3.3. Here, we will describe how the ecosystem services which could be affected by measures to improve water status in the two rivers were quantified and valued in what the authors call a screening process for assessment of benefits and – potential disproportionate costs. The two rivers, and the water bodies they consist of, are illustrated in figure 3.14.

Figure 3.14: Case study rivers in urban Oslo, Norway. River Hovinbekken on the left, divided into two waterbodies (Hovinbekken 1 and Hovinbekken 2). River Alna on the right divided into five+two water bodies (Alna 1-Alna 5 plus two water bodies consisting of side streams side streams of Alna and stream Tokerudbekken)

Source: Magnussen *et al*. 2014.

The authors follow the same procedure for assessing the value of improved water environment as we described in chapter 3.1 in this report:

- Identify relevant benefits resulting from carrying out relevant measures, based on the description of ES.
- Quantify the identified, affected ecosystem services.
- Yalue the identified benefits in money terms.

By using the table, reproduced in section 3.1 (table 3.2) in our report, as a point of departure, the relevant measures are discussed with respect to which ecological and other effects they will have, and which ecosystem services will be affected. Following this procedure, it is concluded that cultural ecosystem services will be most affected (improved) and some regulating ecosystem services may be slightly improved, but not much in the present period (until 2021). The most important ecosystem services affected are shown in box 3.6.

Box 3.6: Important ecosystem services affected by relevant measures in Hovinbekken and Alna rivers

Identification of the most important ecosystem services affected by relevant measures:

- Cultural ecosystem services:
  - Recreational ecosystem services.
  - Aesthetic ecosystem services.
  - Knowledge and learning.
- Regulating ecosystem services:
  - To some degree: reduced flow and erosion, however modest effects of suggested measures in the period considered (until 2021).
- Supporting ecosystem services:
  - Preservation and improvement of the ecosystem, including values related to biodiversity: non-use-values.

Source: Magnussen *et al*. 2014.

Quantification and valuation of effects of water improvements

- *Quantification of improved recreation*
  As a proxy for quantification of increased recreation services, the number of people who can potentially enjoy the improvements is used. Since the rivers in the case study are small and local, and there are 11 rivers running through Oslo, it is assumed that these rivers have local use and non-use-values only. Therefore, the number of people living at a certain distance from the rivers (water bodies) is used in order to quantify the ecosystem services improvement.
- *Valuation of benefits*
  Since the measures in PoMs first and foremost will affect cultural ecosystem services, the main emphasis has been on valuation of these services. There was no existing primary valuation study for the two rivers in question, and very few valuation studies of improved water environment in general in Norway, and even fewer with relevance to urban rivers. However, there was one, quite recent pilot study using the Contingent Valuation Method to value improved water status in another river in Oslo, called river Akerselva. Although this is a larger and more often sought river for recreation, the estimates from this was used as a screening benefit transfer procedure in order to give a rough estimate of the values of improved water environment in all the water bodies in the two rivers. The results from Akerselva was that the WTP per household was 137

(2012) Norwegian kroner (NOK) per year in a ten-year period in order to secure good swimming water quality.

Using the same estimates for the number of households living within different distances from the water bodies, and the WTP estimate referred to above, a rough estimate for the value of increased recreation and non-use-values from the improvements are presented in the table below. There is a thorough discussion of these estimates in the Magnussen *et al*. (2014) report, whether they should be seen as maximum values etc. However, for our purpose, we do not repeat this discussion, but present the results, which can be used for comparison with estimated costs for relevant measures in the same water bodies. It should be pointed out though, that the study and the discussion, underlines that the uncertainty by using benefit transfer instead of primary studies, increases the uncertainty in CBA and assessment of disproportionate costs.

Table 3.4: Estimated Economic Present Value (PV) of total Willingness to pay (WTP) for the population living at a certain distance from different water bodies (less than 100; 300 or 1,000 meters), under the assumption that WTP per household is NOK 140 per year in a 10-year period. Numbers are in million NOK

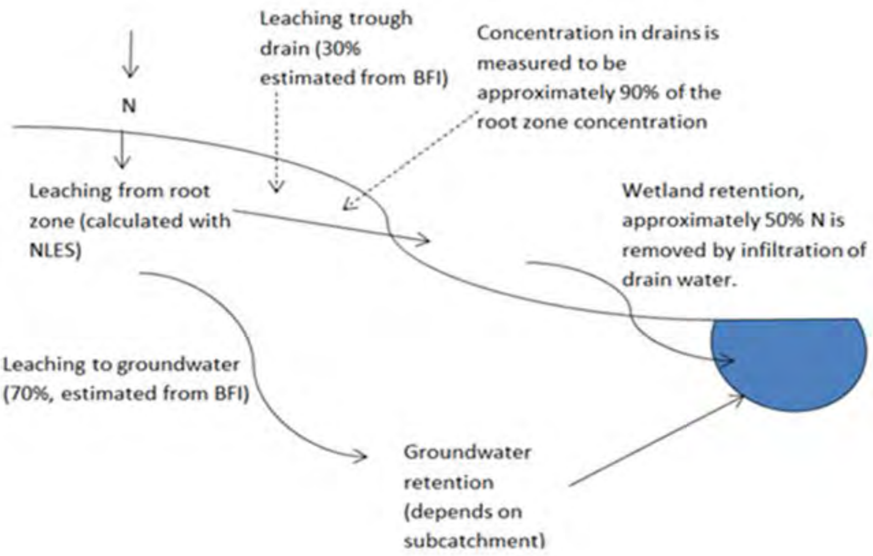
	Distance from water body		
	1,000 meter	300 meter	100 meter
Water body	PV in mill. NOK	PV in mill. NOK	PV in mill. NOK
Alna 1	7.2	0.7	0.1
Alna 2	26.0	9.3	2.3
Alna 3	11.5	0.3	0.1
Alna 4	22.2	4.1	0.7
Alna 5	23.8	4.6	0.7
Side streams to Alna	25.6	7.8	2.4
Tokerudbekken	30.3	13.2	4.6
Total Alna	*Ca. 147*	*Ca. 40*	*Ca. 11*
Hovinbekken 1	13.2	6.2	1.8
Hovinbekken 2	38.5	10.6	2.9
Total Hovinbekken	*Ca.52*	*Ca. 17*	*Ca.5*

Source: Magnussen *et al*. (2014).

Danish example on quantification and valuation of the regulating ecosystem service, retention

Retention is a regulating service, because nutrients and other pollutants are transformed or kept in the soil, and prevented from loading into the water bodies. The illustration in figure 3.15 is from a Danish yet unpublished study (Termansen *et al*. 2014); where root-zone leaching and leaching through drain is retained in soil, in groundwater and in wetlands, before the final loading enter the water body shown in blue. The retention in a Danish catchment can be up to 80–90% of the initial loading from the root zone, and down to between 0 and 10%. The high variation in retention calls for targeted regulation utilising the retention as an ecosystem services. The value is high as its replacement cost is high: if we did not have this retention farmers had to reduce non-point pollution for their fields by costly actions, and some places lost retention capacity would result in double efforts from farmers.

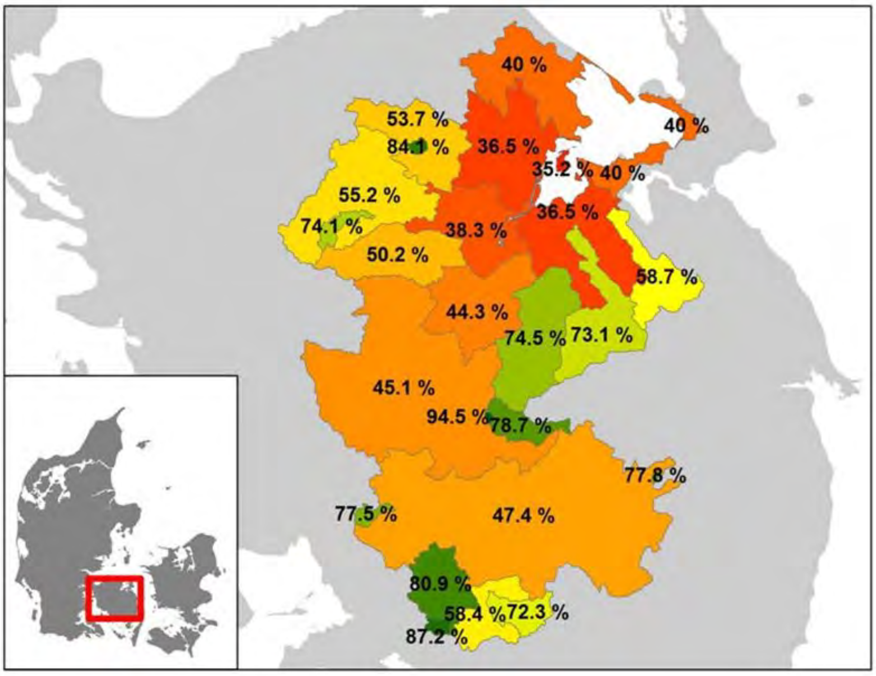
Figure 3.15: Regulating services from fresh water



Source: Maes *et al*. 2013.

In a recent study in Denmark the value of increasing the retention in the Odense catchment is valued. The modelling of the retention is based on 1) a retention map for the sub-catchments within the Odense fjord catchments. As can be seen from map 1 in the figure 3.16 below, the retention (the retainment of nitrogen in soil, groundwater and surface water) is relatively high in the catchment, but it varies a lot; between 35 and 95% of the initial nitrogen load is retained in the sub-catchments, and do not reach the fjord. The retention map has been linked to a map of the agricultural production in the area and an agricultural model used for modelling changes in land use from changes in regulation – nitrogen tax, requirement for wetland restoration etc. Subsequently the benefits of these nutrient reductions are mapped using the Odense valuation study carried out in Aquamoney (see section 3.2.2). These results are not yet published but are forthcoming.

Figure 3.16: Mapping nitrogen reductions from the root zone



Source: Termansen *et al*. 2014.

3.5 Main findings in this chapter

This chapter has demonstrated that there are some interesting examples on the use of the ecosystem services framework in order to identify, quantify and value the benefits from freshwater in general, and the improvement of freshwater status according to WFD in particular, across the Nordic countries. The presentation of the examples also seek to demonstrate that there are a number of studies that do not use the ecosystem services approach per se, but the studies provide information about the value of ecosystem services.

However, when we move from the ambition of illustrating and demonstrating the different ecosystem services from freshwater and improvements in freshwater status, the examples show that it is demanding to identify, and particularly quantify and value in monetary terms the benefits of reaching good ecological status (GES).

Identification of ecosystem services can be done, and is done, on different scales (water body, river basin, country, region) depending on the purpose. In some studies the identification and valuation has been done with focus on one or a few ecosystem services. In a WFD context the most interesting question is how the benefits from all ecosystem services are changed (increased) by reaching the goal of good ecological status. There are only few primary valuation studies that follow the ecosystem services terminology to full extent. Even rather new valuation studies aiming at valuation of the benefits of reaching the WFD goals of reaching GES (e.g. the Aquamoney study in Denmark and Norway; and two recent valuation studies in Finland) do not use the ecosystem services terminology per se, however the benefits valued are easily recognisable in ecosystem services terminology. Since the number of new primary valuation studies has been limited in the Nordic countries, we need to use the information from valuation studies where the ecosystem services terminology has not been used. This was the approach taken in the study from local rivers in Oslo, where ecosystem services affected by water status improvements were evaluated in an ecosystem services framework, and prior valuation results were used to give a rough estimate of the value of the changes in these identified ecosystem services. What one would wish for are new primary valuation studies for fresh water that use the ecosystem services terminology from the beginning, and follow the identified steps.

Most of the studies and reports so far do not, or only to a minor degree, take into account the need for trade-offs, and other issues we discussed in 3.1 and 3.2. There is an on-going discussion of these issues in the academic literature. The ecosystem services framework is still new in applied work, however, and much emphasis so far has been on how the framework can be used, which ecosystem services are involved, how they could be described etc. Probably, the issues of concern will be taken more into account as the framework is more commonly applied.

The ES framework can be a tool for systematic identification of benefits and to investigate the connection between ecological changes and welfare gains, and this chapter shows that the framework is coming into use across the Nordic countries. However, this framework is no "quick fix". Much work is still needed on all aspects of identifying, quantifying, and not at least valuing ecosystem services – both with respect to the ecological underpinnings and the economic methodology.

4. Assessment of disproportionate costs

In this chapter we:

- Start with a brief reminder about disproportionate costs in WFD (section 4.1).
- Give some examples on how assessment of disproportionate costs can be done. We start with a couple of examples from outside the Nordic region, and then show how this assessment is done in two Nordic examples (section 4.2).
- Discuss and conclude regarding findings and what can be learnt from this chapter (section 4.3).

4.1 Disproportionate costs in WFD

According to the WFD costs can be disproportionately high compared to financial ability to meet the targets, or compared to the benefits of meeting the targets. Financial ability cannot be a justification for reducing the targets of good ecological status (GES), and will not be treated further here. However, if the costs are disproportionately high compared to the benefits of reaching the water status targets, this may justify less ambitious targets or justify postponement of reaching the targets.

In the latter case, we need to compare the benefits with the costs. We often assume that the costs are easier to estimate than the benefits. Still, a review analysis carried out would most likely reveal that there are only a few examples of estimates of costs to society of reaching the targets of WFD.

This is noted in Martin-Ortega (2012) in her review of economic analysis applied in WFD, that while cost effective analysis (CEA) has been widely adopted by national guidelines, and the estimation of the benefits has received significant attention in literature, the way these two should be joined up in a CBA has received much less attention. This is in accordance with our own experiences from the Nordic countries. Although we would add that the interest in benefit estimation has also been very limited in these countries, and the attempts made have been mainly on a research basis, and to a very little extent included in practical implementation of WFD. The extent to which CEA has been carried out in practice is also varied, though all countries have some guidelines telling that this is supposed to be part of the work.

The guidelines for WFD suggest that the most cost-effective measures should be carried out in order to reach the goals, and there are calculations of program of measures around. In many countries and PoMs the financial costs are calculated only, not the economic costs for society.

Still, we believe it is easier to calculate the costs to society than to calculate the total benefits of environmental improvements. Therefore, it seems reasonable to start with calculating the real economic costs of measures, and to use these to compare with the benefits.

If we get the costs reasonably correct, we have a "benchmark" towards which we can compare the benefits.

Getting from financial costs to economic costs to society may take some necessary steps, which are explained on a country basis in Jensen *et al*. (2012) and on a water body level in Magnussen *et al*. (2014).

Like assessment of benefits, assessment of disproportionate costs may be carried out at different scales /levels – on a country scale, as a screening procedure to identify river basins where the costs of fulfilling the WFD may be disproportionate, on a river basin level and at the water body level. A focus of our report is to assess whether the costs are disproportionate which could lead to exemptions (time delay or permanent exemption), and this procedure will need to be carried out on the water body scale, as exemptions are to be made on the water body level.

4.2 Examples of assessment of disproportional costs according to WFD

4.2.1 Introducing the examples

There are not many examples of using CBA to assessment of disproportional costs in the Nordic countries, or in EU as a whole. Even if the benefits of improved water status are estimated, it is not necessarily the ES framework which is used. Therefore, the number of case studies or applied work to choose among in order to present examples in this chapter is somewhat restricted. Still, there are a few examples, as we will show, and this is probably an area where more work will be carried out in the coming years.

Our first example is from Scotland (section 4.2.2.1) where the authorities have suggested to use cost estimates only in order to assess disproportionate costs, but where researchers recently have suggested how the known information about costs in different lochs in Scotland can be combined with results from benefit estimates for reaching GES in the same lochs, in order to include more thorough cost benefit assessment for reaching GES in all the lochs in Scotland.

This framework is interesting because it combines information about costs and benefits on a national (Scotland) scale and uses existing data on benefits and costs in order to reach more rigorous results about costs and benefits, and in which lochs the costs seem to be disproportionate. This framework does not include the ES framework.

Our next example from region Emilia-Romagna in Italy (section 4.2.2.2) does not include the ES framework either. Still, it is briefly mentioned because it develops a methodology for the assessment of disproportionate costs, using estimates of benefits and costs, and the study identifies areas where disproportionate costs are more likely to occur.

We then move to Nordic examples. The first is a study where the Aquamoney study in Odense (described in section 3.4.) is transferred to 22 Danish river basins and used for å screening of the costs and benefits of achieving GES. As mentioned in section 3.4; this primary study from Odense did not use the ecosystem services terminology per se, but the goods and services valued would be characterised as ecosystem services. This is an interesting and seldom Nordic example in which there is a good quality primary study of one river basin which has been transferred to the other rivers in the country, and where the researchers also have put in emphasis in order to calculate the costs to society (which are the costs we aim for in CBA). It is interesting that Swedish authorities chose to make a benefits transfer from Danish and Norwegian valuation studies when they wanted to value the benefits of reaching GES in Swedish rivers. In Norway, the case study from Aquamoney (Morsa) has not been used in order to estimate benefits for the other rivers in the country, and no estimates of the benefits of reaching GES, or comparison of benefits and costs on a national or river basin scale exist, as far as we have knowledge of. In Finland several valuation studies for freshwater benefits of reaching GES are carried out, and also CBAs of reaching GES, as we saw in chapter 3 (section 3.4.2).

As our last example in this chapter we have included one recent study for two local rivers in urban Oslo, which has been introduced in chapter 3 (sections 3.3 and 3.4), and where also an assessment of disproportionate costs was carried out. This was also done as a screening process in order to see in which water bodies more detailed benefits and cost assessments need to be carried out in order to decide whether costs are disproportional so that exemptions from the general goal of GES must be given.

4.2.2 Examples

National level analysis – An example from Scotland – disproportionate costs, but not ES framework

Martin-Ortega (2012) mentions the framework of the Scottish Environment Protection Agency (SEPA), which has been largely to rely on CEA alone (SEPA 2005). This decision was based on the outputs of the Impact Assessment of the River Basin Management Plan (RBMP) (Scottish Government, 2008), which includes a qualitative assessment of benefits and led the regulator to assume that mitigation is usually proportional unless costs seem particularly high or if concern is raised. This implies, in principle, that there has been an assessment of benefits, but not all environmental benefits are estimated quantitatively.

In order to include cost benefit assessment more thoroughly within the disproportionate cost assessment in Scotland, Vinten *et al*. (2012) proposed a framework for proportionality assessment of phosphorus (P) pollution mitigation in 544 Scottish lochs at national and local water body scales. For 293 (31%) of the lochs GES already occurred. Mitigation cost-effectiveness was assessed using combined mitigation cost curves for managed grassland, rough grazing, arable land, sewage, and septic tank sources. These provided sufficient mitigation for GES to be achieved in another 31% of lochs areas at annualized cost of GBP 2.09 million per year. Mitigation of the residual P loading preventing other lochs achieving GES was considered by using a "mop-up" cost of GBP 200 per kg P (assumed cost-effectiveness of removal of P directly from lochs), leading to a total cost of GBP 189 million per year. Lochs were ranked by mitigation costs per loch area to give a national scale marginal mitigation cost curve.

A published Choice Experiment valuation of WFD targets for Scottish lochs (Glenk *et al*. 2011) was used to estimate marginal benefits at national scale and combined with the marginal cost curve. It can be assumed that at a national level, benefits will decline as the area (number) of lochs restored to GES increases due to limited national WTP and scare resources for government support. The national scale benefit estimates were derived from Glenk *et al*. (2011). Vinten *et al*. note that these national scale benefit estimates were quite well suited to guide decisions about national improvement targets if ecological standards adopted by the regulator constitute reliable local estimates of the balance between societal benefits of clean water and the social costs of achieving it. At a local scale, it is pertinent to characterise benefits and costs, independent of management of other lochs in the country. However, in the absence of suitable benefit transfer studies, benefits of mitigation for individual lochs were estimated by reference to a national average mean WTP for achieving GES, derived from Glenk *et al*. (2011).

This gave proportionate costs of GBP 5.7 million per year leading to GES in 72% of loch area. Using national mean marginal benefits with a scheme to estimate changes in individual loch value with P loading gave proportionate costs of GBP 25.6 million per year, leading to GES in 77% of loch area (491 lochs). That is, according to these results, 72% of the lochs that could be mitigated proportionately at a cost of GBP 5.7 million per year. Mitigation beyond this point would be disproportionate.

Regional level analysis – An example from Italy – disproportionate costs, but not ES framework

Galioto *et al*. (2013) develop a methodology for the assessment of disproportionate costs according to the WFD guidelines for the region Emilia-Romagna in Italy. According to the authors the originality of the framework lies in the focus on the interdependencies between water bodies and the consideration of the multiple interactions between measures and pressures. However the broad architecture of the study fits into a wider assessment procedure already developed in recent studies. Specifically, a cost-effectiveness analysis, implemented to select an efficient combination of measures, is integrated with a cost benefit analysis, which allows for the evaluation of the economic feasibility of the proposed actions. This methodology is applied to the Emilia-Romagna Region in Italy. In spite of the uncertainties in the estimations of costs and benefits, the study enables the identification of areas where disproportionate costs are more likely to occur. The results show that disproportionality tends to increase from foothill regions, where most of the functional uses of regional water resources are found, to plain areas, where the sources of pressure tend to be located.

Danish example – Screening procedure for 23 rivers – disproportionate costs, but not ecosystem services per se

Jensen *et al*. (2013) use the benefit results from the Aquamoney study in Odense (see chapter 3.4.2 and Barton *et al*. (2012), as well as Hasler *et al*. 2010), and transfer the results to the other 22 Danish river basins for a screening of the costs and benefits of achieving good ecological status. The study demonstrates a methodology designed to investigate disproportionate costs in the 23 river basins. The CBA which is performed as a basis of the screening applies a conservative framework where the lowest levels of the benefit results are compared to the highest levels of costs, to ensure that all river basins where the benefit cost ratio is positive is surely so, while those with a negative benefit cost ratio will be further investigated as a basis for whether the costs are disproportionate.

The study is done for all 23 river basins. The water status of these basins is described in table 4.1 below. As can be seen the water status of the streams is generally good, while the status of the lakes, fjords and coastal areas is much worse.

Table 4.1: The current average ecological status in the 23 river basins

River basins	Streams	Lakes	Fjords	Coastal waters
Kattegat and Skagerrak	Moderate	Good	No Fjord	Poor
Limfjorden	Moderate	Poor	Poor	Poor
Mariager fjord	Good	Poor	Poor	No coastal water[c]
Nissum fjord	Good	Good	Poor	Poor
Randers fjord	Good	Moderate	Poor	No coastal water[c]
Djursland	Good	Moderate	No Fjord	Poor
The Bay of Aarhus	Moderate	Poor	Poor	Poor
Ringkøbing fjord	Good	Good	Poor	Poor
Horsens fjord	Good	Poor	Poor	No coastal water[c]
Wadden sea	Moderate	Moderate	No Fjord	Poor
The Belt sea (Lillebelt, Jutland)	Good	Moderate	Poor	Poor
The Belt Sea (Lillebelt, Funen)	Moderate	Poor	Poor	Poor
Odense fjord	Moderate	Poor	Poor	Poor
The Belt sea (Storebelt, Funen)	Moderate	Poor	Poor	Poor
The Sea south of Funen	Moderate	Moderate	Moderate[a]	Moderate
Kalundborg	Moderate	Poor	Poor	Poor
Isefjord and Roskilde fjord	Moderate	Poor	Moderate	No coastal water[c]
Oresund	Moderate	Good	No Fjord	Poor
The Bay of Koge	Moderate	Moderate	No Fjord	Moderate
The sea south-west of Zealand	Moderate	Moderate	Poor[a]	Moderate
The Baltic Sea (Baltic Proper)	Moderate	Moderate	Moderate	Moderate
Bornholm	Good	Good	No Fjord	Moderate [b]
Kruså/Vidå	Good	Moderate	No Fjord	Poor

[a] The status is not classified in the final RBMP. Instead we use the status from the hearing version of the RBMP. The classification in Denmark will in the second planning phase be based on the "one-out-all out" principle, but this is not the case in the above classification.

[b] The status of water quality of coastal waters is not classified in either the final or the hearing version of the RBMP. Instead we have used the status of the Baltic Sea basin to indicate the water quality of coastal waters.

[c] In these basins the water exchange between the fjords and the surrounding coastal waters is particularly weak and therefore coastal waters has not been included in these basins. While water does flow from fjord to sea it is assessed that sea quality impacts will mainly occur further from the coastline than what is covered by the WFD. Furthermore, these basins have very limited direct coastline were benefits can be found.

The table 4.2 shows the benefit cost ratio under different assumptions.

Table 4.2: Annual welfare gains and benefit cost ratios (B/C) under baseline and sensitivity analysis scenarios

River basin	Baseline Baseline benefit- Baseline cost		Scenario 1 Alternate benefit Baseline cost		Scenario 2 Baseline benefit Alternate cost		Scenario 3 Alternate benefit Alternate cost	
	€ (Mill)	B/C	€ (Mill)	B/C	€ (Mill)	B/C	€ (Mill)	B/C
Category 1: Costs are higher than benefits								
Bornholm	-1.340	0.0	-0.833	0.4	-1.308	0.0	-0.801	0.4
Kruså/Vidå	-6.723	0.1	-1.962	0.7	-4.576	0.1	0.186	1.0
Djursland	-3.361	0.1	-0.768	0.8	-2.147	0.1	0.446	1.2
Category 2: Cost and benefits are at the same level								
The Belt sea (Lillebelt, Jutland)	-6.819	0.7	6.487	1.3	-5.885	0.7	7.421	1.4
Kattegat and Skagerrak	-1.229	0.8	5.476	1.9	-0.514	0.9	6.190	2.2
Limfjorden	-12.210	0.8	10.981	1.2	1.783	1.0	24.975	1.5
Nissum fjord	-1.427	0.8	3.608	1.6	0.506	1.1	5.541	2.2
Randers fjord	-5.175	0.8	-5.175	0.8	-3.153	0.8	-3.153	0.8
Ringkøbing fjord	-0.934	0.9	4.931	1.7	-0.966	0.9	4.900	1.7
Wadden sea	-0.314	1.0	10.963	2.1	-0.150	1.0	11.127	2.1
Mariager fjord	0.336	1.1	0.336	1.1	0.666	1.3	0.666	1.3
The Sea south of Funen	-0.239	1.0	0.993	1.2	-0.189	1.0	1.043	1.2
Category 3: Benefits are higher than costs								
The sea south-west of Zealand	9.342	1.5	13.072	1.7	10.737	1.7	14.466	1.9
The Baltic sea (Baltic Proper)	1.879	1.9	2.798	2.3	1.874	1.9	2.793	2.3
Horsens fjord	4.674	2.0	4.674	2.0	4.979	2.1	4.979	2.1
The Belt Sea (Lillebelt, Funen)	7.358	4.0	11.385	5.6	7.489	4.2	11.515	5.9
Odense fjord	20.453	4.0	31.636	5.6	21.377	4.6	32.559	6.5
The Belt sea (Storebelt, Funen)	3.553	4.1	5.476	5.8	3.558	4.1	5.482	5.8
The Bay of Koge	17.361	4.2	25.664	5.7	17.380	4.2	25.684	5.7
Kalundborg	6.135	4.4	9.391	6.2	6.453	5.3	9.708	7.5
Isefjordand Roskilde fjord	25.776	4.5	25.776	4.5	25.936	4.6	25.936	4.6
Oresund	29.390	10.0	67.581	21.8	31.103	21.1	69.581	45.9
The Bay of Aarhus	42.201	15.8	60.661	22.3	43.053	22.6	61.513	31.8
National	128.687	1.6	293.438	2.3	158.006	1.8	322.756	2.7

After Jensen *et al*. 2013.

The so-called base line assumption use the benefit estimates from the Odense valuation results, and compare to the costs of achieving GES, calculated by Jacobsen (2012). In this baseline it is assumed that the benefits in coastal areas are zero (because they were not assessed in the Odense study). In the scenario 1 a sensitivity analysis is performed assuming that the benefits of improving coastal areas are equal to the benefit estimated for the fjord. Naturally fewer river basins are at risk of negative benefit cost ratios when the coastal areas are included. The scenario 2 assumes the benefits used for the baseline, while the costs are lower because the localisation of the measures is optimised. In the last scenario 3 both alternatives are used, i.e. lowest costs and highest benefits.

The authors (Jensen *et al*. 2013) conclude that the results from the study indicate that this procedure may serve as a first step towards assessing disproportionate costs. By using the river basins as the geographical scale for the analysis existing data is utilised for describing the average water quality status and whether GES is obtained. The costs and benefits of achieving GES relative to the current situation are assessed, also using existing knowledge on both costs and benefits. For three basins the study conclude that there is a high likelihood of obtaining disproportionate costs, and the recommendation is that more detailed analyses in these areas should be performed, as well as in nine areas where costs and benefits are around zero. This recommendation does not mean that costs are disproportionate in these areas but that further analyses should be carried out to investigate the benefits. In addition one could add that further analyses of more cost-effective measures and instruments could also improve the solutions, cf. chapter 5. In this respect both the ecosystem services framework and a more targeted use of instruments and measures has a strong potential in order to obtain solutions where benefits outweigh costs.

Norwegian example – Hovinbekken and Alna rivers in Oslo – screening of disproportionate costs, using the ES framework and benefit transfer

In a study for the regional environmental agency in Oslo and Akershus, Norway, Magnussen *et al*. (2014) show how benefits and costs that arise from reaching the environmental goals of WFD can be compared. The study aims at methodological development and use of the methodology in a case study in two rivers in urban Oslo: rivers Hovinbekken and Alna. The project used the ecosystem services framework in order to assess and value the benefits of improved water quality, as described in chapter 3. Further, the project developed a “screening procedure” for assessment of disproportionate costs. The project suggests a step-wise framework using economic cost benefit analysis as the methodological framework and point of departure. The project emphasized the need for simplified and not too demanding and cost-driving framework for screening of benefits and costs, because this procedure should be possible to use in rivers around the country on the water body level.

Suggested step-wise framework for assessment of benefits and costs of improved water environment in urban rivers

- *Main steps in the analysis*
  The suggested steps are shown in figure 4.1. The first step is to describe the rivers and water bodies to be analysed, their present status, whether the water bodies are "natural" or heavily modified (HMWB). Thereafter, relevant measures and their effects and mitigation costs are estimated. The benefits of the measures (reaching GES) were identified, quantified and valued in monetary terms using the ecosystem services framework (See chapter 3) and finally, benefits and costs were compared and an assessment of whether costs were disproportionate were discussed, and whether GES could be reached in the first period (until 2021).
  The case study revealed that there were very few examples where economic analysis has been attempted used in order to estimate benefits and costs in Norwegian water regions. Therefore, it was emphasised the need to develop and document rather detailed – how costs as well as benefits can be assessed and calculated. Limited access to data, and restrictions on time and money, made it necessary with adjustments and simplifications from an ideal cost benefit analysis. Figure 4.1. summarizes the steps taken in this assessment of benefits and (disproportionate) costs.

Figure 4.1: The steps in the analysis

1.Determine current status, normal or heavily modified water body and the need for improvements in each water body		
1a) Current status in each water body with respect to: - Hydro-morphological quality elements - Biological quality elements (like fish) - Physical-chemical parameters (eutrophication and hazardous substances)	1b) Determine which kind of water body: - Natural water bodies, in which Good Ecological Status (GET) is the goal - Heavily modified water bodies in which Good Ecological Potential (GEP) is the goal	1c) Assess the need for improvements Assess the need for improvements considering the distance between goal and current status

2.Identify measures, effects and costs to society in each water body (WB)		
2a) Identify relevant measures and groups of measures in each WB	2b) Assess, and if possible, quantify, the effects of different groups of measures	2c) Calculate economic costs to society of different measures

3. Identify, assess and calculate benefits of improved water status		
3a) Identify benefits using the Ecosystem Services Framework	3b) Quantify benefits as far as possible	3c)Value the benefits as far as possible

4. Compare costs and benefits, and make an assessment of whether the costs must be considered to be disproportionate (screening), and possibilities to reach environmental goals in year 2021		
4a) Assess the decision rule for screening of whether the costs must be considered to be disproportionate.	4b) For natural water bodies: i) Assess benefits and (disproportionate) costs ii) Assess the possibilities for reaching environmental goals in 2021, and eventually the need for postponement or exemption	4c) For heavily modified water bodies: i) Assess which measures should be included in GEP. ii) Assess the possibilities for reaching GEP in 2021 and eventually the need for postponement or exemption

Source: Magnussen *et al*. 2014.

Regarding the fourth step, Magnussen *et al*. (2014) write: Comparing costs and benefits, assess whether costs are disproportionate, and the possibilities for reaching environmental goals until 2021.

In this step benefits and costs were compared and an assessment of disproportionate costs was made, in order to consider the need for exemptions from the general goals of environmental status (from GES in natural water bodies and from Good Ecological Potential – GEP – in Heavily Modified Water Bodies, HMWB). For HMWB an assessment of which measures should be included to reach GEP was discussed.

In the case-study-rivers, it was found that there was substantial uncertainty in calculations, benefit estimations in particular, because it was only to a minor extent certainty about which exact effects the mitigation measures will give, and because they could not carry out primary valuation studies in order to estimate the benefits, they had to rely on benefit transfer (see chapter 3) from one pilot study in a nearby river. Cost calculations are often considered less uncertain. However, there are made several simplifications in the cost calculations in the case studies. The authors argue that it is a good way of proceeding in assessment of disproportionate costs, to estimate the economic costs of mitigation with as much certainty as appropriate, and use these cost estimates as a "bench mark" towards which the benefit estimates can be compared, and then assess the identified ecosystem services, and quantified and valued benefits seem to be larger, less or approximately the same as the costs.

Magnussen *et al*. (2014) see this process with assessment of benefits and costs as a screening in order to find the approximate size of benefits and costs of relevant mitigation measures and as a basis to assess benefits and costs in more detail later. If the benefits are unquestionable much larger than costs of reaching the environmental goals, they suggest there is no need to go further in discussing exemptions. Mitigation measures should be carried out in order to reach the environmental goals of WFD. If the costs unquestionably are much larger than the benefits, there is good reason to consider exemptions. If the costs and benefits are of approximately equal size, there is reason to make more thorough assessments and calculations. The study discussed benefits and costs in the first period of the WFD, till 2021, and because of considerable uncertainties about relevant measures and their effects, the authors suggest that a reasonable decision rule is to let the environment have the benefits of doubt. That is; if the costs and benefits are nearly equal according to the analysis, this should trigger more thorough analyses in the period to the next six-year-planning period according to WFD, in order to reduce uncertainty. If the calculated costs obviously are larger than benefits, one should consider making exemptions in time and consider that the goals should not be reached in the first period. And at the same time they suggest that the authorities should carry out more detailed investigations and calculations towards next planning period in WFD. This was the framework suggested and used in the case studies. The same decision rule was used as the basis for considering measures to include in Good Ecological Potential.

The study does not compare benefits and costs for individual mitigation measures, but for the total of mitigation measures, because it is deemed too time consuming and demanding to estimate benefits from each measure. The Norwegian guidelines do not recommend estimating benefits for each measure but for the total of measures. In follow-up analyses which are more detailed, however, the authors suggest that it may be needed to assess benefits and costs for individual measures. This may be of particular relevance for particularly costly measures, and for discussion exactly which measures should be included in setting GEP in HMWB.

4.3 Main findings in this chapter

There are relatively few examples of Cost Benefit Analysis (CBA) in the context of the Water Framework Directive (WFD), and even fewer where the ecosystem services framework is used for benefit assessment in such analyses. This is not only the case in the Nordic region, but holds true for all of Europe.

Martin-Ortega (2012) in her paper on economic prescriptions and policy applications in the implementation of the European Water Framework Directive concludes that "… while CEA [Cost Effectiveness Analysis – authors note] has been widely adopted by most national guidelines in Europe, and the estimation of the environmental benefits has received a significant attention from the literature, the way these two should be joined up in a CBA has received much less attention".

We could add that even if the benefits are estimated, it is not necessarily the ES framework which is used.

We refer to a couple of examples from Scotland and Italy in which it is suggested how CBA can be used to assess benefits and costs – and potentially disproportionate costs – of reaching the goals of good ecological status in WFD. These studies do not use the ecosystem services framework directly, but still represent interesting examples of economic analysis for water management.

We also have identified a few Nordic examples, in which the ecosystem services framework more or less directly has been used for assessment of disproportionate costs, mainly as screening procedures, on a national, regional and local (water body) level.

This is exemplified in Jensen *et al*. (2013) who use information on the ecosystem services included in the Aquamoney study, i.e. the economic valuation results of water quality and ecological improvements in Odense river basin, in a benefit transfer to other Danish water bodies. The benefit transfer results by river basins are subsequently used for a cost-benefit analysis for the WFD implementation in Denmark. The CBA is used as a conservative screening of where costs appear to be disproportionate, i.e. exceed the benefits provided by these ecosystem improvements. Much of the same procedure and framework is used on the local water body scale in two rivers in urban Oslo as a screening procedure to evaluate benefits and potentially disproportional costs (Magnussen *et al*. 2014).

The ES framework is seen as useful, because it helps make a systematic and comprehensive picture of all benefits (valued in monetary terms, quantified or just verbally described) which is necessary to assess the total benefit of the improvements in water status. Implementation of a more full use of the ecosystem services framework should be implemented by including more services into the assessment than what is often done. It will also be important to pay attention to the supporting ecosystem services which in many respects are the basic foundation for providing the regulating, provisioning and cultural. Many of these supporting services will probably be part of the so-called non-use values in economic terminology (see textbox 2.2.). This is an area where more work is needed and probably will be carried out in the coming years.

5. Perspectives for locally adapted instruments, including PES, for enhanced ecosystem services provision

In this chapter we:

- Provide a number of examples and lessons of locally adapted or targeted economic policy instruments that have an impact on meeting WFD objectives and targets and which are introduced in section 5.1.
- Give background from agri-environmental policies that play an important role in the achievement of the WFD targets (section 5.2).
- Discuss some examples and policy recommendations in the Nordic countries that suggest moving towards more locally adapted instruments in the agricultural sector for the benefit of the ecosystem services of the aquatic environment (section 5.3).
- Present some examples where farmers are paid as climate adapters for cities (section 5.4) and a Nordic payment schemes for ecosystem services from restored and/or managed wetlands (section 5.5) and voluntary and mandatory PES programmes from Europe and the US that relate to paying land owners for actions that lead to improving water quality and thereby ecosystem services in a catchment area (section 5.6).
- Discuss different types of water quality trading programmes that exist in practice, operating at catchment level (section 5.7).
- Discuss and conclude regarding findings and what can be learnt from this chapter (section 5.8).

5.1 Introducing the examples

Achieving the targets of the WFD depends to a large extent on limiting negative externalities from land use practices. Negative externalities include excess leakage of nutrients (phosphorous and nitrogen) to water bodies causing eutrophication and the unattended spreading of environmental toxins from pesticides and herbicides impacting the chemical status of surface and groundwater bodies. As mentioned in Chapter 2, non-point pollution is difficult to control in practice, in particular when using uniform instruments that ignore differences in soil retention capacities, farm typologies and farmer characteristics. This "wicked" problem, cf. chapter 2, calls for a mix of instruments and measures that are adapted to local conditions and the involvement of a mix of stakeholders. The following sections provide a number of practical examples, policy trends and research insights of how locally adapted policy instruments, including PES, have been or are intended implemented that all have a direct impact on WFD targets.

The first section provides an overview of the greening of the EU CAP and the attempts to integrate water policy objectives. The section shows how the role of the ecosystem services framework increase in the new CAP in the attempts to reduce negative environmental externalities from agriculture. This applies to Pillar I (direct payments to farmers) with the obligatory introduction of Ecological Focus Areas as well as for Pillar II (Rural Development Programme) where voluntary agri-environmental measures profit from a significant budget increase. This is relevant for EU Member States only. The second section provides two examples from Denmark and Norway of moving towards more locally adapted instruments primarily in the agricultural sector. This implies new policy instruments that work with local hydrological and biogeochemical conditions to improve aquatic ES. The third section describes the preliminary thoughts and investigations in Denmark to set up locally based contracts with land owners to deliver climate regulating services from their land in partnership with cities. This is an example of how local stakeholders work together to enhance flood regulating services on agricultural land, which also have positive side-effects on WFD targets. The fourth section lists PES schemes in Denmark, Finland and Sweden in relation to wetland construction, an effective use of the ecosystem services framework to reduce nutrient load to the aquatic environment. The fifth gives examples from outside Nordic countries on establishing comprehensive PES programmes at catchment level to improve water quality, making use of a wide range of local measures that combined lead to improved water quality. The local measures range from ecosystem based approaches to changes in land use management and improved sewage water treatments. The final section deals with water quality trading as a market-based compulsory incentive for dischargers to comply with capped emissions. The ecosystem services approach in water quality trading is central where mitigation measures involve improving functions of natural ecosystems.

5.2 Agri-environmental policies

Common agricultural and agri-environmental policies play an important role in the achievement of the WFD targets in EU Member States given the importance of diffuse pollution to water bodies from agricultural practices and water abstraction. In Europe, agriculture accounts for around 33% of total water use and is the main source of nutrient pollution in water (EEA, 2012). The Common Agricultural Policy (CAP) contains two instruments which can be used to integrate the EU's water policy objectives: cross-compliance[17] and the European Agricultural Fund for Rural Development (EAFRD), often referred to as rural development programmes. RBMP measures can in some cases be financed through the CAP.

The reform of the European Common Agricultural Policy (CAP) 2014–2020 that started in 2010 and was formally adopted in December 2013 aimed at creating a better targeted, more equitable and greener support framework with increased emphasis on rural development and enhanced safety net (DG Agriculture, 2013). Both Pillar I (direct payments to farmers) and Pillar II[18] (Rural Development Programme) are maintained but links are strengthened and the green dimensions of both pillars have clearly been stepped up:

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[17] Cross-compliance is a mechanism that ties direct payments and a number of rural development payments to compliance with a series of rules relating to the environment, food safety, animal and plant health and maintaining agricultural land in good agricultural and environmental condition (GAEC). Cross-compliance rules cover currently 18 statutory management requirements and 15 GAEC standards; non-compliance can lead to a reduction in CAP payment to the farmer.

[18] Pillar I is offered to 100% of agricultural land and is 100% EU-funded, whereas Pillar II is offered to a part of agricultural land and is 50% EU-co-funded. In terms of budget, of the total CAP budget of EUR 362.8 billion (2011 prices), slightly more than ¾ of the total CAP budget is allocated Pillar I and slightly less than ¼ Pillar II. In real terms the budget for Pillar I has been cut by 1.8% and Pillar II by 7.6% (2011 prices). (DG Agriculture, 2013).

In Pillar I, a new direct payments system for farmers replaces the former Single Payment Scheme. On top of the new Basic Payment Scheme, a mandatory share of 30% of national direct payment envelopes is earmarked a new policy instrument: the Direct Green Payment. This pays farmers for mandatory agricultural practices beneficial for the climate and the environment (See Box 5.1).

In Pillar II, at least 30% of the budget of each Rural Development programme must be reserved for voluntary measures that are beneficial for the environment and the climate, such as agri-environment-climate measures, organic farming and Natura 2000. This has been increased from 10% of total CAP expenditures in the past CAP.

The requirement to establish ecological focus areas under the new Green Direct Payment (See Box 5.1) contains elements, such as buffer strips, which could serve as Natural Water Retention Measures (NWRM), a type of Green Infrastructure with beneficial effects on water quality.

Box 5.1 : Green Direct Payment in the EU Common Agricultural Policy 2014–2020

The new Green Direct Payment consists of three obligatory agricultural practices, deemed beneficial for climate and the environment:

- *Maintenance of existing permanent grassland* – Member States are required to maintain the ratio of permanent grassland at national, regional or farm-level such that it does not decrease by more than 5% compared to a reference period to be determined in 2015.
- *Ecological focus areas* (EFAs) – aim at safeguarding and improving biodiversity on farms either directly through land lying fallow, terraces, buffer strips, afforested areas and agro-forestry areas or indirectly through reduced use of inputs on the farm, such as areas covered by catch crops and winter green cover. Farms above 15ha are required to safeguard ecological focus areas covering five percent. Farms with more than 75% grassland or forests are exempted. Member States and farmers have the possibility to implement the EFAs at a regional or collective level to obtain adjacent areas.
- *Crop diversification* - small farms (10-30ha) are required to have at least two different crops of which the main crop may not cover more than 75%. For farms larger than 30ha, at least three different crops are required where the main crop may not cover more than 75% and the two main crops not more than 95% of the arable land. Farms with more than 75% grassland are exempted.

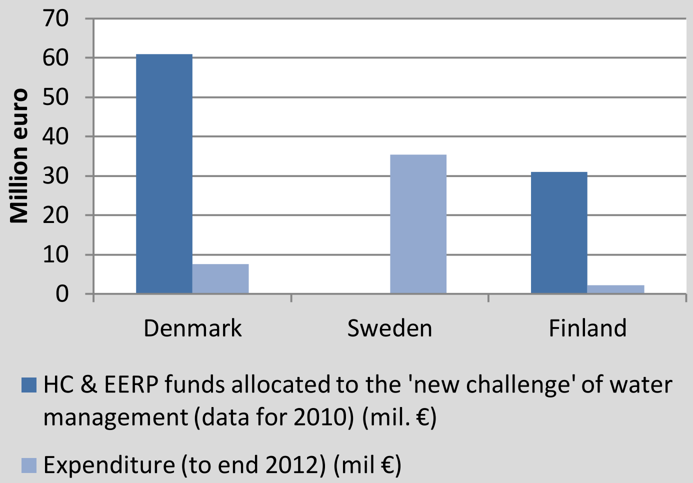
Source: Regulation (EU) No 1307/2013 – LOJ L 347/608 of 20.12.2013

Despite the greening of the CAP, the vast majority of subsidies under Pillar I (70%), the basic payment, still goes to intensive farming systems. Despite expectations, the new cross-compliance rules under CAP 2014–2020 do not include the basic measures from the RBMPs under the WFD (article 11.3) nor do they include compliance rules with the principles of sustainable use of pesticides and integrated pest management. Nevertheless, there was a declaration from Parliament and Council when the CAP was adopted that the Commission should come forward with a proposal for the inclusion of relevant parts of the WFD once the obligations for farmers were clarified.[19] The timing of this inclusion is dependent on the progress made by Member States in implementing the Directives, which implies that the implementation of a very important policy decision could be very slow (European Court of Auditors, 2014).

[19] Joint statement by the European Parliament and the Council on cross compliance attached to Regulation (EU) No 1306/2013.

Box 5.2: Lack of use of additional funding to tackle water management as one of the crucial new challenges for European agriculture

In 2009, the Council decided to strengthen the response to a number of 'crucial new challenges' identified in 2003. Water management was identified as one of the new crucial challenges for European agriculture (Council Decision 2009/61/EC). In line with this decision, the Health Check made available an additional budget through the EAFRD funds of a total of EUR 1,332 million, representing close to 27% of the total new budget for new challenges. The European Court of Auditors found that by the end of 2012, the additional financial instrument targeting water management has barely been used. Of the total EU funding, 17.5% was spent by the end of 2012. For Denmark and Finland the rate of imple mentation was lower at 12.5% and 7,3% respectively. There was no information available on total funds allocated in Sweden, although a significant 35.4 million euro was spent by the end of 2012.



Source: European Court of Auditors (2014).

At member state level, work is now on-going to define the detailed national rules of how to apply the new direct payment system, which comes into operationalization in January 2015 and to formulate the national Rural Development Policies.

Comparing EU and US agri-environmental policies (AEPs) directed at payments for environmental services produced by agriculture, Baylis *et al*. (2008) find pronounced differences in how instruments are targeted. While the EU member states through pillar II pays farmers for technologies or activities that are expected to reduce negative externalities of farming activities (e.g. nitrogen leakage), US farmers are paid by the Federal State to directly reduce negative externalities, regardless of the method(s) applied. The main conservation programme in the US, the Conservation Reserve Program (CRP), also takes opportunity costs of farmers into account by requiring competitive tendering for contracts that enhance ES. The CRP requires farmers wishing to apply for funding to submit bids based on environmental benefits to their land. Contracts are then allocated based on highest benefits for least cost. In comparison, the EU payments for providing ecosystem services under AEPs are typically based on national or regional set fees based on the individual member state's calculation of the income foregone and additional costs resulting from the commitment.

5.3 Moving towards more locally adapted instruments in Nordic countries

Some examples and policy recommendations exist in the Nordic countries to move towards more locally adapted instruments in the agricultural sector for the benefit of the aquatic environment.

5.3.1 Denmark

Current nutrient regulation in Danish agriculture is applied by setting general norms, buffer zones, general requirement of catch crops as well as a general requirement for animal manure utilisation. Each farm is given a per hectare quota of nitrogen, differentiated between crops. It is possible to choose different implementations of the quotas dependent on the yield level, crop distribution, catch crops etc. The current norms are general in the sense that they regulate the nitrogen input everywhere regardless of the resultant loads of nutrients to the water bodies. Since 1998, fertilizer norms are at 10% below economic optimal level of land use (Ministry of the Environment of Denmark, 1998).

A Nature and Agriculture Commission was established in Denmark in 2012 as an independent committee to analyse the current economic and environmental status of agriculture and provide recommendations of how to obtain a resource-efficient agricultural production in balance with nature, climate and the environment. The Commission produced 44 recommendations and 144 proposals of action by spring 2013 (Natur- og Landbrugskommissionen 2013).

The Commission recognized that pursuing the current environmental regulation with general fertilizer norms and limits on agricultural production would not be a cost-effective instrument as it would both be too costly for farmers and would not obtain sufficiently good environmental quality. The recommendations aim at changing fundamentally the regulation framework away from placing restrictions on production towards meeting local environmental targets and to create a complete regulation of nutrients with the primary aim of meeting the requirements in the WFD with regard to surface water. The recommendations included a more targeted and efficient environmental regulation:

- A new model for regulating the use of fertilizer in the fields.
- Conversion of several sensitive agricultural areas to areas with more extensive farming, nature, grass or perennial crops.
- Establishing new emissions-based regulation of stables and installations for livestock production, allowing for a higher nitrogen input than today depending on the retention in the area and the resultant nitrogen loads to the fjord.

The Commission conclude that with locally adapted and differentiated regulation of fertilizer use and the application of buffer strips and catch crops flexibly where it makes most sense, it's possible to obtain both a better aquatic environment and in some more robust areas increase production. Also, the emissions-based regulation of livestock production focuses on discharge requirements rather than size and design of production, allowing for innovation and new investments while keeping within environmental targets. The recommendations on these locally adapted and differentiated instruments are combined with general instruments such as requirement of buffer strips, catch crops etc. Farms with low retention fields and located in a catchment with high nutrient vulnerability should then move towards more extensive land use practices or land be taken out of production. Such changes in land use could be financed through increased flexibility (modulation) of EU CAP funds.
Spatial agricultural catchment models have been developed for two catchments: the Odense Catchment and the Limfjorden Catchment (the *TargetEcon* models) (Termansen, 2014), and these models allow for assessment of the least cost localisation of measures to reduce the nutrient loads to the fjords – such as fertiliser reductions, wetland restoration, catch crops and a large number of other measures (Konrad *et al.*, 2012). The models include data on the agricultural production at field block level in the catchments, the nutrient inputs and the retention of

nutrients in soil, groundwater and surface water in the sub-catchments. The models minimize the costs of achieving load reduction targets of nutrients, e.g. the load reductions required to fulfil the WFD. Utilisation of this characteristic of the models are applied in cost-minimising scenario modelling, and the main feature of the models is the ability to map the results regarding the implementation of measures, fertilizer inputs and resultant loads to the fjord. A similar model structure is used in Hasler *et al*. (2014), which models the entire Baltic Sea region. Since wetlands and the retention in soil, ground- and surface water are included, the models can be run to estimate the saved costs (or replacement costs) of these regulating ecosystem services. Furthermore, nitrogen uptake by mussel farm production in Limfjorden is included in the Limfjord *TargetEcon* model, and the model is used to estimate the cost-effectiveness of this type of measure, which utilize the regulating ecosystem service of the mussels, compared to other measures, but the models can also be used to assess the value of this nutrient uptake in terms of saved costs in agriculture. The models provide information about how the most cost-efficient solutions can be achieved because solutions are optimised. The modelling framework is usable for assessments of i) spatial targeting of measures ii) sensitivity analysis of the importance of the retention as regulating service and iii) the comparison of measures at land and sea, including the cost-effectiveness of utilisation of the regulating services in the sea. The model studies are not published yet, so more documentation is yet not available, but papers are in preparation.

5.3.2 Norway

The catchment area for river Morsa, situated in the south-eastern part of Norway, includes two counties (Østfold and Akershus) and eight local authorities. In 1999 "The Morsa project" was started as a local initiative with co-operation between local authorities, counties, and other local and regional actors. This river basin is an area with many user interests, like recreation and drinking water, and the water quality (water status) has been severe for a long time. Despite several national programs with the aim of reducing eutrophication, the water quality has not improved. Therefore it was a need for a special program in this area. The main purpose was to improve water quality in the catchment area, because of severe problems particularly related to eutrophication. This initiative came before the WFD was implemented, and represented a new and more locally oriented perspective on water management.

The Morsa project was re-organised as a catchment management council ("vannområdeutvalg") in 2007 to facilitate the implementation of the WFD. The strategy for the Morsa project has been to create voluntary participation among land owners based on motivation, counselling and co-operation. This included meetings with farmers, visiting farms, environmental planning for each farm, co-operation with research institutions, local programs of measures, and juridical binding agreements with farmers combined with economic incentives. Farmers were encouraged to sign an agreement whereby they would be compensated for extra costs of implementing a set of restrictions and measures that reduce phosphorus run-off for a period of three years (Magnussen and Holen 2011).

Box 5.3: Example points in the Morsa agreement

Points in the Morsa agreement:

- Use less Phosphorus fertiliser than nationally recommended level.
- No use of manure.
- No tillage during autumn.
- No growing of potatoes or vegetables in fields exposed to flooding.
- Establishment of 10 meter buffer zones along open water.
- Establishment of grass covered water ways in areas prone to erosion.
- Establishment of artificial wetlands if this is recommended.
- Accept experiments on potato/vegetable fields in order to increase knowledge on how to reduce phosphorus.

Source: www.morsa.org

Since the inception in 1999, the project has led to more than 2,000 households connecting their wastewater to public waste water treatment or improving decentralised waste water treatment; common local rules and control systems for drainage in 7 of 8 municipalities; increased area of agricultural land under reduced tillage from about 30% to close to 80%; phosphor fertilisation reduced by 50% in general and by 75% around one of the particularly vulnerable lakes; significant increase of buffer zones and afforested areas. The actions have to date cost about EUR 18.4 million in improved draining and sewage management and EUR 61 million in agricultural management changes (Vannområdeutvalget Morsa, 2012). As a result, water quality has improved in several rivers and lakes.

5.4 Farmers paid as climate adapters for cities

In addition to policy recommendations and research that attempt to develop more locally adapted instruments and measures (See 5.2), a pilot project and partnership from Denmark looks at the scope for setting up locally based contracts with farmers to deliver climate regulating services from their land in a partnership with cities. A number of towns and cities in Denmark are looking for more cost-effective and natural ways of reducing risks of flooding from increasing climate driven precipitation. While the objective of the scheme is adaptation to climate change, most of the practical measures will have positive impacts on local water quality levels.

The pilot project "landmanden som vandforvalter" [the farmer as a water manager] and a national pilot partnership "vandet på landet" [water in the countryside] [20] have recently been initiated in Denmark to develop innovative climate adaptation measures and instruments to protect cities from inundation due to more frequent cloudbursts during summer time and increased precipitation during winter months. Measures are thought implemented on agricultural land, and farmers paid for the water retention services of their land by the local municipality or region. Where the pilot project "the farmer as a water manager" looked at the business case and effects of individual measures on agricultural land, the pilot partnership "water in the countryside" models the practical dimensions and effects of using agricultural land for retention and storage at catchment scale.

Different measures have been suggested:

- Compensatory measure – a land owner receives a lump sum for entering into contract with the local municipality for making his land available for temporary inundations. When inundation occurs, an independent specialist assessor estimates the damage, and the farmer is compensated for the specific crop loss.
- Competitive tendering – the local municipality sets up competitive tendering where land owners can make their bids in terms of payment for retaining a certain level of water.

[20] https://www.landbrugsinfo.dk/miljoe/landmandensomvandforvalter/sider/startside.aspx (accessed 29.04.2014). http://ecoinnovation.dk/64690 (Accessed 29.04.2014).

In both cases, ideas are under development of how to incentivise farmers to cooperate such that larger contiguous areas within a sub-catchment area are created.

Different instruments have been described in fact sheets, and follow-up projects now look at designing and dimensioning at local catchment level. The instruments have potential synergy effects with the aims of the WFD and in all cases, negative effects are debated and sought eliminated. Examples of instruments include:

- *Intelligent buffer zones* – work by cutting drainage tubes and lead water to a ditch parallel to a stream. When water reaches the maximum level, it flows across the buffer strip to the stream. The buffer strip is forested with native trees, helping infiltrate water to the soil; uptake nutrients and create significant shadow that benefits biodiversity in the stream. This instrument has not been tested to date in Denmark, but experts reckon that it is more efficient in reducing nutrient leakage than in traditional buffer strips where drainage water flows below the buffer strip to the stream. At the same time, intelligent buffer zones retain and delay water during extreme weather events (Gertz, 2013).
- *Changes in watercourse management and water course renaturation* – many water courses in the Danish agricultural landscape have been straightened in the past and aim at leading as much water as fast as possible away. By reducing weed cutting in water courses and re-meandering water courses, water flow speed is reduced and water can be retained in targeted inundation zones. This can help protect downstream sensitive areas from flooding but also contributes to nutrient leakage reduction and sedimentation during flooding of meadows or through the increase of weeds in water courses, to enhanced biodiversity and more natural aquatic environment (Sørensen, 2014a; Kronvang, 2014).
- *Water retention in river valleys and wetlands* – controlled flooding of either existing wetlands or river valleys could avoid flooding downstream during cloudbursts. In the case of river valleys, a study of Brenstrupkilen in the vicinity to the city of Aarhus looked at the case of constructing three artificial basins crosswise in the valley with tubes allowing for water to flow normally during non-extreme events. Under extreme events, the tubes would slow down the water flow until water rises to the top of the barriers and flows to the next basin, and then to the third basin. In this way, the maximum water flow is reduced and risk of flooding downstream reduced. Experts

believe this would also help retain nutrients, in particular particle bound phosphorous; also physical conditions in the water course would be improved (Sørensen 2014b). In the case of flooded wetlands, a re-meandering of Odense stream, combined with an elevation of the bottom led to a reduced rate of flow and establishment of wetlands as the stream would flood more easily. These wetlands function today as reservoirs during extreme rain events. Added effects of creating and flooding wetlands include nutrient reduction, enhanced biodiversity, more natural aquatic environment and enhanced recreation services (Poulsen and Kronvang, 2014).

5.5 Nordic Payments for ecosystem services from restored/managed wetlands

Voluntary PES schemes exist in the Nordic countries that pay land owners for constructing/restoring and managing wetlands over several years, mainly in order to improve nutrient retention and biodiversity in intensive agricultural areas. The PES schemes are part of the EU financed agri-environment-climate instruments under the national Rural Development Programmes (RDPs). Wetlands provide a number of provisioning, regulating, cultural and supporting ecosystem services. In relation to the WFD, important services include the retention, recovery and removal of excess nutrients and other pollutants; groundwater recharge/discharge; and storage, recycling, processing, and acquisition of nutrients. Norway does not currently have PES schemes that provide financial support for the creation and maintenance of wetlands. Each Member State decides on a national RDP which is subsequently submitted to the European Commission for final approval. The RDPs from Nordic Member States were submitted in April 2014 in Denmark and Finland (Ministry of Food, Agriculture and Fisheries of Denmark, 2014; Ministry of Agriculture and Fisheries of Finland, 2014) and in June 2014 in Sweden (Swedish Board of Agriculture, 2014c). Typically, the approval process at the Commission takes 6 months. PES for wetland establishment and management are proposed to continue as a scheme under the RDPs 2014–2020 in all three countries. The following descriptions of national wetland PES schemes under the RDPs are based on the specific decisions of RDPs 2007–2013.

5.5.1 *Denmark*

The conditional performance contracts on wetlands in Denmark aim at reducing nutrient loads to water bodies; and at the same time produce co-benefits such as improved nature conservation, e.g. the creation of more habitats for birds (Ministry of Food, Agriculture and Fisheries, 2008). Eligible actions cover the creation and sustainable management of wetlands, lakes, natural meadows and other landscape elements by changing drainage, making wetter or creating waterholes, lakes and streams. The area must have a reduction potential of minimum 100 kg nitrogen/ha, be able to reduce a non-specified amount of phosphorous; have positive effects on wild flora and fauna and cause no net ochre leakage.

The payment part of the conditional performance contract between the State and land owners can be either via traditional compensation for income loss or an offer to reparcel agricultural land:

- Land owners are paid 100% of eligible investment costs up to a ceiling of EUR 2013/ha and a monetary compensation for loss of income over 20 years. The yearly payments are differentiated according to the prior land use: EUR 470/ha for former cropland; EUR 242/ha on former grassland; and EUR 40/ha for land not in agricultural production during the last 5 years. In addition to the 20-year contract, land owners are offered payments for managing the wetland in 5-year contracts. The contracts are set at five levels of fixed payments for different efforts of management varying from EUR 27/ha to EUR 540/ha. The condition for obtaining the payment is a wetland registration on a deed for 20 years.
- The AgriFish Agency offers land reparcelling to land owners who prefer to continue production on non-marginal land, typically closer to their farm. The Agency places a conservation easement on the wetland area and subsequently sells the land in a public tender.

An assessment of the previous wetland restorations in these two types of programmes has been made by Hansen *et al*. (2011), evaluating the different types of wetland restoration, the contracts and the effects of them. The conclusion was, among other, that the costs of wetland restoration varies a lot depending on the lost production at the farm but also because of differences in construction costs etc. The conclusion was furthermore that the negotiated contracts where the farmers were enrolled in a land rotation scheme where they could buy other parcels of land to compensate the lost land were better than lump-sum payments.

5.5.2 Sweden

A similar conditional performance contract exists in Sweden with a few differences in payment levels and the addition of the instrument to *improve* the effectiveness of existing wetlands (Swedish Board of Agriculture, 2014a&b). The objectives are explicitly to improve biodiversity and reduce negative effects of nutrient leakage from agriculture. Eligible actions for the contract include the creation, restoration, improving effectiveness of existing wetlands and management of created wetlands.

The payment part of the contract includes coverage of investment costs of up to 90% of actual costs. In particularly motivated cases, coverage is up to 100% of actual costs. Ceiling of compensation is set at either EUR 22,600/ha or EUR 45,150/ha depending on the region. Yearly management payments are offered at either EUR 339/ha or EUR 452/ha on cropland depending on the region and EUR 170/ha on grassland and other fields. Yearly management payments when improving effectiveness of existing wetlands: EUR 102/ha.

5.5.3 Finland

The conditional performance contracts on wetlands in Finland in the RDP 2007–2013 period targeted areas where fields cover more than 20% agricultural of the catchment area, notably the catchment areas of rivers flowing to the Gulf of Finland, Archipelago Sea, Bothnian Sea, the Kvarken Archipelago, the Bay of Bothnia as well as catchment areas of lakes with intensively managed agriculture. Eligible actions for the contracts comprise construction and management of multifunctional wetlands or wetland-like flooding areas in places where they would be naturally formed, on arable areas susceptible to flooding, on terraced drainage areas as well as the restoration of natural streambeds (Berninger *et al*. 2012).

Based on modelled and observed research, the Finnish RDP 2007–2013 introduced the requirement of a relative minimum size for new wetlands in order to ensure optimal nutrient leakage reduction. Research showed how misleading it can be to focus solely on the mass retained per unit wetland area as a measure of wetland efficiency. Although maximum retention efficiency per hectare of wetland is reached in small wetlands, large wetlands are more efficient in retaining large quantities of nutrients lost. In practice this is done by requiring a minimum wetland-to-catchment ratio when determining the minimum size wetland, which should be at least 0.5% of upstream catchment area. The maximum support level for construction is euro 11,500/ha of wetland and EUR 3,226/ha of wetland if these are between 0.3–0.5 ha. These payment levels were increased by 2010. Payments for maintenance are dependent upon cost estimates made by the land owner but with a ceiling of EUR 450/ha/year. Contract length can be made for either 5 or 10 years.

The current draft RDP 2014–2020 extends the eligible areas for wetland restoration contracts to areas where fields upstream in the catchment area cover at least 10% (compared to 20% in previous programme) (Ministry of Agriculture and Forestry of Finland, 2014).

5.6 Watershed programmes

A number of voluntary and mandatory PES programmes are found in Europe and the US that relate to paying land owners for action that lead to improving water quality in a catchment area. Most of these types of programmes aim at improving drinking water quality either from groundwater or from surface water, to replenish aquifers and to protect general catchment functions against pressures of development. In Europe, 15 payment programmes have been identified by the non-profit association Ecosystem Marketplace in the latest State of Watershed Payments 2012 (Bennet *et al*. 2013). In three instances, private beverage companies have initiated compensation mechanisms while the other nine are driven by a utility or municipality engaging private forest landowners and farmers in protecting drinking water supplies. We describe two examples of catchment programmes initiated by municipalities, one in Munich, Germany, the other in the state of New York, US.

5.6.1 Munich watershed programme

Grolleau and McCann (2012) describe and evaluate two payments for ecosystem services at watershed level, the Munich and New York City watershed programmes that have used ecosystem based and locally adapted measures at watershed level to ensure adequate water quality for drinking water purposes. This section describes the conditional payment scheme in Munich watershed and the following the scheme in the state of New York.

Munich Waterworks noticed during a prolonged period (1974 to 1992) a slow but significant increase in nitrate and pesticides in the drinking water, which originate from springs about 40km from the city. Although the levels at the time were below regulatory requirements for tap water, the city decided in 1991 to undertake a targeted ecosystem based framework to improving water quality (or avoiding any further deterioration). The targeted area was composed of forests and conventional dairy farms (120 farms covering 2,250 ha), a total of 6,000 ha.

A first public information campaign towards the farmers encouraging them to switch to more environmentally friendly practices was met by reluctance as those practices would fundamentally change their production method. The city then moved to organizing farmer meetings providing information, guidance and possibility of financial incentives. At first the financial incentives were tailored to different types of practices, e.g. one level of incentive for limitations in nitrogen use, another level of incentive for transition to pasture etc. The practice by practice framework however was abandoned as it would have required a too expensive set of monitoring and verification system.

Instead the city took a more comprehensive approach by offering farmers initial support to switch to organic farming. The city cooperated with three different organic producer associations and offered to pay for the first evaluation by producers' union on the suitability of the farms to convert towards organic farming. This initial step helped overcome uncertainty about what a switch would entail in practice. Then, contracts that follow the existing organic farming standards were offered farms in the targeted area; farmers could choose which of the three standards they would opt for. The city offered an annual per hectare payment for the first six years of the contract (EUR 280/ha/year) and a bit lower payment for the following 12 years (EUR 230/ha/year) reflecting the loss of income and need for investments for the conversion. In addition, farmers could benefit from European CAP subsidies for five years (EUR 250/ha/year). On average, payments for farmers choosing to convert to organic farming amounted to more than EUR 10,000/ha/yr. The city developed a flexible framework such that for farms that couldn't convert to organic farming could sign up for adopting practice changes that favoured water quality. These farms were offered EUR 200/ha/yr and have the status of "supporting members". Separate controls are made on the supporting farms by independent examining teams and the state level Department of Agriculture.

The targeted ecosystem based framework to ensuring water quality in Munich city proved to be a cost efficient arrangement compared to cleansing water at end-of-pipe. Several circumstances contributed to this cost-effectiveness:

- Firstly, the city capitalized upon the expertise and experience of organic producers unions to convince farmers to convert to organic farming practices.

- Secondly the city saved on enforcement costs as monitoring and verification of organic farms are performed by third party certification bodies.
- Thirdly, the farmers were both eligible to European subsidy for converting to organic farming *and* locally financed performance payments. The financial synergies reduced the local level of performance payments.
- Finally, payments were limited in time as previous evidence shows that organic farming after initial 7–8 years are more profitable than conventional farming. The risk of farmers dropping out of the certification scheme and reverting to conventional unfriendly water quality practices is therefore very limited.

By now, 80% of the agricultural area in the targeted area, representing 110 of the 120 farms, is now under contract. It is considered to be the largest contiguous area of organic farming in Germany. In terms of drinking water quality, nitrate concentrations in drinking water have dropped from 15 mg/l to 7 mg/l and pesticide concentrations have also decreased significantly. The price increase for the whole water scheme to consumers is estimated to about EUR 0.005/m^3 (SVM, 2005; cited in Grolleau and McCann, 2012), while the estimated avoided costs of water treatment equipment was estimated at EUR 0.23/m^3 (Simonet 2005; cited in Grolleau and McCann 2012).

5.6.2 New York Watershed Programme

New York is not only the largest city in the US it also has the largest unfiltered water supply in the country, supplying some 9 million inhabitants. Water is supplied from 518,000 ha in the Catskill Mountains and Hudson Valley regions. In the 1980s, changes in land use in the Catskill Mountains threatened the filtering capacity of the ecosystem, compromising drinking water quality. This included more intensive agricultural practices, faulty sceptic tanks and increased discharges from waste water treatment plants as well as increased erosion and reduced filtering capacity of the ecosystem as residential areas and related impermeable infrastructure expanded in the mountains.

The City of New York would normally have had to invest USD 4–6 billion in a filtration plant (plus ¼ billion yearly management costs) in order to comply with the Safe Drinking Water Act Amendment of 1986. This would have doubled the cost of water and deteriorated the taste significantly. An alternative was to successfully petition the US EPA for a "filtration avoidance determination" through a comprehensive watershed management programme involving landowners and communities to build infrastructure and improve natural filtration. The City opted for the natural filtration framework which cost the City USD 1.4 billion in payments to farmers and other landowners to implement changes to preserve water quality.

Activities for which the City of New York paid included the development of innovative agriculture and forestry programmes; conservation easements;[21] upgrading sewage treatment plants and septic systems; as well as buying up land. Farmers could sign up for a "Whole Farm Planning Program" to reduce pollution by introducing best management practices. The programme was voluntary but farmers had to commit to getting 85% of farmers on board within five years, otherwise the city of New York would impose restrictive regulation.

By 1998, ca. 500 dairy and livestock farms had signed up; within five years, 93% of farms in the Catskill Mountains had signed up and New York City has since 1993 avoided having to filter their drinking water.
Despite several references to the case of the New York Watershed Programme [e.g. Chichinilsky and Heal, 1998; Pagiola *et al*., 2004; Grolleau and McCann, 2012], one author has contested the basis of the case (Sagoff, 2002): water quality had not decreased prior to the watershed programme, but due to new legislation (the Surface Water Treatment Rule), the City of New York had to either start an expensive technical filtration of drinking water or petition for a filtration avoidance determination by complying with EPA requirements to improve natural filtration services in the catchment area.

5.7 Water quality trading

Different types of water quality trading programmes exist in practice, all operating at catchment level (Selman *et al*. 2009):

- Point-to-point source trading – trade between regulated point sources, e.g. two sewage treatment plants where both seller and buyer are subject to regulatory discharge permits. This type of trade

[21] A conservation easement is an instrument where conservation agencies buys up land, places a conservation status on the land, limiting the type of land use practices on the land and then sells it again.

does not relate to the use of ecosystem services to obtain emission reductions and we do not describe this further.

- Trading between regulated point sources and non-regulated non-point sources – regulated dischargers can purchase pollution reduction credits (also known as offsets) from nonpoint sources with lower costs, e.g. between sewage treatment plants and farmers (See Box 5.4).
- Trading between non-point sources – one or both of the non-point sources involved in the trades have been regulated (See 5.7.1 Nitrogen Sourcing and Trading).
- Trading between point sources/nonpoint sources and nutrient sequestrators (See 5.7.2 Compensation Mussel Offsets).

Stanton *et al*. (2010) identify some 66 water quality trading programmes based in the US, four in Australia, one in Canada and one in New Zealand. Cap-and-trade systems also exist to control air pollution such as for greenhouse gasses (e.g. EU ETS, New South Wales Greenhouse Gas Abatement Scheme (Australia), NZ ETS (New Zealand), Regional Greenhouse Gas Inventory (RGGI, US)), sulphur dioxide allowance-trading programme (US), Volatile Organic Compounds (State of Illinois, US) and wood burning (Telluride city council, Colorado, US). Water quality trading programme exist mostly in the US, but is also found in Canada, New Zealand and up to 2010 was also attempted in Sweden (See Section 5.7.2).

Box 5.4: Ohio River Basin Trading Project – Trading between point sources and nonpoint sources

Ohio River Basin Trading Project is the world's only interstate water quality trading program that started operating in March 2014. The project trades surface water quality across three US states, Ohio, Kentucky and Indiana. The Ohio River is the largest tributary to the Mississippi river meandering through eight states westward from Pennsylvania to Illinois. Twentyfive million people live within its basin and three million rely on the river for their drinking water supply (Barret 2014). Effluent pollution from city run off, waste water treatment plants (WWTPs), power plants and agricultural land has deteriorated water quality while inter-state water regulation has made it difficult to obtain acceptable results. The trading project attempts to reduce nutrient pollution flowing into the Ohio River by 30 tonnes of nitrogen and 15 tonnes of phosphorous over a five year period by paying farmers to keep nutrients from reaching the river through conservation practices. Farmers then sell emission reduction credits to point source polluters. The area would then be subject to a cap, i.e. a maximum amount of nutrient emissions allowed to the river. At full scale, the project could create a market that fits all eight states, allowing for the participation of 46 power plants, thousands of wastewater utilities and 230,000 farmers. Starting out though, the project includes up to 30 farmers in a pilot run.

Source: Barret, K., (2014).

5.7.1 Nitrogen Sourcing and Trading in the Lake Taupo Catchment, New Zealand – Trading between non-point sources

Lake Taupo is the largest lake in New Zealand and is known together with the surrounding catchment area as an iconic feature of the North Island of New Zealand. The water quality is still exceptional but land use development from intensified agriculture and expanding urban areas have over the past 30 to 50 years gradually led to increasing levels of nutrients in the lake and subsequent declining water clarity. With a considerable time lag between the activity causing nutrient discharge and the consequences on water quality indicators as well as the concern to limit costs to the agricultural sector and urban development, Wakaito Regional Council chose to cap the amount of nitrogen entering the lake from agricultural and urban areas and to allow nitrogen trade between land owners (http://www.waikatoregion.govt.nz/). The target of the policy is to maintain current water quality well into the future (by 2080) by capping all sources of manageable nitrogen at their 2001 levels (Acting Group Manager 2012). This corresponds to a reduction of 20% of nitrogen discharges across the catchment area.

Each landowner in the Lake Taupo catchment area now has an individual nitrogen discharge allowance corresponding to the average emission level between 2000 and 2005. Any landowners wishing to increase their nitrogen discharge need to purchase allowances from other landowners in the catchment area. In addition, a separate fund, the Lake Taupo Protection Trust, was established with a mandate to achieve the 20% reduction in nitrogen emissions primarily through a mix of land retirement, land conversion and purchasing allowances thereby retiring permits to pollute from the market.

Evaluation of the nitrogen sourcing and trading programme shows the policy has successfully limited additional nitrogen leaching that would otherwise have been emitted in the absence of regulation, especially from expansion in dairy farming (Duhon *et al*. 2011). The Lake Taupo Protection Trust had entered into agreement with farmers to remove 100 tonnes of nitrogen by July 2011 and was ahead of schedule to secure its required 153 tonnes of nitrogen reductions by 2018. Trading among farmers is picking up, but is expected to increase in future as productivity increases.

5.7.2 Compensatory mussel offsets – Trading between nutrient dischargers and nutrient sequestrators

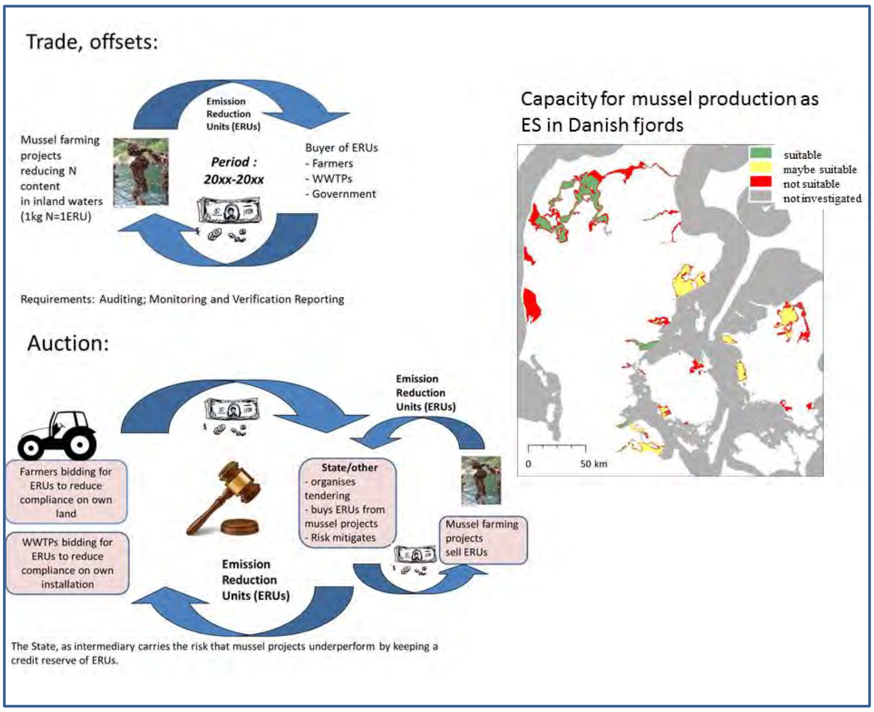
A 3-year research project in Limfjorden, Denmark, investigated the economic and environmental feasibility and effects of setting up compensatory mussel farming that would remove excess nutrients from coastal waters in order to improve surface water quality (http://forskning.skaldyrcenter.dk/muslinger-som-virkemiddel-mumihus/). Mussel farmers would be paid to establish and harvest additional cultured long-line produced blue mussel colonies. The blue mussels provide a nutrient regulating service by removing nutrients from the water column. When harvesting these mussels, nutrients are removed from coastal waters. Mussels can in turn be utilised as fertiliser on agricultural land, bringing back the nutrients bound in the mussels from sea to land, reducing demand for artificial fertilisers and increasing the source of phosphorous.

The pilot project showed a total amount of nutrients incorporated in the mitigation mussels between 10.5–16 t nitrogen and 0.5–0.7 t phosphorous, corresponding to a potential nutrient removal of 0.6–0.9 t nitrogen per ha per year and 0.03–0.04 t phosphorous per ha per year (Petersen *et al*. 2014).

Trading of nutrient emission units could be possible between compensatory mussel producers as sellers of reduced emission units and buyers of emission reduction units, for instance i) farmers who could be allowed to either increase the current level of fertiliser use or avoid meeting future more stringent restrictions on fertiliser use; ii) waste water treatment plants as part of their compliance with urban waste water treatment Directive; or the State as a way of cleaning up past emissions of excess nutrients stored in the sediments and water columns. Box 5.5 illustrates the trading mechanism and the potential for compensation mussel production in coastal waters in Denmark.

A compensatory mussel offset project was lauched in Lysekil municipality, Sweden in 2004, where the waste water treatment plant off-set part of their nutrient emissions through compensatory mussel production, thereby avoiding a more expensive extension of the treatment plant. A private entity obtained the contract to harvest 3,500 tonnes of mussels yearly which would offset 39 tonnes of nitrogen emitted from the waste water treatment plant (Lindahl and Kollberg 2008). In addition to income generated from nutrient removal from the water body, the mussel producer had based its business plan on the sale of mussels for human consumption as well as for fertiliser and fodder. Irregularities in the management and statutory control of the mussel production for safe human consumption, however, led to police charges against the company (FiskeribladetFiskaren, January 2010). The waste water treatment plant therefore ceased the contract by 2010 and invested in its own treatment unit. Combined with difficulties in finding readily open retail markets, the mussel producing company closed down.

Box 5.5: PES for shellfish/mussel production in Denmark



Source: Petersen *et al*. (2013).

5.8 Main findings in this chapter

This chapter has focused on market-based policy examples and approaches for managing non-point pollution from land use in (primarily) agriculture. Agriculture is recognised as the main contributor of nutrient pollution to the aquatic environment while receiving substantial public funding.

Funding for voluntary agri-environmental policies under the new EU CAP 2014–2020 for EU Member States has been significantly strengthened compared to the former CAP (from 10% of CAP budget to 30%) and a new mandatory Green Direct Payment representing one third of Pillar I budget has been introduced for recipients of first Pillar support, of which several instruments and requirements can have a direct beneficial impact on the aquatic environment and ecosystem services. The strengthening is notable despite an overall cut in the EU CAP budget of 1.8% of Pillar I and 7.6% in real terms (2011 prices) compared to the previous CAP period. However, the new cross-compliance rules, which represent one of the two instruments available to integrate water policy objectives in the CAP, do not include rules on sustainable pesticide use and integrated pesticide management nor on the basic measures under the RBMPs (WFD's Article 11.3). Although the Commission has subsequently been asked by the EU Parliament and the EU Council to propose to include relevant parts of the WFD, this is not due until obligations for farmers have been clarified in all Member States. Substantial additional funds made available under the EAFRD funds to tackle water management as one of the "crucial new challenges" for European agriculture has only sparingly been implemented, despite the greening of the CAP. Still Member States can choose to spend up to 15% of the CAP on Pillar II for the benefit of i.a. more wetland areas and afforestation, and the mandatory 5% Ecological Focus Areas further strengthen ecosystem services on agricultural land, of which for instance buffer strips may benefit the aquatic environment.

Non-point pollution is difficult to control in practice, in particular when using uniform instruments that ignore differences in soil retention capacities, farm typologies and farmer characteristics. This so-called wicked problem requires a mix of instruments and measures that are adapted to local conditions as well as the involvement of a mix of stakeholders. The three examples from Morsa watershed in Norway, Munich in Germany and Catskill Mountains in the State of New York, USA, represent programmes at watershed levels that produce significant and positive results for water quality within relatively few years using the ES framework and to a large extent PES, and involving a mix of stakeholders, both beneficiaries, polluters and ecosystem service providers. Common for the programmes is locally adapted measures and instruments, some voluntary and others mandatory; an appropriate mix of different policies and the active involvement and engagement of land owners and households. The examples indicate that it's possible to obtain significant results at lower costs within relatively few years when making a concerted effort at catchment level with all relevant stakeholders. Despite the apparent opportunities and benefits, we have not come across many examples of this type in general or for the Nordic countries in particular.

At a national level, the Danish Nature and Agriculture Commission recommended moving towards more targeted and efficient environmental regulation of agricultural discharges that is based on actual loads and vulnerability of individual recipients. This represents a fundamental change in how water pollution is regulated today, and currently the Danish Ministry of Environment is analysing and testing how new regulatory models can be designed to target the regulation more to the vulnerable areas and thereby differentiate the nutrient regulation more compared to current practice. There are modelling tools available (the TargetEcon models) for this testing (e.g. Konrad *et al.*, in prep.; Hasler *et al.* 2014) where examining assumptions of effort distribution between farmers can be important, e.g. the assumptions on retention of nutrients in sub-catchments, both for economic and ecological reasons.

The idea of developing locally adapted PES instruments at catchment level was also part of pilot projects in Denmark to look at how farmers could enter contracts with towns and cities to provide ecosystem services on their land that would regulate excess water and avoid inundations in the built environment. Many of the practical measures investigated could also have positive impacts on water quality. This exemplifies the scope for mainstreaming policies at catchment level i.e. with a firm basis set on local conditions and a clear understanding of which specific ecosystem services are enhanced for the benefit of whom and when.

Wetland PES schemes, which have a direct relevance to the WFD, are found in the three Nordic EU Member State countries, co-financed through the second Pillar of the EU CAP. Whereas the measure and objectives are largely similar across the countries, the payment levels and conditions in the contracts differ. In Sweden, payment is also offered for the *improvement* of existing wetlands and in Finland wetland-to-catchment ratios between wetland area and total catchment area are used to guide the required *minimum size* of wetlands, recognising that larger wetlands are more efficient in retaining as large a quantity of nutrients lost from upstream fields as possible than small wetlands, even if these are more efficient on a per unit wetland basis. Norway appears not to use wetlands as a measure to combat excess nutrient leakages. This could be something to look into for Norway.

Water quality trading does not currently exist in the Nordic countries or in the EU but could in principle be established as an instrument at river basin level as a cost effective way of reducing emissions. The EU Commission proposed in the Communication "A Blueprint to safeguard Europe's water resources" (COM 2012; 673) to develop CIS Guidance on trading schemes by 2014. Water quality trading exists primarily in the US where about 66 schemes are in operation, spurred by the Clean Water Act from 1972. Voluntary off-sets of nutrient loads to recipients has been attempted through compensatory blue-mussel farming in Lysekil municipality, Sweden, but stopped due to irregularities with the ecosystem service provider. A full-scale pilot in Denmark has recently been carried out, indicating that compensatory mussel farming can be both an environmental and economic efficient and effective measure. Water quality trading can be an instrument to save costs across polluters when complying with abatement targets, but it necessitates a sufficient market size to bring about cost savings, which a catchment level market may or may not provide, depending on local conditions.

When targeting economic policy instrument to catchment or even sub-catchment levels the challenge becomes striking the right balance between policies and measures that make sense locally while keeping transaction costs down in relation to management, coordination and control of both measures and policies. The US CRP uses effects on the environment as determinant for paying land owners for ecosystem services rather than measures. This can be one approach to circumvent the issue of high transaction costs when targeting policies and measures, but necessitates that effluent reductions are both measurable and controllable. The problem with measurement, monitoring, control and enforcement has in many cases led to water protection policies directed towards restricting the inputs of e.g. nutrients to agricultural crops or to improved utilisation of the nutrients by catch crops and requirements for manure handling by general measures. A regulation where the provision of ecosystem services is more in focus necessitates a more targeted regulation where focus is on the outcome in the water bodies instead of the inputs to the fields, but still the problem of measurement and control is severe. Setting up differentiated point based systems for payments for ecosystem services, based on scientific evidence of different outcomes between localities, could be an avenue to make different local measures comparable in terms of environmental effects and hence more cost-effective to manage.

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Norsk sammendrag

Abstract

Økosystemtjenester (ØT) er økosystemenes bidrag til menneskelig velferd. Økosystemtjenester kan klassifiseres, kartlegges og vurderes innenfor en ØT-tilnærming, som bygger på en forståelse av sammenhengen mellom økosystemer og menneskelig velferd. Formålet med dette prosjektet er å utforske bruken og anvendeligheten av ØT-tilnærmingen i vannressursforvaltning, spesielt forvaltning i tråd med vanndirektivet (EUs Water Framework Directive) i de nordiske land. Det finnes en rekke eksempler på bruk av ØT-tilnærmingen i studier knyttet til vanndirektivet i alle de nordiske land. De fleste studiene inkluderer kartlegging, beskrivelse og kategorisering av økosystemtjenester, mens det er færre nyttekostnadsanalyser og analyser av uforholdsmessige høye kostnader ved å oppnå målsettingen om bedre vannmiljø. Relativt få nordiske studier verdsetter økosystemtjenester fra ferskvann som sådan. Noen flere verdsetter forbedret vannmiljø, inkludert det å nå direktivets målsetting om god økologisk tilstand. I disse benyttes ikke ØT-tilnærmingen eksplisitt, men direkte eller indirekte kan man utlede hvilke økosystemtjenester som er vurdert og verdsatt. Det finnes en rekke eksempler på bruk av målrettede og lokaltilpassede virkemidler i de nordiske land, først og fremst innenfor landbruket. Lokaltilpasning og bruk av ØT-tilnærmingen understrekes, men det er ofte ingen direkte sammenheng mellom forbedrede økosystemtjenester, de økonomiske mekanismene og størrelsen på betalingen for økosystemtjenestene. Rapportens eksempler viser at ØT-tilnærmingen er på vei inn i nordisk vannressursforvaltning. Det er fortsatt behov for mer kunnskap om økosystemtjenester og verdien av dem i vann, men eksemplene og diskusjonen i rapporten viser at ØT-tilnærmingen kan være til stor nytte i sammenheng med nordisk vannressursforvaltning, inkludert implementering av vanndirektivet.

Bakgrunn

Økosystemtjenester (ØT) er økosystemenes bidrag til menneskelig velferd. Ved ulike klassifiseringssystemer kan økosystemtjenester kartlegges og vurderes innenfor en ØT-tilnærming, som bygger på en forståelse av sammenhengen mellom økosystemer og menneskelig velferd.

I prosjektet VALUESHEDS ("Valuation of Ecosystem Services from Nordic Watersheds", se Barton *et al*. 2012) og flere andre prosjekter om økosystemtjenester i Norden, er det lagt vekt på å beskrive og kartlegge de økosystemtjenester vi får fra ulike økosystemer. Det er nå behov for å utforske videre hvordan man kan integrere og bruke lærdommen fra arbeid med ØT-begrepet og -tilnærmingen i praktisk forvaltning.

ØT-tilnærmingen er ikke en del av vanndirektivet, men når man diskuterer bruk av ØT-tilnærmingen i ferskvannsøkosystemer, kan det være hensiktsmessig å knytte an til vanndirektivet, som er en viktig pilar for nordisk vannforvaltning. Å undersøke hvilken rolle ØT-tilnærmingen kan spille for forskjellige vannforvaltningsoppgaver generelt, og oppgaver som følger av vanndirektivet spesielt, anses som et naturlig steg i vurderingen av økosystemtjenester i ferskvann.

Mål med prosjektet

Prosjektets mål er å utforske bruken og nytten av ØT-tilnærmingen i forvaltning av ferskvannsressurser i Norden, spesielt knyttet til følgende fire temaer:

- Metoder for å benytte ØT-tilnærmingen ved vurdering av nytten av økologiske forbedringer i vassdrag.
- Metoder for kostnadsvurdering, særlig det vanndirektivet betegner som uforholdsmessig høye tiltakskostnader.
- Hvordan ØT-tilnærmingen kan bidra til utvikling av målrettede og lokalt tilpassede virkemiddel- og tiltakspakker på nedbørfelt-/vannregionnivå.
- Mulighetene for bruk av betaling for økosystemtjenester ("Payment for Ecosystem Services, PES") som et virkemiddel for målrettet ferskvannsforvaltning.

Vår tilnærming

De fire hovedtemaene, nevnt i avsnittet over, har til en viss grad vært beskrevet og diskutert tidligere i sammenheng med vanndirektivet. Rapportens hovedbidrag er å gi eksempler på hvordan ØT-tilnærmingen har blitt benyttet, hovedsakelig i en nordisk sammenheng. Mens VALUESHEDS-rapporten diskuterte grunnleggende metodiske og prinsipielle spørsmål, vil denne rapporten legge mer vekt på praktiske problemstillinger og gi eksempler. Vi gir ikke en komplett oversikt over nordiske studier om økosystemtjenester her, siden dette foreligger i Barton *et al*. (2012). Vi har valgt eksempler med tanke på å illustrere bruksområder for ØT-tilnærmingen i forskjellige land og med forskjellige hensikter, i håp om at de kan inspirere og være til nytte.

Økosystemtjenester, betaling for økosystemtjenester og EUs vanndirektiv

ØT-tilnærmingen har fått mye oppmerksomhet, og verden rundt legges det ned betydelig arbeid i å utvikle økosystemtjenestebegrepet videre og implementere det i praktisk forvaltning. ØT-tilnærmingen kan brukes for å kartlegge og måle verdien av endringer i støttende, forsynende, regulerende og kulturelle økosystemtjenester, og avveininger mellom disse.

Vanndirektivet er det viktigste direktivet for regulering av kvalitet og bruk av fersk- og kystvann i EU-land. Norge og Island har også innført direktivet i sin lovgivning.

Målet med direktivet er å opprettholde og forbedre vannmiljøet, med særlig vekt på økologisk og fysisk-kjemisk kvalitet i de omfattede vannområdene. Direktivets mål er å oppnå god økologisk tilstand (GØT) i alle vannforekomster, og godt økologisk potensial (GØP) i vannmasser klassifisert som sterkt modifiserte. Hovedområdene der økonomisk analyse innenfor direktivet kan knyttes til ØT-tilnærmingen er karakterisering av nedbørfelt (Artikkel 5), bruk av vannprising og kostnadsdekning (Artikkel 9), vurderingen av uforholdsmessige kostnader (Artikkel 4), og krav til identifisering og implementering av kostnadseffektive kombinasjoner av tiltak for å oppnå god økologisk status i vannforekomstene, som del av tiltaksprogrammet (Artikkel 11).

Vanntjenester defineres som del av direktivets artikkel 2(38) ("definisjoner"):

> "Vanntjenester er alle tjenester som forsyner husholdninger, offentlige institusjoner eller annen økonomisk aktivitet med (a) abstraksjon, oppdemming, lagring, behandling og distribusjon av overflate- eller grunnvann, (b) fasiliteter for innsamling og behandling av avløpsvann, som tilbakeføres til overflatevann."
>
> EU-kommisjonen, 2000

Økosystemtjenester er altså en bredere definisjon av tjenester enn de vanntjenestene som er definert i vanndirektivet. Vi mener likevel at ØT-tilnærmingen kan være nyttig i analyser som er knyttet til implementering av vanndirektivet.

Vår gjennomgang viser at ØT-tilnærmingen kan være nyttig for å vurdere og illustrere hvordan ulike økosystemtjenester påvirkes av ulike valg av tiltak og virkemidler for å oppfylle vanndirektivets målsettinger, og avveiningen mellom forskjellige goder og tjenester. Særlig kan ØT-tilnærmingen bidra til å illustrere hvordan forskjellige strategier for å oppnå målsettingen om godt vannmiljø kan føre til forskjellige resultater for forsyning av ulike økosystemtjenester, og dermed vise forskjellene mellom den totale nytten av forskjellige tiltaksstrategier, og hvordan nytten fordeles mellom ulike brukere, tid og sted. ØT-tilnærmingen gir mulighet for å vurdere nytten av positive miljøendringer i et komplekst økosystem med et metodisk fundament for sammenhengen mellom endringer i økosystem og følgende endringer for ulike økosystemtjenester. ØT-tilnærmingen kan derfor bidra til å forbedre metoder for vurdering av uforholdsmessige kostnader i vanndirektivet, samt være til hjelp ved analyse av tiltaks-programmet, og ved vurdering av ulike tiltaks kostnadseffektivitet.

ØT-tilnærmingen er en av flere som ligger til grunn ved utforming av økonomiske virkemidler for å redusere forurensning av vann. Betaling for økosystemtjenester ("Payment for Ecosystem Services, PES), går ut på at de som "produserer" økosystemtjenester får betalt for dette. Kvotehandel ("cap-and-trade") med vannkvalitet er et annet eksempel på et virkemiddel som er basert på ØT-tilnærmingen, der økosystem-baserte kvoter for f.eks. mengde med forurensende stoffer (som nitrogen og fosfor) omsettes mellom forurensere. PES-systemer for å redusere vannforurensing er i bruk i de nordiske land og ellers i Europa. PES-systemene er ikke initiert som følge av vanndirektivet, men er ofte forankret EUs felles landbrukspolitikk (CAP), eller i virkemidler knyttet til drikkevann, men disse bidrar likevel til å møte kravene i vanndirektivet og kan potensielt spille en større rolle i vanndirektivet enn de gjør i dag.

Felles for virkemidler som tar sikte på å forbedre vannkvalitet, er en økende erkjennelse av at de må tilpasses lokale forhold, fordi både kostnader og nytte (økosystemtjenester) varierer fra område til område.

Bruk av ØT-tilnærmingen for beskrivelse og verdsetting av nytteeffekter av forbedret økologisk tilstand i vann

De nødvendige trinnene for nyttevurderinger av forbedringer i vannmiljøtilstand basert på ØT-tilnærmingen, er identifisering/beskrivelse, kvantifisering og verdsetting. Identifisering av økosystemtjenester kan gjøres og blir gjort på forskjellige geografiske nivåer (vannforekomst, vassdrag, nedbørfelt, land, region) avhengig av formål. I noen studier er identifikasjon og verdsetting gjort med fokus på én eller noen få, utvalgte økosystemtjenester. I sammenheng med vanndirektivet er det mest interessante spørsmålet hvordan nytteeffektene fra økosystemtjenester endres (øker) når målet om god økologisk tilstand nås.

En gjennomgang av studier om økosystemtjenester fra ferskvann og forbedringer i ferskvannstilstand viser at det er krevende både å identifisere, kvantifisere, og verdsette, nytteeffekten av å oppnå god økologisk tilstand.

Det er mange interessante eksempler på bruk av ØT-tilnærmingen for å identifisere, kvantifisere og verdsette nytteeffektene fra ferskvann generelt, og forbedringer i ferskvannsforhold (økologisk og kjemisk status i direktiv-terminologi) spesielt, særlig i de nordiske landene. Fram til nå har de fleste studiene ikke, eller i liten grad, tatt hensyn til behovet for å vurdere avveininger eller dobbeltelling. I den vitenskapelige litteraturen om økosystemtjenester pågår det en diskusjon om disse temaene. ØT-tilnærmingen er fortsatt ny i forvaltningssammenheng, og hittil er hovedvekten lagt på hvilke ØT som påvirkes, og hvordan de kan beskrives og kartlegges. Trolig vil problemstillingene knyttet til avveininger og dobbeltelling bli tillagt større vekt i takt med at tilnærmingen blir mer anvendt.

ØT-tilnærmingen kan være et verktøy for systematisk identifisering av nytteeffekter og for å undersøke sammenhengen mellom økologiske endringer og velferdsøkninger, og eksemplene viser at tilnærmingen er i ferd med å bli tatt i bruk i de nordiske landene. Likevel er tilnærmingen åpenbart ikke noen "quick fix". Mye arbeid er fortsatt nødvendig på alle områder knyttet til identifisering, kvantifisering, klassifisering og ikke minst verdsetting av økosystemtjenester, både med tanke på det økologiske fundamentet og de økonomiske metodene.

Vurdering av uforholdsmessige kostnader

Det er relativt få eksempler på nytte-kostnadsanalyser i vanndirektiv-sammenheng, og enda færre slike analyser der økosystemtjenester brukes i nyttevurderingen. Dette gjelder både for Norden og Europa forøvrig.

Martin-Ortega (2012) konkluderer i sin artikkel om bruk av økonomiske metoder i implementering av vanndirektivet at "… while CEA ["Cost Effectiveness Analysis" – vår merknad] has been widely adopted by most national guidelines in Europe, and the estimation of the environmental benefits has received a significant attention from the literature, the way these two should be joined up in a CBA has received much less attention."

Vi kan legge til at selv om nytteeffekter estimeres, er ikke ØT-tilnærmingen i utbredt bruk. For eksempel verdsetter mange studier "god vannkvalitet", som er målet med vanndirektivet, men det kan være vanskelig å innhente informasjon om verdien av spesifikke økosystemtjenester, som rekreasjon, fiskeri og fiskehabitater osv. fra slike studier. ØT-tilnærmingen representerer derfor et nytt konsept i verdsettingsstudiene.

Det finnes likevel eksempler på nasjonalt, regionalt og lokalt nivå der ØT-tilnærmingen er brukt for vurdering av uforholdsmessige kostnader, hovedsakelig som screening-prosedyrer. Et eksempel er Jensen *et al.* (2013) som benytter informasjon om økosystemtjenester fra Aquamoney-studien, dvs. resultatene av den økonomiske verdsettingen av forbedringer i vannkvalitet og økologi i vannregion Odense, i en nytteoverføring til andre danske vannregioner. Resultatene av nytteoverføringen brukes deretter til en nyttekostnadsanalyse for gjennomføring av direktivet i Danmark. Nyttekostnadsanalysen brukes som en konservativ screening der kostnadene ser ut til å være uforholdsmessige, dvs. at de overstiger nytteeffektene av økosystem-forbedringer. Mye av samme prosedyre og tilnærming benyttes på lokalnivå for to elver i Oslo som en screeningprosedyre for å vurdere nytteeffekter og potensielt uforholdsmessige kostnader (Magnussen *et al.* 2014).

ØT-tilnærmingen anses som nyttig, fordi den bidrar til en systematisk og dekkende oversikt over alle nytteeffekter (i økonomiske enheter, fysiske enheter, og/eller kvalitativt beskrevet) som er nødvendig for å vurdere nytten av forbedringer i vanntilstand. Konklusjonen i Jensen *et al.* (2013) er likevel at en videre anvendelse av ØT-tilnærmingen burde inkludere flere økosystemtjenester i vurderingen av de områdene hvor screeningen indikerer at kostnadene overstiger nytten, fordi ikke alle relevante økosystemtjenester er dekket i Aquamoney-studien. Dette er et område hvor mer arbeid er nødvendig, og trolig vil bli gjennomført de nærmeste årene.

Lokalt tilpassede virkemidler, inkludert PES, for å bedre forsyningen av økosystemtjenester

Mange eksempler og mye kunnskap kan hentes fra lokalt tilpassede og målrettede virkemidler som bidrar til oppnåelse av vanndirektivet. Noen er gjennomført i praktisk politikk, mens andre er i form av anbefalinger fra pilotstudier eller pågående forskning.

Blandede virkemidler ("mixed instruments") er mye brukt i de nordiske landene (for eksempel innen landbrukssektoren), men de fleste av disse er generelle og ikke lokalt tilpasset. Det er derfor stort potensial for mer målrettet tilpasning til lokale forhold, for eksempel for å konstruere nye eller re-etablere våtmarksområder. Eksemplene vi presenterer fokuserer på markedsbaserte virkemidler og tilnærminger som tar sikte på håndtering av forurensning fra diffuse kilder. Eksemplene hentes hovedsakelig fra landbruket, fordi problemer og eksempler herfra anses som svært relevante i en nordisk sammenheng.

Forurensing fra diffuse kilder er i praksis vanskelig å kontrollere, særlig ved bruk av uniforme virkemidler som ikke tar hensyn til forskjeller i jordens retensjonskapasitet, type gårdsbruk osv. Dette er et sammensatt problem (ofte kalt "wicked problem" i engelsk litteratur) og krever en blanding av virkemidler og tiltak som er tilpasset lokale forhold. Involvering av interessenter er også ofte både ønskelig og nødvendig. Tre eksempler på programmer på vannområdenivå fra henholdsvis Morsa i Norge, München i Tyskland og Catskill Mountains i New York State, USA, representerer programmer som ser ut til å gi betydelige og positive resultater for vannkvalitet innen relativt få år (Selv om det har vært noe diskusjon om motivasjonen for Catskill Mountains-eksemplet). Felles for programmene er bruk av lokalt tilpassede tiltak og virkemidler, noen frivillige og andre obligatoriske, samt en hensiktsmessig blanding av ulike virkemidler og aktivt engasjement fra jordeiere og husstander.

Ideén om å utvikle lokalt tilpassede PES-instrumenter på nedbørfeltnivå har også vært undersøkt i pilotprosjekter i Danmark, som så på hvordan gårdbrukere kunne inngå kontrakter med byer og tettsteder om å produsere økosystemtjenester på eiendommen som så kunne håndtere flomvann og dermed unngå oversvømmelser og overløp i de bebygde områdene. Tilnærmingen er også utgangspunkt for en foreslått regulering av utslipp av forurensende næringsstoffer (nitrogen og fosfor) i Danmark, hvor kravene differensieres ut fra lokale forhold som jordas retensjonskapasitet og effekten på økosystemtjenester i resipienten (Kjær, 2014). Våtmarks-PES-systemer, som har direkte relevans for vanndirektivet, finnes i de tre nordiske EU-landene, finansiert gjennom "Pillar II" i EUs felles landbrukspolitikk (CAP). Tiltakene og målene er mye like i alle land, men betalingsnivå og utforming av kontrakter varierer.

Kvotehandel med vannkvalitet ("water quality trading") er ikke kjent fra de nordiske landene eller i EU, men kan i prinsippet etableres som et tiltak f.eks. for å redusere forurensende utslipp på nedbørfelt-nivå. EU-kommisjonen foreslår i "A Blueprint to Safeguard Europe's water reources"[22] å utvikle såkalte "Common Implementation Strategies (CIS) Guidance" (veiledning for felles implementeringsstrategier) for slike mekanismer ("trading schemes") innen 2014. Et annet eksempel utenfor EU er nitrogenkvotehandel i området rundt innsjøen Taupo på New Zealand, som har som målsetting å opprettholde dagens gode vannkvalitet, som står i fare for å bli dårligere på grunn av intensiv jordbruksdrift og økende urbanisering i området. I følge Stanton *et al*. (2010) er det nå 66 kvotehandelssystemer knyttet til vannkvalitet i funksjon i USA, fire i Australia og ett i henholdsvis New Zealand og Canada. Frivillige avtaler ("off-sets") har vært forsøkt i Sverige, og et full-skala pilotprosjekt er nylig gjennomført i Danmark, og indikerer at blåskjellanlegg kan være et miljøvennlig og kostnadseffektivt tiltak for å redusere konsekvenser av næringsstofftilførlser til vann.

Når økonomiske virkemidler skal tilpasses vannregionområder eller vannområder, er en utfordring å finne riktig balanse mellom virkemidler og tiltak som er fornuftige lokalt, samtidig som transaksjonskostnadene knyttet til forvaltning, koordinering og kontroll, holdes på et akseptabelt nivå.

Konklusjoner

Det er flere eksempler på bruk av ØT-tilnærmingen i studier knyttet til vanndirektivet i alle de nordiske landene. Det er flest eksempler på identifisering og kartlegging, beskrivelse og kategorisering av økosystemtjenester, mens det er forholdsvis få omfattende nytte-kostnadsanalyser og vurderinger av uforholdsmessige kostnader.

Relativt få studier i de nordiske landene verdsetter økosystemtjenester som sådan, mens det er noen flere som verdsetter forbedringer i vannmiljø, inkludert oppnåelse av god økologisk tilstand, som er målsettingen i vanndirektivet. Ved siden av Aquamoney-studien beskrevet i VALUESHEDS (verdsettingsstudier i Morsa, Norge og Odense, Danmark)

[22] (COM (2012) 673).

finnes det et par nye finske studier som verdsetter forbedret vannmiljø i tråd med målsettingene i vanndirektivet på lokalt nivå. Disse verdsetter ikke økosystemtjenester direkte, men forbedringen i vannmiljø kan knyttes til ulike økosystemtjenester. Nytteoverføringer er mye brukt for å si noe om verdien av forbedret vannmiljø, og det er mange eksempler på slike studier; innen Danmark, fra Danmark og Norge til Sverige, fra én elv i Oslo til andre Oslo-elver osv. Det er imidlertid stor mangel på relevante primærstudier å overføre fra, og særlig mangel på gode primærstudier som tar utgangspunkt i vannmiljøforbedring og derfra utleder hvilke økosystemtjenester som blir påvirket i hvilken grad.

Det er flere studier, pilotprosjekter og fullskalaprosjekter som bruker målrettede og lokaltilpassede virkemidler i de nordiske landene. Disse finnes hovedsakelig innen landbrukssektoren. Den direkte sammenhengen mellom forbedrede økosystemtjenester og økonomiske mekanismer og betalingsnivå i PES (betaling for økosystemer) er imidlertid ofte indirekte. Man må ha enda bedre kunnskap om økosystemtjenester og deres verdi for å målrette disse virkemidlene ytterligere. Videre er det en økende bevissthet om at tiltak og virkemidler for redusert vannforurensing må tilpasses lokale forhold og at ØT-tilnærmingen kan være nyttig i denne sammenheng.

Det er kanskje ikke så overraskende at det tar tid å innarbeide ØT-tilnærmingen i praktisk vannforvaltning, og at de mer økonomiske delene av tilnærmingen, monetær verdsetting og nytte-kostnadsanalyse, tar lengre tid enn resten. Økosystemtjeneste-begrepet og -tilnærmingen har vært i bruk en stund, men det var ikke før TEEB-prosjektet fra 2008 og utover at fundamentet for de mer økonomiske og praktiske bruksområdene i tilnærmingen ble utviklet. Det tar tid å integrere nye tenkemåter i offentlig ressursforvaltning, men mye har skjedd, og det pågår mye arbeid på dette feltet i Norden, som eksemplene i denne rapporten illustrerer.

TemaNord 2014:561

norden

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Ecosystem Services – In Nordic Freshwater Management

Human wellbeing is dependent upon and benefit from ecosystem services which are delivered by well-functioning ecosystems. Ecosystem services can be mapped and assessed consistently within an ecosystem service framework. This project aims to explore the use and usefulness of the ecosystem service framework in freshwater management, particularly water management according to the Water Framework Directive (WFD).

There are several examples of how ecosystem services have been used in WFD related studies in all the Nordic countries. Most of them involve listing, describing and categorizing freshwater ecosystem services, while there are few comprehensive Cost Benefit Analyses and analyses of disproportionate costs that apply this framework. More knowledge about ecosystem services and the value of ecosystem services for freshwater systems is needed.

TemaNord 2014:561
ISBN 978-92-893-3851-6 (PRINT)
ISBN 978-92-893-3853-0 (PDF)
ISBN 978-92-893-3852-3 (EPUB)
ISSN 0908-6692

9 789289 338516