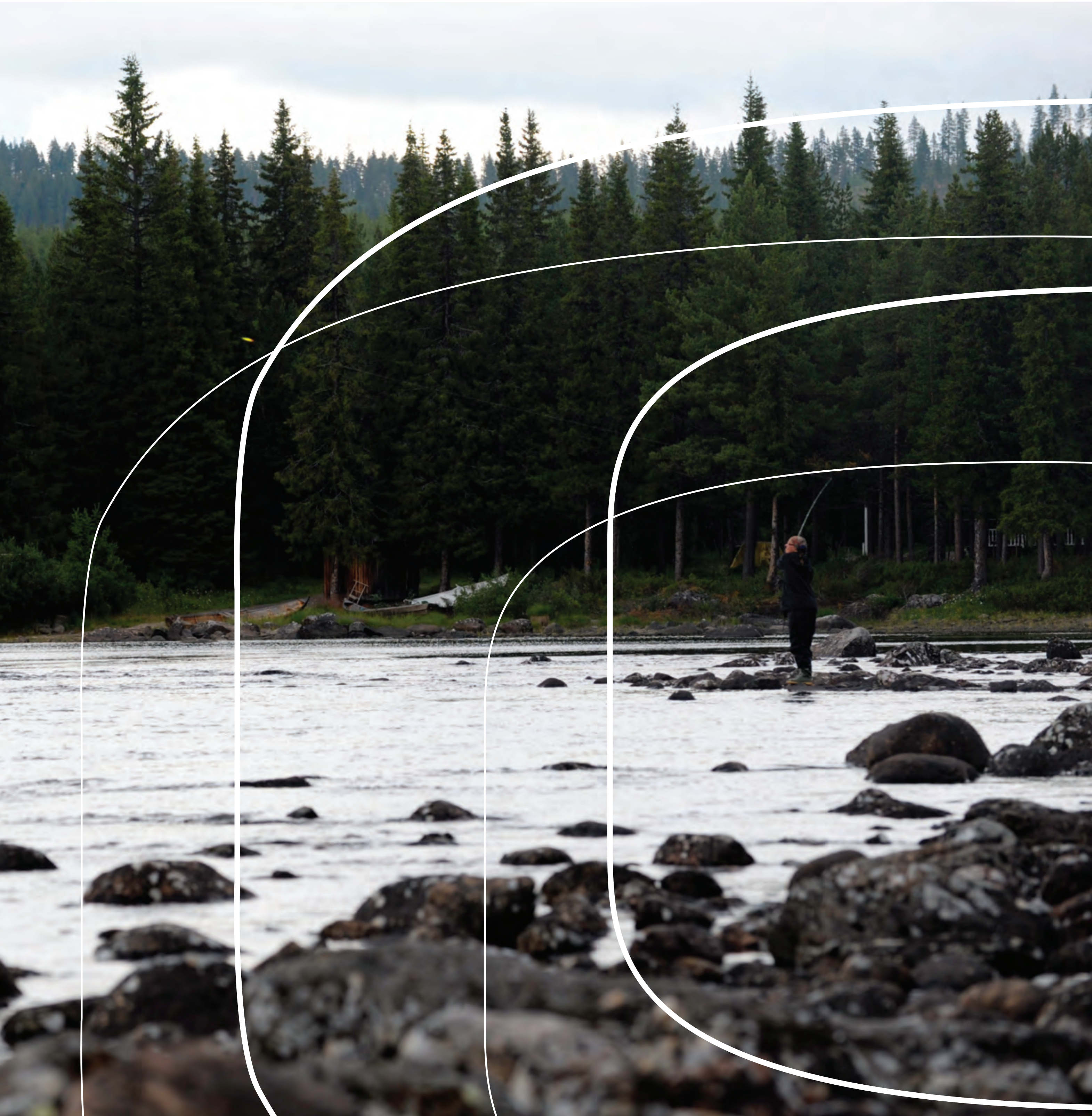
Valuation of Ecosystem services from Nordic Watersheds

– from avareness raising to policy support? (VALUESHED)



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David N. Barton, Henrik Lindhjem, NINA. Kristin Magnussen, Sweco Norge and Silje Holen, NIVA

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Foreword

David N. Barton (NINA), Henrik Lindhjem (NINA), Kristin Magnussen (Sweco Norge) and Silje Holen (NIVA) have written the report. We would like to thank participants in the Nordic valuation experts' workshop held the 21st of September 2011 in Oslo for their presentations and contribution to the discussion in the report. Nordic experts have assisted in checking the representativeness of literature reviews for each country and have in some cases also contributed text box/case study examples throughout the text:

- Marianne Zandersen, Nationalt Center for Miljø og Energi (DMU), Aarhus Uni.
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Preface

Nordic countries have in common a large number of water bodies and a myriad of catchments. The Nordic Council of Ministers commissioned The Norwegian Institute for Nature Research (NINA), The Norwegian Institute of Water Research (NIVA) and Sweco Norge to prepare a synthesis report on economic valuation of ecosystem services from watersheds in the Nordic countries.

The report demonstrates that the Nordic countries have considerable experience in valuation of provisioning services related to biomass, food and water supply, and valuation of cultural services, especially the value of recreation in water bodies and the economic consequences of eutrophication. Much less work has been done on regulating services such as water quality purification and flood reduction of watershed management. The report gives a number of examples of such valuation studies, and discusses the challenges in using the estimates for policy assessment.

The challenge, and the opportunity, for Nordic watershed management authorities is to use economic valuation of watershed services in the context of the Water Framework Directive (WFD). The family of ecosystem service concepts – provisioning, cultural, regulating and supporting services – is a framework for defining the benefits to society of improvements in ecological status of water bodies. In particular, the WFD states that derogations from attaining the objective of "good ecological status" in water bodies can only be given if costs of measures are disproportionate to benefits. From an economic point of view, disproportionality is in terms of costs exceeding benefits by a considerable margin. How much costs should exceed benefits of good ecological status is an empirical and political question that we hope will be the subject of new valuation studies in Nordic countries in the time to come.

The EUs Biodiversity Strategy to 2020[1] has as a target #2 that "*By 2020, ecosystems and their services are maintained and enhanced by establishing green infrastructure and restoring at least 15% of degraded ecosystems"*. Action 5 of the strategy is that "Member States, with the assistance of the Commission, will map and assess the state of ecosystems and their services in their national territory by 2014, assess the economic value of such services, and promote the integration of these values into accounting and reporting systems at EU and national level by

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[1] http://ec.europa.eu/environment/nature/biodiversity/comm2006/2020.htm

2020." The VALUESHED report provides access to the Nordic literature discussing how this could be done for watershed services, and some challenges for such studies to become more policy relevant. An additional ambition of the report is to serve as input to discussions in national level TEEB follow-ups in Nordic countries, with the aim of specific assessments of the feasibility and relevance of economic valuation studies to particular ecosystem services.

The analysis has been carried out during the period May – November 2011. Main sources for the work have been existing valuation studies carried out in the Nordic countries. Main authors of the report have been David N. Barton (NINA), Henrik Lindhjem (NINA), Kristin Magnussen (Sweco Norge) and Silje Holen (NIVA). Marianne Kettunen (IEEP) has provided substantial comments to the manuscript. Nordic valuation experts have assisted in checking the representativeness of literature reviews for each Nordic country and contributed examples; Marianne Zandersen and Berit Hasler, National Environmental Research Institute(NERI), Aarhus University, Denmark; Anni Huhtala, Government Institute for Economic Research, Finland; Virpi Lehtoranta, Finnish Environment Institute (SYKE), Freshwater Centre; Jens Mentzer, "Vattenmyndigheten Västerhavet, Länsstyrelsen i Västra Götalands län", Sweden.

Halldór Ásgrímsson

Secretary General
Nordic Council of Ministers

Summary

The VALUESHED report first discusses a definition of "watershed services" (Chapter 2), a review of valuation studies in Nordic countries in comparison to international experiences (Chapter 3); looks at detailed examples of valuation studies from two Nordic watersheds (Chapters 4 and 5); discusses methodological challenges and possibilities for policy application (Chapter 6), and finally makes recommendations for policy and further study (Chapter 7). The report also contains supporting material in Appendix 1 – providing further details on the stepwise approach to valuation based on the AQUAMONEY project, funded by the European Commission. Appendix 2 provides a more detailed explanation of economic valuation methods applied to watershed services for the reader unfamiliar with this literature.

Main findings

Our literature review shows that the watershed ES valued are quite similar across the Nordic countries. The services addressed are *mainly provisioning services as food and fresh water supply*, as well as cultural services as aesthetic information and opportunities for recreation and tourism. Despite some examples reviewed in this report, valuation studies of regulating and supporting/habitat services seem to be underrepresented. Our review has sought studies that address non-market values of final ES. Regulating and habitat supporting services are difficult to classify as final services for economic valuation. *Regulating services* may also be *poorly covered* in our review because we focused on aquatic systems, and to a lesser extent on impacts of land-use on ecological status of aquatic systems.

Establishing the link between *flood risk* and the condition of ecosystems in the watershed is *a complex biophysical modelling task*. The value of flood reduction services provided by upstream ecosystems is more difficult to identify the larger the watershed, the larger the storm event, and the more regulated the watershed is by man-made infrastructure (reservoirs, transfers, channeling). The value of flood damage reduction depends on a combination of preventive, avoiding and mitigation actions throughout the catchment, and in particular in the downstream areas at risk of flooding. Aggregation of value of flood reduction damages from case studies to the whole watershed were not attempted, because *site specific flood damage* modeling is required. The reliability of transferring economic damage functions is limited, in particular for buildings and agriculture. Local trade-offs and interdependencies between ES mean that they are generally non-additive for a particular wetland or catchment land-use.

A fairly large number of survey-based stated preference studies of water quality, in particular related to *eutrophication*, have been conducted in the Nordic countries. Such contingent valuation and choice experiment studies have either focused on improving bundles of goods and services through hypothetical management measures of "whole watersheds", or focused on valuing incremental changes in suitability for specific water uses, using different variations of a water quality ladder. Valuation studies looking at definitions of "*good ecological status"* under the Water Framework Directive, while designed to be directly policy relevant, are not necessarily useful for finding per hectare values for ecosystems, or for benefits transfer to other watersheds where such studies have not been conducted. *Run-off and pollution modelling* are required to assign water quality service values to land uses. Aggregation of values of water quality improvements and defining "the extent of a market" is possible with valuation studies that evaluate "*distance decay" of willingness to pay* depending on how far respondents live from water bodies. Research findings are mixed on the strength of "distance decay" for use values of water bodies. Non-use or existence values related to improvements in watershed services, which may also be substantial, are likely to be more stable across spatial scales

Based on our review we argue that valuation studies framed to address economic analysis of a particular policy such as the Water Framework Directive are responding to a different policy need than studies for example aiming at calculating average per hectare values of ecosystems. Commissioned valuation studies must start by addressing what *kinds of policy* they are aimed at informing as a function of *how reliable and accurate* the valuation method is relative to policy requirements. Beyond using valuation studies as information for framing policy debates through *raising awareness*, it should be made clear whether specific studies of valuation of ES are to be used for (i) *accounting* (e.g. in green national accounts), (ii) *priority-setting* (e.g. ranking abatement measures) or (iii) *instrument design* (e.g. payment for ecosystem service schemes). Valuation studies are required to be increasingly reliable and accurate as their purpose progresses beyond recognising and demonstrating value to capturing value in policy.

Associating values of water quality to states of the ecosystem involves combining pressure-state-impact modelling of run-off from land and water uses to status of water bodies. Watershed management in Nordic countries is seldom about how many hectares of land to allocate to "natural" ecosystems such as forests, versus agriculture. It is more about identifying and *targeting* the most *cost-effective agricultural practices and run-off mitigation measures*, and determining whether the aggregate benefits to downstream users exceed the total costs of a *programme of measures*. The focus on valuing ecosystems' contribution to human well-being, is laudable, but one must be aware of not over-focusing on *trying to isolate the value of "natural" ecosystems*. In cases where ecosystem service values cannot be identified, economic analysis still has an important role to play in decision support using cost-effectiveness analysis of ecosystem management alternatives.

Modelling of regulating services such as flood reduction and pollution control *needs to be spatially explicit* if it is to address economic interests and their locations, and in turn be policy relevant. Different interests live and use different "hectares" of an ecosystem. Average values of ES do not address income distributional issues, except at the level of differences between ecosystems – average land-users in one ecosystem can be identified as having different income levels from average land-users in another ecosystem. Average per hectare *ES values "hide" conflicts of interest* between different users using the same ecosystem and trade-offs between them. *Priority setting between alternative land-uses, projects, and measures* is at its core about identifying how land and water use values differ between interests at specific locations.

While we agree that calculation of average per hectare ES values may be useful for awareness raising and accounting at aggregate levels, we think it is not equally useful for the part of policy addressing priority-setting and instrument design.

Main research recommendations

- review the policy impact of almost three decades of non-market valuation studies related to watersheds in the Nordic countries and evaluate *the criteria for uptake of valuation estimates in policy* and differences between countries
- initiate comparative primary valuation studies to further demonstrate the use of the AQUAMONEY Guidelines for using *economic valuation under the Water Framework Directive*
- undertake *primary valuation studies across Nordic populations* that are representative at national and county/regional level – for generic hypothetical policies – for other cultural services, following the example of NMC funding for valuation of recreational fisheries (Toivonen, Appelblad et al. 2000)
- fund *site and project, policy, and measure specific valuation studies of populations within particular watersheds*, and particular regulating services in watersheds studies using production function and damage function approaches
- demonstrate possibilities and limitations in scaling available water body and watershed specific valuation studies for purposes of *ecosystem capital accounting*
- initiate valuation studies that evaluate the *spatial patterns of ES values and their dependence on distance, direction, scale and*

resolution, and implications for improvements in national accounting, priority-setting and instrument design

- sponsor *research on the design of economic instruments in the policy mix*[2] *of WFD "programmes of measures" for watershed management* (such as payments of ES) – assessing their ecological effectiveness, benefits of derived ES, technical and transaction costs of implementation, distributional impacts and legitimacy, institutional and political barriers and opportunities for implementation
- support the development of Nordic *visualizations and illustrations of ecosystem services* and in Nordic languages to help promote public awareness, as an alternative to economic valuation
- promote similar reviews to VALUESHED of specific other ecosystems in Nordic countries (e.g. forests, coastal wetlands and open sea ecosystems), addressing interdependencies of valuation estimates between ecosystems (e.g. off-site ES of forests), as a complement and follow-up to Nordic TEEB

[2] http://policymix.nina.no

1. Introduction

The VALUESHED study was commissioned by the Nordic Council of Ministers early 2011, started in May and finished in November. It responds to a need for a Nordic synthesis of experiences with valuation of ecosystem services from watersheds, in light of the recent study "The Economics of Ecosystems and Biodiversity" (TEEB) (Kumar 2011)[3] and the Millennium Ecosystem Assessment (MEA) before that.[4] There is a need for Nordic references and cases of the value of ecosystem services in watersheds, to complement references compiled by MEA and TEEB valuation study databases. This report attempts to fill this gap.

1.1 Objectives

1.1.1 Objective

The objective of the report is to estimate the scope of economic values of ecosystem services (ES) in selected watersheds in at least two Nordic countries as decision-support for specific policy scenarios and for general demonstration of the importance of such services.

1.1.2 Sub-objectives

- Give an overview of which ES are received from watersheds.
- Describe, assess and – to the extent possible – give estimates of the values of the ecosystem services identified in selected Nordic watersheds
- Discuss and – to the extent possible – aggregate the value estimates for the identified ES
- Consider issues of distribution (who receives and delivers ES) and potential payment for ES mechanisms applicable for watersheds (i.e. so-called PES schemes)
- Cooperate with the Nordic TEEB study in providing input to their review on watershed related ES

[3] http://www.teebweb.org/
[4] http://www.maweb.org/en/index.aspx

1.1.3 Scope

VALUESHED can be seen as a preliminary example of a focused and methodological assessment on ES of particular economic importance for Nordic countries. However, VALUESHED is not an example of an in-depth assessment of the value of nature from Nordic watersheds. VALUESHED has had a budget of an equivalent of roughly 50 person days which has been sufficient to compile an overview of valuation work done in Nordic countries for watersheds and provide some detailed examples based on existing studies, as a basis for a discussion of methodology, a discussion of policy and recommendations for further study.

1.2 Policy relevance

The authors hope VALUESHED may be useful to Nordic watershed authorities discussing how to conduct valuation in the context of the Water Framework Directive or its national equivalents. In particular, the WFD art. 4 states that derogations from attaining the objective of "good ecological status" in water bodies can only be given if costs of measures are disproportionate. From an economic point of view, disproportionality is in terms of costs exceeding benefits by a considerable margin. How much costs should exceed benefits of good ecological status is an empirical and political question. However, the family of ES concepts – provisioning, cultural, regulating and supporting services – is a framework for defining the benefits to society of improvements in ecological status.

The VALUESHED report provides access to the literature showing how this can be done and some challenges for such studies to become more policy relevant.

An additional ambition of the report is to serve as input to discussions in national level TEEB follow-ups in Nordic countries, with the aim of specific assessments of the feasibility and relevance of economic valuation studies to particular ES and ecosystems. The EUs Biodiversity Strategy to 2020[5] has as target #2 that "*By 2020, ecosystems and their services are maintained and enhanced by establishing green infrastructure and restoring at least 15% of degraded ecosystems*". Action 5 of the strategy is that "Member States, with the assistance of the Commission, will map and assess the state of ecosystems and their services in their national territory by 2014, assess the economic value of such services, and promote the integration of these values into accounting and reporting systems at EU

[5] http://ec.europa.eu/environment/nature/biodiversity/comm2006/2020.htm

and national level by 2020." The EUs "Roadmap for resource efficiency – an integrated approach",[6] also devotes a chapter to ES.

1.3 Report overview

The report moves from a review of valuation studies in Nordic countries in comparison to international experiences, looks at detailed examples of valuation studies from two Nordic watersheds, leading to a discussion of methodological challenges and possibilities for policy application, and finally recommendations for policy and further study.

Chapter 2 briefly provides a definition of our shorthand for "watershed services," a justification for choosing to focus on "flood reduction" and "water pollution reduction" services of ecosystems in watersheds, followed by a brief overview of economic valuation methods drawn from the TEEB report, and finally an overview of the stepwise approach we use to discuss our detailed examples for Nordic watersheds. Chapter 2 links to supporting material in Appendix 1 – providing further details on a stepwise approach to valuation based on the AQUAMONEY project.[7] Appendix 2 provides a more detailed explanation of economic valuation methods applied to watershed services for the reader unfamiliar with this literature.

Chapter 3 provides an introduction to selected international reviews of valuation studies, which serves as a backdrop to our review of the availability of valuation studies on watershed services in the Nordic countries. This chapter provides the background for identifying gaps in the type of ES studied and the methods used. It is also our "data" supporting recommendations for further research on particular ES using particular methods.

Chapters 4 and 5 take a different approach, diving into some detail from valuation studies selected in two Nordic watersheds, the Glomma-Lågen Water Region and the Odense River Basin, respectively. The Glomma-Lågen Water Region is a WFD construct encompassing both the Glomma-Lågen River Basin proper, as well as the neighbouring Morsa and Halden catchments. We look at approaches to valuing reduction in flood damages in Glomma-Lågen River Basin, improvement in the ecological status of lake water across the three catchments of the Glomma-Lågen Water Region, and improvement in the ecological status of river

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[6]http://europa.eu/rapid/pressReleasesAction.do?reference=IP/11/1046&format=HTML&aged=0&language=EN&guiLanguage=en

[7] AquaMoney, funded by DG RTD, brought together 16 leading European research institutions to develop and test practical guidelines for the assessment of environmental and resource costs and benefits (ERCB) in the European Water Framework Directive (WFD).

water quality in the Odense River Basin. A detailed look at valuation studies at the river basin and local within-river basin scale serve as a source of discussion of methodological challenges and solutions for valuation methods applied to different watershed services across different geographical scales.

Chapter 6 discusses the data gaps uncovered in the review of Nordic valuation studies, followed by a discussion of different challenges to valuation methodology in the context of ES, challenges to increase the policy relevance of ES valuation, and to dissemination of the concept of "ecosystem services" in public debate. This discussion provides additional support for our recommendations for further research.

In Chapter 7 we focus our conclusions on general recommendations to policy makers in using (or not using) valuation results in different contexts, data gaps and recommendations for further research, and some recommendations for national TEEB follow-ups in Nordic countries, based on our material from the literature review and case studies.

Appendices 1 and 2 provide supporting material for the reader interested in some further detail on valuation methodology.

2. Approach

Chapter 2 briefly provides a definition of our shorthand for "watershed services," a justification for choosing to emphasise "flood reduction" and "water pollution reduction" services of ecosys-tems in watersheds, followed by a brief overview of economic valuation methods drawn from the TEEB report, and finally an overview of the stepwise approach we use to discuss our detailed examples for Nordic watersheds. Chapter 2 links to supporting material in Appendix 1 – providing further details on a stepwise approach to valuation based on the AQUAMONEY project. Appendix 2 provides a more detailed explanation of economic valuation methods applied to water-shed services for readers unfamiliar with this literature.

2.1 Definition of "watershed services" – ecosystem services from watersheds

The term "watershed" is commonly used to refer to an area; specifically, the area in which all surface waters flow to a common point. A great deal of confusion and misunderstanding is created by the inconsistent use of terms to describe the relative size of watersheds-basin, watershed, drainage, catchment.[8] Other terms that are used to describe a watershed are drainage basin, catchment, catchment area, catchment basin, drainage area, river basin, and water basin.

Most of the world's aquatic and terrestrial biomes are distributed across watersheds. In order to keep a direct reference to the TEEB study we will use their classification of biomes and ES. TEEBs biomes of relevance to our review of studies from Nordic countries include:

- Inland wetlands (we do not look at coastal wetlands)
- Lakes and rivers
- Forests (temperate and woodlands)
- Grasslands
- Polar and high mountain systems

[8] See e.g. http://www.watershed.org

Box 1. Ramsar Convention definition of wetlands

"For the purpose of this Convention wetlands are areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water the depth of which at low tide does not exceed six metres."

Source: http://www.ramsar.org/

TEEB's definition of wetlands includes freshwater floodplains, swamps/marshes and peat lands, but not lakes and rivers. The TEEB definition of wetlands is not as comprehensive as that of the Ramsar Convention (Box 1).

Kumar's (2010) overview of ecosystem services due to major biomes can be further classified into different final ecosystem services, as well as their relationship to specific ecosystem functions. It is beyond the scope of this report to do this for all the biomes listed above. As an example, guidelines prepared for the AQUAMONEY research project list the final goods and services of freshwater aquatic ecosystems ("outcomes)," and the ecosystem functions they relate to (Brouwer, Barton et al. 2009) An important message from the matrix of ecosystem functions and goods/services is that there are multiple correlations between functions and final services. For example, water recharge and discharge are part of the hydrological balance of a watershed; water as a good is rival in consumption between end-users; services derived from e.g. water discharge such as navigation and hydropower are also competing uses. The overview is thus a list of "potential" ecosystem services in watersheds, before site-specific conditions and uses have been considered. In Chapter 6 we discuss the difference between intermediate and final ecosystem services, benefits and values further.

In order to see how valuation is conducted in site specific conditions, we have focused our examples of valuation of watershed services in selected Nordic watersheds on two regulating services as defined by the TEEB study by Kumar (2011):

- Moderation of extreme events, and specifically moderation of floods. We call this *"flood reduction"* in the shorthand of this report
- Waste treatment / water purification, and specifically water pollution. We call this *"pollution reduction"* in the shorthand of this report

Two other ES categories reviewed in Kumar (2011), which are related are:

- Regulation of water flows
- Erosion prevention

ES as classified by TEEB above are made more concrete by specifying the types of land- or water use pressures that are the subject of policy scenario analysis and modelling. For example:

- Moderation of extreme events is more specifically called “land cover impact on storm peak mitigation”(Ennaanay, Conte et al. 2011), or
- Water purification and erosion control are more specifically called “retention of nutrients and sediment by vegetation” (Conte, Ennaanay et al. 2011)

Also, land cover promoting storm peak mitigation, more generally often promotes “regulation of water flows” (both peak and low flows). An accurate description of the ES being valued is essential. Its neglect can lead to very lengthy discussions between economists and natural scientists about the policy relevance and boundaries of a valuation study. We return to these challenges in the detailed case studies and the discussion sections of the report.

Table 1. Aquatic system functions and their potential outcomes in terms of goods and services

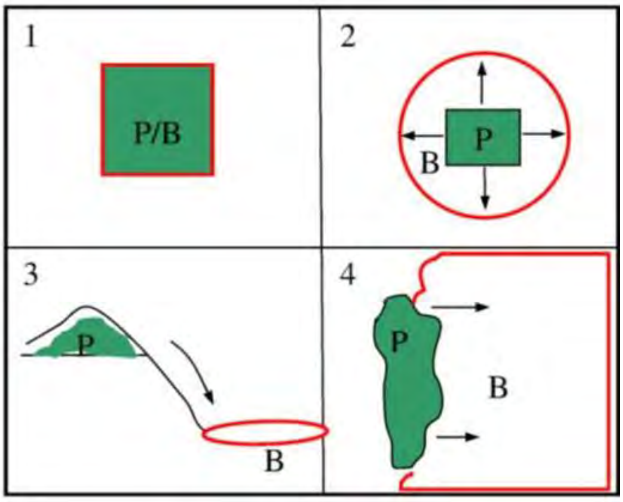
	Goods								Services																			
	Outcomes																											
Function	**Potable water for household use**	**Water for landscape maintenance and peat soil**	**Water for crop irrigation**	**Water for livestock consumption**	**Water for food processing**	**Water for other manufacturing processes**	**Cooling water for power plants**	**Water transport**	**Prevention of saline intrusion**	**Water/soil support for prevention of land subsidence**	**Natural erosion, flood and storm protection**	**Shoreline stabilization**	**Sediment removal**	**Transport, treatment, medium for wastes and other by-products of human activities**	**Improved air quality through the support of living organisms**	**Biological diversity provision**	**Recreational swimming, boating, fishing hunting, trapping and plant gathering**	**Commercial fishing, hunting, trapping and plant gathering**	**Energy production**	**On and off site observation and study for leisure, education and scientific purposes.**	**Micro-climate regulation**	**Macro-climate regulation**	**Toxin removal**	**Toxin export**	**Cultural value provision**	**Historical value provision**	**Aesthetic value provision**	**Wilderness value provision**
Water discharge	■	■	■	■	■	■	■	■						■	■	■	■	■		■	■			■				
Water recharge	■	■	■	■	■	■	■		■	■					■	■	■			■	■							
Flood mitigation										■	■					■				■								
Sediment retention	■							■		■	■	■	■		■	■	■	■		■			■					
Nutrient retention														■	■	■	■	■		■							■	
Nutrient export														■	■	■	■			■								
Trace element storage																■	■	■		■			■					
Trace element export																■				■				■				
Carbon sequestration																■	■		■	■		■						
Biodiversity maintenance															■	■	■	■	■	■	■						■	■
Culture/heritage																	■			■					■	■	■	

Source: AQUAMONEY Guidelines (Brouwer, Barton et al. 2009) at www.aquamoney.org

2.2 Spatial characteristics of watershed services

ES can be classified according to whether benefits (B) are on-site or off-site relative to the ecosystem location that provides (P) them, and their spatial direction (Fisher, Turner et al. 2009)(Figure 1). "Watershed services" regulating run-off and affecting the hydrological cycle are upstream-downstream services (type 3, Figure 1), for example flood reduction, erosion reduction and water flow regulation. Water pollution reduction can be provided by vegetation in the catchment (type 3) and limnological processes on-site in the water bodies themselves (type 1).

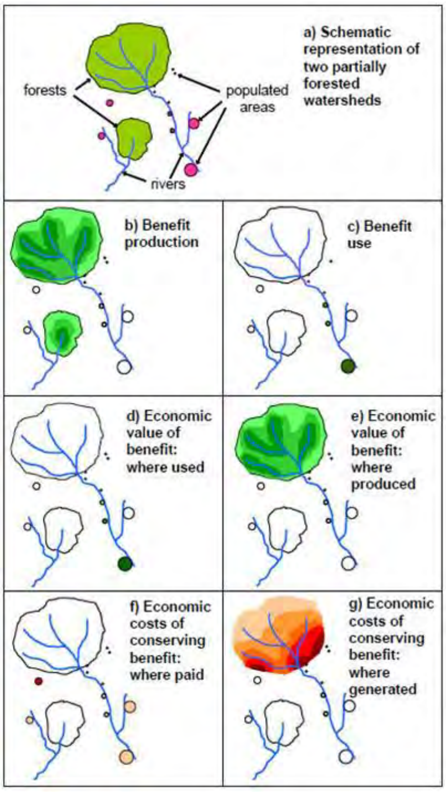
Figure 1: Spatial directionality of eco-system services relative to location of eco-system.



Source: Fisher et al. 2009.

Benefits of watershed services such as potential flood protection, potential reduction of e.g. sediment and nutrient loads in run-off and resulting water quality are enjoyed by water users at different points in the downstream (Balmford et al. 2008) (Figure 2).

Figure 2: Heterogenous spatial distribution of costs of providing and the benefits derived from watershed services.



Source: Balmford et al. 2008

The potential value of the service is produced in the upstream e.g. by landowners managing vegetation cover such as forests (b). In general, costs of managing the service providing ecosystem are incurred on-site by land users of the ecosystem in question (g). Associating the downstream value to the user (d) of specific land uses upstream is required in order to assign watershed service values to upstream land uses (e). Being able to quantify these costs and benefits of watershed services with sufficient accuracy and reliability to compare them, has policy implications. If the value of provision (e) exceeds the economic costs of conservation (g), some form of collective action or policy that recognizes these net benefits to the downstream is socially optimal. Payments for ecosystem services (f), as a minimum to compensate land users' opportunity costs, or as a maximum to pay for the benefits, have been widely reported in the literature. Most PES schemes recognise some *compensation* measure of opportunity costs, while very few examples exist of actual *payment* for benefits (Porras, Grieg-Gran et al. 2008).

2.3 Selection of watershed services

Why were flood regulation and water purification chosen as examples in the Valueshed report?

A meta-analysis by Brander et al. (2010) for the European Environment Agency points out that "flood control and storm buffering," as well as "water quality improvement" are ES characteristics of wetlands that elicit significantly higher WTP than the average across all valuation studies included in their review[9] (Table 2). The values of these regulating services were higher than in studies looking at provisioning services such as hunting and fuel wood gathering. Studies that focused on revealed preference methods (where people's preferences and resulting valuation is derived from market behaviour – see next section and Appendix 2, reported lower per hectare willingness to pay values than stated preferences studies such as contingent valuation and choice experiments (where preferences are derived from Regulating services of "flood control and storm buffering" and "water quality improvement" have positive and significant coefficients – willingness to pay per hectare wetland in studies of these services is systematically higher relative to other ES whose coefficients are not significantly different from zero or negative. The meta-analysis is based on 383 value observations from 166 independent valuation studies (Ghermandi, Bergh et al. 2007) questionnaire surveys).[10]

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[9] It is not clear in Brander et al.(2010) what ecosystem service category they use as a basis for comparison in their meta-regression (i.e. omit from the analysis to avoid multicollienarity in dummy coding of variables).
[10] Again it is not clear which category of valuation methods is omitted in the dummy coding

Table 2. Meta-analysis of wetlands ecosystem service values

	Variable	Coefficient	P-value
	(constant)	-3.078	0.187
Study variables	Contingent valuation methods	0.065	0.919
	Hedonic pricing	-3.286***	0.006
	Travel cost method	-0.974	0.112
	Replacement cost	-0.766	0.212
	Net factor income	-0.215	0.706
	Production function	-0.443	0.523
	Market prices	-0,521	0.317
	Opportunity cost	-1.889**	0.035
	Choice experiment	0.452	0.635
	Marginal	1.195***	0.008
Wetland variables	Inland marshes	0.114	0.830
	Peatbogs	-1.356**	0.014
	Salt marshes	0.143	0.778
	Intertidal mudflats	0.110	0.821
	Westland size	-0.297***	0.000
	Flood control and storm buffering	1.102**	0.017
	Surface and groundwater supply	0.009	0.984
	Water quality improvement	0.893*	0.064
	Commercial fishing and hunting	-0.040	0.915
	Recreational hunting	-1.289***	0.004
	Recreational fishing	-0.288	0.497
	Harvesting of natural materials	-0.554	0.165
	Fuel wood	-1.409**	0.029
	Non-consumptive recreation	0.340	0.420
	Amenity and aesthetics	0.752	0.136
	Biodiversity	0.917*	0.053
Context variables	GDP per capita	0.468***	0.001
	Porpulation in 50 km radius	0.579***	0.000
	Wetland area in 50 km radius	-0.023	0.583

OLS results. R^2 = 0.49; *Adj*. R^2 = 0.43. Significance is indicated with ***, **, and * for 1, 5, and 10% statistical significance levels respectively.

Source: Brander et al. 2010 (table 4.3: results obtained with meta-regression model of wetland values)

The TEEB study (Kumar 2011) collected over 1300 original values from 160 valuation studies in an Ecosystem Services and Valuation Database (ESVD) (after a screening of many hundreds from a number of databases.[11] In Table 3 we have extracted information for the biomes addressed in TEEB relevant for Nordic countries and watershed services. There were no observations for polar and high mountain systems with respect to the flood and pollution reduction services we chose to focus on in this review.[12]

[11] See http://www.fsd.nl/esp/77395/5/0/30

[12] Freshwater storage in ice caps is mentioned in the TEEB report, but there are no valuation references.

Table 3. Economic value of biomes in watersheds based on TEEB review

Biomes	Ecosystem services	#estimates	Minimum ($/ha/yr)	Mean ($/ha/yr)	Maximum ($/ha/yr)
Inland wetlands	Moderation of extreme events (i.a. floods)	7	237	1569	4430
	Regulation of water flows	4	14	4660	9369
	Waste treatment / water purification	9	40	1356	4280
	Erosion prevention	1			
Rivers and lakes	Moderation of extreme events (i.a. floods)	NA			
	Regulation of water flows	0			
	Waste treatment / water purification	2	305	2642	4978
	Erosion prevention	NA			
Temperate forests & woodlands	Moderation of extreme events (i.a. floods)	1			
	Regulation of water flows	2	0	2	3
	Waste treatment / water purification	4	130	0	701
	Erosion prevention	2*			
Grass-lands	Moderation of extreme events (i.a. floods)	NA			
	Regulation of water flows	NA			
	Waste treatment / water purification	3	13	170	358
	Erosion prevention	2	38	43	47
Tropical forest (for comparison)	Moderation of extreme events (i.a. floods)	4	8	92	340
	Regulation of water flows	4	2	19	36
	Waste treatment / water purification	6	0	216	665
	Erosion prevention	11	562	11	3211

Source: ecosystem service and valuation database (ESVD). * ESDV does not provide valuation estimates for ecosystem services with only one observation per ecosystem service per biome – temperate forest and woodlands had one observation each. Values are per household per hectare per year in inflation and purchasing power adjusted 2007 US$. NA – not applicable to the biome.

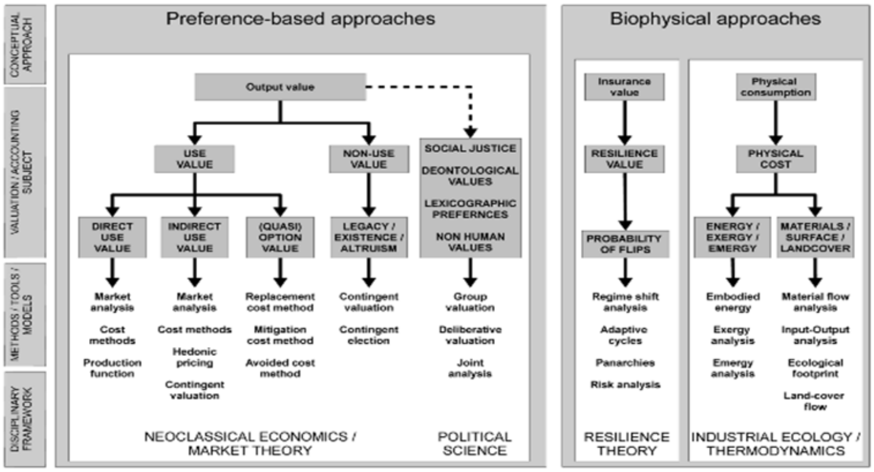
For all biomes there are only a handful of valuation studies representing "moderation of extreme events and "waste treatment /water purification" and related services of regulation of water flows and erosion prevention.

In summary, justification for focusing our detailed case study examples (Chapters 4 and 5) on the aforementioned regulating services is (i) their potential value as indicated in meta-analysis, and (ii) the need to explore in more detail why there are so few valuation studies available as characterised by the TEEB study's ecosystem service and valuation database (ESVD).

2.4 Valuation methods reviewed

Our review of Nordic valuation studies of watershed services group methods in five large categories of methods; stated preference, revealed preference, production/damage function, cost-based, and benefits transfer.[13] Given that an important reference point for VALUESHED's review is the TEEB study (Kumar 2010) we can briefly relate the categories of our review to valuation methods reviewed in TEEB (Figure 3). VALUESHED has reviewed methods based on neoclassical economics/market theory. VALUESHED has not reviewed the use of "biophysical approaches" to valuation, nor methods from political science addressing alternative concepts of value. The interested reader is referred to a more detailed description of neoclassical/market theory based valuation methods in Appendix 1.

Figure 3: Valuation methods referred to in the TEEB report.



Source: (Kumar et al. 2010).

Neoclassical economics / market theory based methods included in this review:

Stated preference methods – willingness to pay/or to accept compensation for changes in provision of ES are "stated" by respondents in surveys using structured questionnaires. Well known methods include contingent valuation and choice experiments.

Revealed preference methods – values are "revealed" through studying consumers' choices and the resulting price changes in actual markets,

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13 Different authors use different classification schemes, but these categories can be found in almost all text books.

that can then be associated with changes in provision of ES. A well known method is hedonic pricing of property characteristics, i.e. where the impact of environmental quality attributes on prices of properties is distinguished from other factors that affect prices. Travel cost methods used to value recreational benefits of ecosystems are often also included in this category.

Production/damage function – a group of methods used to value regulating and supporting services, where ES are one of several "inputs" to a final service or good enjoyed by people. Ecosystems' marginal contribution to the final service is valued. When a change of ecosystem characteristics leads to off-site or downstream loss of services, biophysical damage functions of this "pressure-state-impact" relationship are used.

Cost-based methods – assume that expenditures involved in preventing, avoiding or mitigating losses of ES represent a minimum value estimate of what people are willing to pay for the ES.

Benefits / value transfer – refer to the use of secondary, existing study estimates, from any of the valuation methods mentioned above. In the study by Brander et al, mentioned above, meta-analysis techniques were used to distil value information from a broad literature for use in benefit transfer.

2.5 Stepwise conceptual approach to valuation of watershed services

In chapters 4 and 5 we discuss selected valuation studies from two Nordic catchments and for the two regulating services in greater detail. We structure the discussion of these case studies according to a stepwise approach to conducting valuation studies (Brouwer, Barton et al. 2009):

- *Step 1:* Policy scenarios as basis for valuation
- *Step 2:* Definition of policy measures
- *Step 3:* Identification of environmental change
- *Step 4:* Identification of goods and services
- *Step 5:* Identification of beneficiaries
- *Step 6:* Identification of economic values
- *Step 7:* Value elicitation / demonstration
- *Step 8:* Value aggregation – demonstrating value
- *Step 9.1:* Validation of valuation assumptions and estimates
- *Step 9.2:* Evaluation of demonstration and policy relevance

For further details regarding the economic valuation steps see Appendix 2.

3. Overview of Nordic and inter national studies of watershed services

This chapter first gives a brief overview of a few central international watershed valuation studies, with emphasis on studies that have reviewed the literature.

Second, we provide an overview of the watershed valuation literature in the Nordic countries. The aim is to give a sense of which types of services have been valued, which methods have been used and where the main gaps and challenges are. The aim is however not to be exhaustive, but to communicate our impressions based on a quick review of a selection of recent Nordic studies. Several short case examples are included from the Nordic countries. These supplement the in-depth cases from Denmark and Norway in the next chapter. In line with the rest of the report, our main emphasis is on flood protection and natural water purification, though studies of the former seem to be rare.

3.1 Some key international valuation studies

The economic valuation literature developed from the 1960s onwards when the regulation of classic air and water pollution problems was the most pressing. It is therefore no surprise that the longest valuation tradition internationally in the context of watershed services is related to (abatement technology-focused) water purification and water quality. Mitchell and Carson (1989), the modern fathers of the contingent valuation method, for example, introduced a "water quality ladder" that they used to explain to respondents how physical indicators of water quality relate to suitability for different recreational and other uses of the water. This ladder has since then been used in many shapes and forms in a large number of valuation studies, including to inform second and third waves of regulation, for example the implementation of EU's Water Framework Directive. The AQUAMONEY research project, for example, has built on this valuation tradition (using another stated preference technique; choice experiments) and has been important in providing guidance to policy makers regarding the quantification of environmental resource costs and benefits of the WFD in different countries.[14]

Most of the water quality valuation studies do not aim to value nature's ecosystem contribution to water purification and water quality. Rather they define policy measures such as reducing run-off from agriculture, end-of-pipe abatement or water treatment that in turn lead to water quality improvements that yield benefits that are valued through different methods (see e.g. meta-analytic reviews of the water quality literature by Johnston, Besedin et al. 2003; Johnston, Besedin et al. 2005; Van Houtven, Powers et al. 2007). In addition to continued valuation of water quality in various contexts, valuation methods have increasingly been used to demonstrate values of a range of watershed-related services. Balmford, Rodrigues et al. (2008) provide a recent review of ES, for example related to fresh water provision, regulation and purification.

Few if any of the international valuation studies related to watershed services, have framed their studies in the ES terminology of MEA and TEEB, but have rather valued (unspecified) bundles of goods and services from specific watersheds or water bodies, rather than individually specified services. This makes it a challenge to classify such studies with the ES framework, which we try to do for Nordic studies below.

A popular subject of study, beyond water quality valuation, has been the benefits of wetlands (see e.g. Brander, Florax et al. 2006; Brander 2010). The most recent and comprehensive compilation of wetland valuation studies, we are aware of, is contained in Kumar (2010), a central TEEB publication (see Table 3 above). In Appendix 2 of this book studies have been categorised along dimensions of valuation methods (stated and revealed preference, production and costs based methods, and benefit transfer) and the four main types of ES (provisioning, regulating, habitat/support and cultural). There are studies covering most wetland ES, though the majority are centred around provisioning services using production based methods and cultural services using stated preference methods. There is a limited number of studies on the regulation of water flows and flood protection and natural water purification, most of which use cost based valuation methods (see one example from Germany in Textbox 2). A high profile case is the water purification costs saved by New York City through measures in the Catskills watershed (though this case is also contested).[15] The gaps in the literature are even larger in the area of habitat/support services, according to the Appendix compiled by Kumar (2010).

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[14] See for example Bateman et al. (2011) for an investigation into how water quality values obtained through contingent valuation may be transferred from studied sites to unstudied sites, where such information may be useful for policy-makers (i.e. benefit transfer).
[15] See http://www.perc.org/articles/article547.php

Box 2. Economic calculation of dike relocation at the German Elbe. An ecosystem services perspective

The paper focuses on the morphological changes of rivers in Germany which have been changed considerably by diking over the last two hundred years. The resulting changes in water quality have not been dealt with systematically in the German Program of Measures of the Water Framework Directive, mostly due to cost considerations, with the exception of fish passability. The relocation of dikes constitutes another option to improve the morphological quality of river water bodies. In the past and in flood management practice, the construction of artificial storage with the inclusion of polders have been considered more effective from a flood control perspective without taking into account the additional benefits of dike relocation in comparison. Thus, the paper presents a cost-benefit analysis of a program of dike relocation at the German part of the Elbe in comparison to an equivalent program of polder construction. The included benefits cover three types of ecosystem services: Changes in flood protection (based on avoided property damages), changes of the biodiversity of the wetlands (based on contingent valuation) and the nutrient retention of the additional wetlands (based on replacement costs). The benefits from these changes of ecosystem services as a result of the program are then compared to the cost of both alternative programs. The comparison shows that the dike relocation program is economically advantageous to the polder program if one includes the two additional ecosystem services.

Source: Grossmann, M., Hartje, V., Meyerhoff, J. (2010): Ökonomische Bewertung naturverträglicher Hochwasservorsorge an der Elbe. Naturschutz und Biologische Vielfalt 89, Bundesamt für Naturschutz: Bonn.

Many of the primary valuation studies of watershed services have been conducted to demonstrate economic values in different contexts, and not specifically to inform policy (capture value). However, this may be changing, for example in the context of the EU Water Framework Directive. More indirectly, valuation studies through recognizing and demonstrating values have spurred a number of payment for ecosystem services (PES) schemes around the world. Worth noting here is the review of such PES schemes specifically for watershed services (Porras, Grieg-Gran et al. 2008).[16] While the literature on watershed service valuation is expanding, there is an increasing emphasis on moving from demonstration of values to ways of capturing values.[17]

As we shall see in the next section, Nordic watershed valuation studies mimic to some extent the ecosystem service coverage and valuation

[16] See also http://www.watershedmarkets.org/.

[17] See e.g. the InVest software developed under the Natural Capital Project, which aims to provide tools for decision-makers for integrated valuation of ecosystem services and trade-offs. The software package has modules on e.g. water purification. See http://www.naturalcapitalproject.org/InVEST.html

methods described for the international literature by Kumar (2010), although there is some variation between the Nordic countries.

3.2 Overview of Nordic valuation studies and some examples

3.2.1 Introduction

We have conducted a quick review of studies in the Nordic countries valuing ES from watersheds. Our review includes studies valuing ES of wetlands and water quality-related benefits, but does not include valuation of e.g. forests and land uses unless studies refer explicitly to the regulating function of land uses for runoff. The studies covered are drawn from web searches, valuation databases, especially the Nordic Environmental Valuation Database,[18] key studies and knowledge within the project team as well as from the Nordic environmental economics reference group. We provide examples of studies and give references to those. Complete lists of references of valuation studies is not compiled for this chapter, but more extensive reference lists can be found in many of the studies we refer to. However, most valuation studies do not use the ES approach, but are included because of their relevance in this context.

A study of the value of recreational fisheries was carried out in national samples from all Nordic countries (Toivonen, Appelblad et al. 2000).

3.3 Norway

Many of the Norwegian valuation studies are centered around eutrophication and the value of cultural services such as recreation and aesthetic values. The majority has been carried out in southeastern Norway where eutrophication is one of the largest environmental pressures related to water (Magnussen 1992, Magnussen et al. 1995, Magnussen and Bergland 1996, Magnussen et al. 1997, Barton et al. 2008), with a couple of examples from western and central Norway (Magnussen et al. 1996, 1997). Few of the eutrophication studies are related to habitat services, but some cover supporting services such as e.g. biodiversity. Several studies also value the impact of environmental change on recreational fishing, in particular in the context of acidification and hydropower de-

[18] http://www.norden.org/en/publications/publications/2007–518

velopment (Mørkved and Krokan 2000; Navrud 2001; Navrud 2001), including benefits transfer (Finstad, Barton et al. 2007).

There are however exceptions such as Barton and Navrud et al. (2010) and Magnussen and Bergland (1996) studying the economic benefits of large scale remediation of contaminated marine sediments in the Grenland fjords in Norway. Both these studies performed a contingent valuation (CV) survey of a representative sample of households from municipalities adjacent to these fjords. The CV method aimed at valuing the benefits perceived by households of removing dietary health advisories on seafood consumption currently in place around the fjords.

There are also studies of the regulation of water flows. Examples of this are cost-benefit analysis of flood protection (Barton and Dervo 2009), economic risk analysis of flooding (Sælthun et al. 2000) and Multi Criteria Analysis (MCA) used to determine environmental water flow in regulated rivers based on stakeholder participation (Barton, Berge et al. 2010). Barton et al used the method to evaluate trade-offs between hydropower generation income, wetland habitat quality indicators, and other wetland user interests and the study is an example of a deliberative or participatory method trying to understand people's preferences and the process of decision-making. There are also examples of studies using benefit transfer related to the value of water purification, biodiversity and recreation.

Many of the Norwegian studies are based on stated preference methodology; contingent valuation and choice experiments, in addition to market prices and replacement costs. In addition to this, there are some interesting examples of the use of production based methods as described above. Studies such as Barton et al. (2008) integrating models of phosphorus (P) abatement costs and effects, as well as models of lake P and eutrophication dynamics are important to explore and evaluate the probable outcomes and uncertainties of the eutrophication problem and the cost-effectiveness analysis of the corresponding abatement measures.

Some examples of valuation studies of different watershed services carried out in Norway are presented in the textboxes below.

Barton and Dervo (2009) demonstrate the use of flood damage functions in the context of benefit- cost analysis and multiple criteria analysis of flood protection measures (Box 3)

Box 3: Benefit cost analysis of flood protection. A methodology assessment with an example from Skarvvollene

Barton and Dervo (2009) demonstrate the use of flood damage functions in the context of benefit- cost analysis and multiple criteria analysis of flood protection measures. To demonstrate the approaches, they used available data from the Skarvvollene flood protection works on river flats along the Lågen River in Ringebu Municipality. The report also evaluates the current benefit-cost analysis guidelines for flood protection works used by the Norwegian Water Resources and Energy Directorate in the context of the EU Water Framework Directive (WFD). Barton and Dervo (2009) argue that multiple criteria analysis (MCA) can be employed as a complement to benefit-cost analysis in the assessment of "disproportionate costs" under the WFD. They state that MCA is particularly useful in evaluating trade-offs between priced and non-priced hydro-morphological impacts of flood mitigation projects. It is also a framework for documenting both expert and local opinion on non-riced impacts and their relative values.

Barton et al. (2008) used Bayesian belief network methodology to integrate models of phosphorous abatement costs and effects as well as models of lake phosphorous and eutrophication dynamics (Box 4). There seems to be a knowledge gap in the areas of valuing regulating services, such as e.g. eutrophication mitigation, and interdisciplinary projects integrating different models is therefore of high importance:

Box 4. EUTROBAYES – Integration of nutrient loading and lake eutrophication models in cost-effectiveness analysis of abatement measures

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

Table 4. Overview of valuation studies of different watershed services carried out in Norway

Ecosystem services	Valuation methods					
	Stated preferences	**Revealed preferences**	**Production /damage function**	**Cost based**	**Benefit transfer**	**Total***
Provisioning services						
1 Food & food safety	X	X			X	>5
2 (Fresh)Water supply	X					<5
3 Raw materials						
4 Genetic resources						
5 Medicinal resources						
6 Ornamental resources						
Regulating services						
7 Influence on air quality						
8 Climate regulation						
9 Moderation of extreme events			X	X		
10 Regulation of water flows	X	X	X			>5
11 Waste treatment/ water purification	X		X	X	X	>5
12 Erosion prevention			X	X		<5
13 Nutrient cycling and maintenance of soil fertility						
14 Pollination						
15 Biological control						
Habitat services						
16 Lifecycle maintenance	X			X	X	<5
17 Gene pool protection (conservation)	X				X	<5
Cultural services						
18 Aesthetic information	X	X			X	<5
19 Opportunities for recreation and tourism	X	X			X	>5
20 Inspiration for culture, art and design						
21 Spiritual experience						
22 Cognitive information (education and science)						

*Number of studies identified.

3.4 Sweden

In Sweden, many of the valuation studies focus on topics connected to eutrophication and the value of cultural services such as recreation and aesthetic values. Few of the eutrophication studies are related to habitat services. There seems to be a knowledge gap in the areas of valuing regulating services, such as e.g. pollution control and eutrophication mitigation. Many studies are on recreational benefits of nutrient reduction, but these benefits do not present the whole picture. Studies like Paulsen (2007), linking eutrophication directly to the values of the provisioning of fish, is therefore of high importance. The majority of the studies are done in the Baltic Sea area (see also discussion for Finland below). Methods used are mainly stated and revealed preferences. There are also many examples of the use of benefit transfer. Production based and cost based methods are not that much applied. The most often used methods in the studies reviewed, are the contingent valuation method, choice experiments, market prices and replacement costs.

Söderqvist and Hasselström (2008) made an overview of the existing information and gaps of knowledge related to the economic value of ecosystem services provided by the Baltic Sea and Skagerak. This study was based on a review of studies of all countries around the Baltic Sea, where environmental economists participated from all countries. Their conclusions confirm our review of Swedish studies valuing watershed ES. Within the sector of recreational values of reduced eutrophication, contingent valuation, choice experiments and indirect approaches have been used. There are, however, few studies of marginal benefits of reducing nitrogen and phosphorus loads. More accurate studies relating to this would be of high importance for decision-making, since marginal costs for reducing nitrogen and phosphorus loads are more thoroughly described in the literature (Söderqvist and Hasselström 2008).

However, following Söderqvist and Hasselström's study, the Nordic Ministers of Environment jointly called for a Stern-like review of the Baltic Sea inspired by "The Economics of Climate Change – The Stern Review" (2007). Following this call, a couple of Stern-related studies on the Baltic Sea were initiated. BalticSTERN[19] studies the willingness to pay for reducing the eutrophication in all countries surrounding the Baltic Sea. BalticSTERN is an international research network with the purpose of doing cost-benefit analysis regarding the environmental problems of the Baltic Sea and give guidance toward cost-effective measures and policy instruments. BalticSTERN also aims at contributing to the science-policy dialogue on the management of the Baltic Sea and to discuss policy instruments for reaching environmental targets.

Söderqvist and Hasselström (2008) also conducted a review of the Swedish literature regarding oil spills. These studies make a contribution to the valuation of different cultural and in some respect also provisioning services, by setting up scenarios of (not "yet" occurred) oil spill accidents in different regions. For these scenarios, potential socioeconomic consequences are presented, and in some cases quantified. The links between further biological consequences of oil spills and the values of ecosystem services are, though, not investigated in the Swedish literature. The same study also calls for addressing the benefits of reduced hazardous substances in Sweden. This is an important gap of knowledge, since this might have consequences for supportive, regulating, provisioning and cultural ecosystem services.

The need for and difficulties in describing changes in ecosystem services in monetary terms, made the background for a report written by Kinell et al. (2009) commissioned by the Swedish Environmental Protection Agency. The study refers to the risk of undervaluing environmental

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[19] See Stockholm Resilience Centre: http://www.stockholmresilience.org/research/centrehostedresearchprogrammes/balticstern.4.7fa5f27212621e9277680001174.html

change and the effects on ecosystem services and claims the need for monetary standard values for environmental change and ecosystem services as well as associated guidelines for how these standard values should be applied. They claim that applying such values will give the authorities a basis for making comprehensive and comparable descriptions of changes of ecosystems and the environment resulting from measures to achieve environmental goals. The report suggests monetary standard values for environmental change in Sweden (Box 5).

Box 5 Monetary standard values for environmental change in Sweden

Kinell et al. (2009) establish monetary standard values for environmental change in Sweden. The project was divided into three phases.

Identify:

- existing Swedish valuation studies
- the values currently used by Swedish authorities
- how standard values should be designed to be practically useful

The survey was conducted partly through a questionnaire to a selection of Swedish authorities and partly by completing a review of existing Swedish valuation studies.

The studies were divided into groups based on the subject of valuation. It proved possible to establish an interval of monetary values for e.g. recreation fishing and water quality.

In order to create useful interval for recreation fishing and water quality, it was necessary to express the environmental change subject to valuation in the same physical unit, and express the estimated economic value in one single monetary unit. These two steps included a variety of recalculations and corrections.

The proposed standard values were set as the mean of the observations from different valuation studies that formed the basis of the interval.

Guidelines were further established for the use of the different kind of standard values in the economic analysis.

The report emphasizes, however, that the valuation studies carried out in Sweden today are far too few to meet the needs for valuing environmental changes and impacts on ecosystem services. It would therefore be necessary to carry out a variety of valuation studies in order to create more intervals and standard values as well as for updating the intervals and standard values already calculated within the Kinell et al. (2009) project. With a larger base of valuation studies, it is claimed in this study that it would be possible to calculate the intervals and standard values by more advanced methods (e.g. quantitative meta-analysis) The study also claims a need for valuation studies covering clearly specified environmental change.

Table *5*. Overview of valuation studies of different watershed services carried out in Sweden:

Ecosystem services	Valuation methods					
	Stated preferences	**Revealed preferences**	**Production /damage function**	**Cost based**	**Benefit transfer**	**Total***
Provisioning services						
1 Food & food safety	X	X	X		X	>5
2 (Fresh)Water supply	X	X		X	X	>5
3 Raw materials						
4 Genetic resources		X				<5
5 Medicinal resources						
6 Ornamental resources						
Regulating services						
7 Influence on air quality						
8 Climate regulation						
9 Moderation of extreme events						
10 Regulation of water flows		X				<5
11 Waste treatment/ water purification	X	X		X	X	>5
12 Erosion prevention						
13 Nutrient cycling and maintenance of soil fertility		X				<5
14 Pollination						
15 Biological control						
Habitat services						
16 Lifecycle maintenance	X	X				<5
17 Gene pool protection (conservation)		X				<5
Cultural services						
18 Aestetic information	X	X			X	>5
19 Opportunities for regreation and tourism	X	X			X	>5
20 Inspiration for culture, art and design	X					<5
21 Spiritual experience	X					<5
22 Cognitive information (education and science)						

*Number of studies identified.

3.5 Denmark

Until around 2007, many of the Danish valuation studies were on forests, biodiversity and particularly the recreational value of forests. More recently, several watershed service related studies have been carried out. It should also be noted that several valuation studies consider restoration (rather than preservation) projects, e.g. reforestation projects and restoration of wetlands. When it comes to watershed services, most of the studies are on the recreational and aesthetic values in addition to water purification. Contingent valuation and choice experiments dominate among the methods, but also CR, HP and TC have been applied. Based on a review of Danish valuation studies, Navrud (2007) concluded with the need for more primary valuation studies for establishing general unit values for benefit transfer related to the priority environmental goods, at that time with the exception of forest recreation. Navrud (2007) however has made a comprehensive practical guideline for value transfer in Denmark. If we compare the Danish and Swedish review, we see much of the same pattern in the valuation studies carried out. The

Danish studies are however fewer and has less focus on provision of food and fresh water supply, though water studies have increased in number since the review by Navrud (2007). Traditionally, there has been lower political-administrative interest in valuation studies in Denmark compared to Sweden, Finland and Norway, and therefore somewhat fewer studies.

An interesting Danish study is Schou et al. (2003) which makes a valuation study of the effects of pesticide use. In their case study encompassing valuation of the effects of pesticide-free buffer-zones along field margins, they found respondents willing to accept an increase in the price of bread of DKK 0.57 (4 percent) if the survival of partridge chickens increased by 10 percentage points. Similarly, respondents accepted an increase in the price of bread of DKK 0.07 (0.5 percent) if the number of wild plants increased by 10 percentage points. Based on their findings, they conclude that economic valuation studies of the effects of pesticide use can be performed based on the current knowledge and methods. However, there is a need for further empirical work with respect to validating study methodology and price estimates in order to discuss if the results can provide a meaningful input to policy analysis.

Some interesting examples of valuation studies of different watershed services carried out in Denmark are presented further down in the text.

Dubgaard et al. (2003) present an example of how to conduct a cost-benefit analysis (CBA) of a nature restoration project using unit value transfer methodology (Box 6):

Box 6: Cost-benefit analysis of the Skjern river restoration in Denmark

Dubgaard et al. (2003) is a good example of how to conduct a cost-benefit analysis (CBA) of a nature restoration project. The cost-benefit analysis of the Skjern River Restoration Project was conducted on behalf of the Danish Forest and Nature Agency as part of the investigations by the Wilhjelm Committee, which was appointed by the Danish Government in March 2000. The Committee's assignment was to establish the scientific basis for formulating a national action program for biological diversity and nature conservation in Denmark.

Dubgaard et al. (2003) used a unit value transfer methodology to assess the following social benefits:

- Value as a factor of production (farm land, reed production etc.)
- Ecosystem services (retention of nutrients, flood risk reduction etc.)
- Consumptive outdoor recreation values (hunting, angling)
- Non-consumptive outdoor recreation values
- (hiking, boating, wildlife observation, etc.)
- Non-use value which individuals place on the mere existence of biological diversity.

The first two benefit components were valued using market prices and the replacement cost method. The focus of this value transfer guide is the Stated Preference methods (CV and CE) and the Revealed Preference methods (TC and HP). Dubgaard et al. (2003) make extensive use of unit value transfer to estimate the social benefits of restoring the Skjern River.

Hasler et al. (2005) valuated groundwater protection versus water treatment by the use of Choice Experiments and Contingent Valuation (Box 7):

Box 7: Valuation of groundwater protection versus water treatment in Denmark by Choice Experiments and Contingent Valuation

The benefits of groundwater protection are in a study carried out by Hasler et al. (2005) estimated to assess the non-marketed benefits associated with increased protection of the groundwater resource, as compared to purification of groundwater for drinking water purposes. The study comprises valuation of the effects on both drinking water quality and the quality of surface water recipients, expressed by the quality of the living conditions for wild animals, fish and plants in lakes and waterways. The methods Discrete Choice Experiments method (CE) and Contingent valuation (CV) are used for the valuation. The results indicate that there is a significant positive willingness to pay for groundwater protection, where the willingness to pay for drinking water quality exceeds that for surface water quality. The value of groundwater protection exceeds that from purification, and this result supports the current Danish groundwater policy and the aim of the Water Framework Directive that aims at a holistic management government of the aquatic environment.

Kataria et al. (2011) used choice experiment data for economic valuation and analysed how disbelief in survey information could affect the retrieved welfare estimates (Box 8):

Box 8: Scenario realism and welfare estimates in choice experiments – A non-market valuation study on the European water framework directive

Kataria et al. (2011) used choice experiment data for economic valuation and analysed how disbelief in survey information could affect the retrieved welfare estimates. In their study, they distinguish between two types of survey information to the respondents. The first type of information concerns the current environmental status of a water body. This information is provided prior to the valuation questions and the corresponding beliefs in the provided information are also elicited before valuation. The second type of information concerns the proposed improvements in the environmental status of the water body. They found that average welfare measures differ considerably according to whether respondents who disagree with the status quo levels and find proposed scenarios unlikely are included or not. Their results show that correcting for these dispersed beliefs, although not straight forward, represents a prerequisite for valid interpretation of results. The study was part of the Aquamoney study in Odense, which is described in more detail in Chapter 5

Several recent papers study the economic value of wetland restoration in Stor Åmose. A summary is made in box 9:

Box 9: Valuation of wetland restoration in Store Åmose in Denmark – improving biodiversity and the protection of artefacts

Store Aamose is an open landscape with a mixture of nature areas, small forest areas, wetlands and agriculture, situated east northeast of Tissø in West Zealand in Denmark. The currently preserved area is 230 hectares, and increased protection and restoration of the wetlands have been proposed by the Ministry of Environment as well as the former Ministry for Culture ("Kulturarvstyrelsen".) The proposed restoration scenarios are between 600 and 1500 hectares of the area, and the purpose of the protection is to preserve ecosystem and cultural services and functions in the area. Restoring wetlands in the area will increase the conditions for the biological diversity in the area as well as the preservation of archaeological artefacts. These artefacts are from Stone Age villages which are presently buried within the topsoil in the area, and the wetland restoration can avoid destruction of the artefacts due to agricultural cultivation and drainage. In addition the biological diversity and recreational opportunities will be improved. A channelled stream flows through the area and the wetland restoration will involve changes in the water level of this stream, by some remeandering, removal of pumps and drainage pipes in the area.

(Box 9 continued)

The study was accomplished for the Danish Ministry of Environment in 2005 as a choice experiment, and it was an internet study accomplished by the survey institute GALLUP. It was submitted to a representative sample of respondents in all of Denmark as well to a subsample in the county of Zealand where Åmosen is situated. The results indicate that even though the artefacts are not visible or usable for the population of today, but may contain information and potential value for future generations, the strongest preferences displayed is for ensuring permanent protection of archaeological artefacts, rather than biological diversity. The willingness to pay for improved biodiversity is positive however, but the willingness to pay for improved recreational services in the area are negligible, and the willingness to pay for improved recreational improvements drop to zero within a very short distance from the wetland area.

In addition to the choice experiment study in Store Åmose a follow up study was done in 2006. The project in Store Åmose was only one of several projects that was considered for publicly funded nature restoration projects at that time, and a number of national nature parks were also considered launched. The obvious question raised by the policy makers was, if the value derived from an environmental valuation study of one of the projects depends on the overall scale of the Danish nature restoration activities?

For instance, would people state the same willingness-to-pay for this particular project if they knew it was going to be one out of three or seven projects producing ecosystem goods and services of a similar character?

The results of this study clearly demonstrate that respondents, through their choices in the choice experiment, actually can relate to the level of attributes, and apply internal consistency and scope sensitivity within an experiment. It was found that when introducing more substitute projects (nature national parks of similar magnitude) the willingness to pay for biodiversity improvement in Store Åmose was reduced between 10 and 20%. The WTP did not change differently between two splits where two and seven national parks were presented respectively. For artefacts in Stpre Åmosen the WTP was not reduced at all because the nature national parks do not deliver these cultural services, and for the artefacts the preferences seem to be more lexicographic than the biodiversity protection.

References:

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Jacobsen, J.B, T. H. Lundhede, L.Martinsen ,B.Hasler, B.Jellesmark Thorsen (2011). Embedding effects in choice experiment valuations of environmental preservartion projects. Ecological Economics. Elsevier, vol. 70(6), pages 1170-1177.

Hasler, B., Jacobsen, J.B., Lundhede, T., Martinsen, L., Thorsen, B.J. 2008. Værdisætning af natur- og kulturgoder. Et metodestudie af betydningen for ændringer i skala og betalingsformat, Danmarks Miljoundersogelser, Aarhus Universitet

Lundhede, T., Hasler, B., Bille, T. 2005. Værdisætning af naturgenopretning og bevarelse af fortidsminder i Store Åmose i Vestsjælland, Skov- og Naturstyrelsen

Table 6. Overview of valuation studies of different watershed services carried out in Denmark

Ecosystem services	Valuation methods					
	Stated prefer-ences	Revealed prefer-ences	Production /damage function	Cost based	Benefit transfer	Total*
Provisioning services						
1 Food & food safety	X	X			X	<5
2 (Fresh)Water supply	X				X	<5
3 Raw materials						
4 Genetic resources						
5 Medicinal resources						
6 Ornamental resources						
Regulating services						
7 Influence on air quality						
8 Climate regulation						
9 Moderation of extreme events						
10 Regulation of water flows					X	<5
11 Waste treatment/ water purification	X			X	X	>5
12 Erosion prevention						
13 Nutrient cycling and maintenance of soil fertility		X				<55
14 Pollination						
15 Biological control						
Habitat services						
16 Lifecycle maintenance	X				X	<5
17 Gene pool protection (conservation)	X				X	<5
Cultural services						
18 Aestetic information	X	X			X	>5
19 Opportunities for recreation and tourism	X	X			X	>5
20 Inspiration for culture, art and design						
21 Spiritual experience						
22 Cognitive information (education and science)						

*Number of studies identified.

3.6 Finland

In the Nordic Environmental Valuation Database, mentioned above, which includes most Nordic studies until the end of 2006, 41 Finnish studies was identified (as compared with 25 Danish, 37 Norwegian and 60 Swedish studies). Finland has an abundance of both forests and water systems, so there is no surprise that the valuation studies are concentrated in these two areas. It seems that around a quarter of the studies are directly related to watersheds services, including, lakes and streams, wetlands, coastal areas and the Baltic Sea. Of the water-related studies, the majority seems to have valued changes in water quality and the resulting recreational benefits. Some studies also include biodiversity, aesthetics and existence values.

Stated preference methods dominate, and in recent years choice experiments have been increasingly used in Finland. Other methods, such as hedonic pricing (within the revealed preference family), has also recently been used to quantify the effect of fresh water quality on recreational house prices (see Artell 2011).

As the water quality of the Baltic Sea has gradually deteriorated, several recent studies investigate the benefit of measures reducing the runoff from agriculture and water treatment in the countries surrounding the Baltic Sea, including several of the willingness to pay of the Finnish population. Good sources for an overview of these studies include the recent studies by Ahtiainen (2009) and Kosenius (2010). Several projects are ongoing investigating further abatement measures to reduce further eutrophication of the Baltic Sea, some of which are explicitly applying the ES framework. Two examples include the research project "Protection of the Baltic Sea: Benefits, Costs and Policy Instruments" (PROBAPS),[20] the perhaps most comprehensive Baltic Sea valuation study to date, and the project "PREHAB – Spatial Prediction of Benthic Habitats in the Baltic Sea".[21]

A recent example of a contingent valuation study of river restoration measures under the Water Framework Directive, that aims to capture a range of ES from small river system, is provided in text box 10.

As for the other Nordic countries, there seem to be a gap in the valuation literature related to regulating/habitat services.

Box 10: The significance of streams for the residents of the City of Helsinki - Contingent Valuation Study for the feasibility of the Small Water Action Plan

The aim of the study was to examine the significance of streams for the residents of the City of Helsinki and to define the monetary value of the planned restoration measures to the area according to the Small Water Action Plan of Helsinki. The motivation behind the Action Plan is to promote biodiversity conservation according to the Finnish Biodiversity Action Plan, e.g. implementing the UN Convention on Biological Diversity (CBD) at the national level. Also the EU Water Framework Directive obligates the Member States to pursue good ecological status of the surface and ground waters by the year 2015 (Box Figure 1). According to the vision of the Small Water Action Plan, the numerous small waters located in the City of Helsinki constitute a diverse network that contributes to biodiversity conservation and forms an integral part of the local identity. In addition the provisions of the EU Water Framework Directive are implemented by means of the restoration measures proposed in the Action Plan.

[20] See www.mtt.fi/probaps

[21] See http://www.prehab.gu.se/research/Ecosystem+valuation/valuation-study---questionnaire/

(Box 10 continued)

The study was carried out in co-operation with the Finnish Environment Institute and the Public Works Department of the City of Helsinki in the year 2010. The contingent valuation method was used. The questionnaire was sent to 700 households in Helsinki and 265 answers were received giving a response rate of 38%. The survey focused on estimating the value of improvement in ecological status of streams, resulting from dedicated restoration actions. The streams and their surroundings supply a range of ecosystem services. The survey focused on the following aspects: prevention of erosion, flood and storm protection, different cultural benefits (e.g. aesthetic value) and maintenance of biological diversity and wilderness values. Changes in water quality, biodiversity and stream morphology were described with the pictures in the survey instrument and portraying the improvement in better water quality and natural water management as well as recovery of the stream profile/channel scheme, vegetation and fauna. Effort was made to describe the objective and foreseen benefits of restoration measures as concrete as possible: storm water would be filtered through wetlands before entering into the stream, floodplains would be constructed to prevent adverse flooding of the stream, and streams would provide shelter for the fauna enabling the spawning of the highly endangered sea trout among other things. Furthermore, possible changes to the scenery and recreational use of streams were described.

The beneficiaries of the restoration measures were the residents Helsinki, with foreseen benefits including both use and non-use values. In the survey, the respondents stated their willingness to pay by choosing a bid from the payment card which allowed them to express their possible uncertainty to each bid.

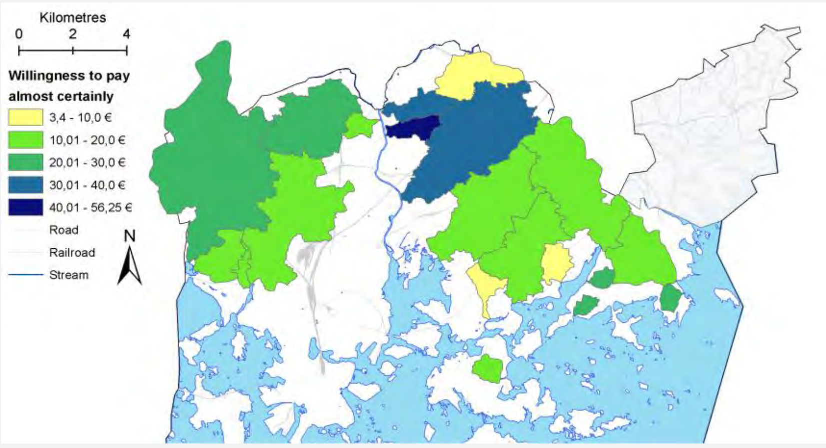
The results of the survey indicated that the residents with high income, low age, exercising outdoors and living near the streams of Tapaninkylänpuro, Tapaninvainionpuro or Longinoja were willing to pay more for the improvement in streams. The total benefit estimate was approximately 1.4 million Euros (2010) per year and about 7.2 million Euros (2010) for the five year period of the fictional and regional Small Water Fund. The estimated total value exceeded manifold the total budget targeted to restoration. Furthermore, respondents' previous experience of the outcomes and benefits of restoration measures may explain their high willingness to pay in specific watersheds (Box figure 2). For example in Longinoja several restoration measures have been carried out in the past, e.g. gaining high publicity in the local media. It appears that the CV-method fits well in monetizing ecosystem services in stream waters. The CV-study – when implemented from the societal point of view – may give essential information on ecosystem services to the general public and stakeholders and contribute to decision making.

Source: Information provided by Virpi Lehtoranta, of Finnish Environment Institute, SYKE. Finland

(Box 10 continued)

Stream water quality in Helsinki according to the Small Water Action Plan of Helsinki (2007)

The average willingness to pay of the residents of the City of Helsinki for better condition of stream water.



Another study carried out as part of the PROBAPS project is Hyytiäinen and Huhtala (2011) who evaluate the profitability of nutrient abatement measures in eutrophied coastal areas exposed to a risk of frequent oil spills.

Box 11. Combating eutrophication in coastal areas at risk for oil spills.

Hyytiäinen and Huhtala (2011) evaluate the profitability of nutrient abatement measures in eutrophied coastal areas exposed to a risk of frequent oil spills. The case studied is the Gulf of Finland, which forms part of the Baltic Sea. They present a dynamic model that integrates land loads of nitrogen and phosphorus, cost of nutrient abatement measures in agriculture, nutrient dynamics in the sea basins adjoining the Finnish coast, exogenous risk of oil spills, and recreational value of the sea, which faces environmental damage of uncertain magnitude and duration. Monte Carlo simulation is applied to evaluate the profitability of nutrient abatement measures carried out unilaterally by Finland or as a joint effort by Estonia, Finland and Russia. They demonstrate that a high exogenous risk of oil damage may render investments in nutrient abatement economically infeasible. On the other hand, several components of the model entail uncertainties owing to the scarcity of data and the limited understanding of the relationship between the ecological processes involved and the values people place on natural resources. For example, the uncertainties related to the curvature of the value function outweigh the uncertainties connected with the oil spills and their potential consequences.

Table 7. Overview of valuation studies of different watershed services carried out in Finland

	Valuation methods					
Ecosystem services	**Stated preferences**	**Revealed preferences**	**Production based**	**Cost based**	**Benefit transfer**	**Total***
Provisioning services						
1 Food and food safety	X	X				>5
2 (Fresh)Water supply	X					<5
3 Raw materials						
4 Genetic resources	X					<5
5 Medicinal resources						
6 Ornamental resources						
Regulating services						
7 Influence on air quality						
8 Climate regulation						
9 Moderation of extreme events	X					<5
10 Regulation of water flows	X			X		<5
11 Waste treatment/ water purification	X			X		>5
12 Erosion prevention						
13 Nutrient cycling and maintenance of soil fertility	X					<5
14 Pollination						
15 Biological control						
Habitat services						
16 Lifecycle maintenance						
17 Gene pool protection (conservation)						
Cultural services						
18 Aesthetic information	X	X			X	>5
19 Opportunities for recreation and tourism	X	X			X	>5
20 Inspiration for culture, art and design	X					<5
21 Spiritual experience	X					<5
22 Cognitive information (education and science)						

*Number of studies identified.

3.7 Iceland[22]

Iceland does not have the same tradition for environmental valuation as some of the other Nordic countries. In the Nordic Environmental Valuation Database only four studies from Iceland have been identified and included. Three of these are related to watershed services and all three use stated preference techniques. Bothe (2003) values preservation of wilderness confronted with hydropower development, with specific emphasis on cultural services, especially existence/non-use values and (more indirectly) ecological functions. Lienhoop and MacMillan (2007) also carry out a variation based on the contingent valuation method, the so-called market stall approach, where wilderness preservation confronted with hydropower development is the main object of valuation. Both studies are on the Karahnjukar hydropower development, the most recent and largest in Iceland.

Toivonen et al. (2000) conduct a contingent valuation study of freshwater recreational fisheries in the Nordic countries, including Iceland. Based on this study, Kristofersson and Navrud (2007) conducts a benefit transfer experiment for economic use values of freshwater recreational fisheries. A more recent study, not included in the database, also use the contingent valuation method to investigate tourists' willingness to pay entrance fees to visit natural wonders, where the Gullfoss waterfall is one of the attractions (Reynisdottir, Song et al. 2008). These are all "classic" valuation studies that do not adopt the ES framework.

More recently, Iceland has started a research project which aims specifically to investigate ecosystem service values related to the Heiðmörk Nature Reserve, following the TEEB/MEA framework (see box 13). The area provides an outstanding example of a multifunctional ecosystem. Many of the services are watershed related. The perhaps most important services the Heiðmörk ecosystem provides are drinking water and recreational services. The area is a key water supply area for the Great Reykjavík area, harboring the Gvendarbrunnar wells that supply drinking water to more than half of the Icelandic population. Also, the area is a widely popular for recreation with accessible forests, lakes and open spaces, attracting over 500,000 visitors the year around. In addition, more indirect services include educational and cultural, carbon sequestration services and habitat services for various bird and fish species. Finally, the area provides the outer range/backdrop sheltering the capital settlement areas.

Even though Iceland has some way to go in terms of reaching the level of application of valuation methods in the other Nordic countries, the

[22] Due to the limited number of studies, we do not make a table summarizing studies and methods, as done for the other Nordic countries.

country may well have one of the most interesting case studies on ES valuation in the Heiðmörk Nature Reserve case.

Box 12: Estimating the Value of Ecosystem Services: the Heiðmörk project

The first research project on ecosystem services in Iceland is a multi-year, multi partner project. The overall objective is to provide the first comprehensive evaluation study for ecosystem services in Iceland, which can serve as a benchmark for future studies. It is expected to lay the foundation for classification of ecosystem services in Iceland, to build capacity in applying appropriate valuation methods for each service and thereby enable the use of the term in economic decision-making. Finally, it is intended to increase awareness of the importance of the multiple services derived from natural capital, and thereby enrich the national discourse on resource use by swaying the discussion away from the conventional one-dimensional view of nature.

Heiðmörk is an extensive, yet clearly defined nature reserve, bordering Reykjavík, Garðabær and Kópavogur. It encompasses around 3500 hectares of forests, lava fields, lakes and open areas. The area provides an outstanding example of a multifunctional ecosystem, where a range of services can be identified. The system components and its associated services are as follows:

- *The Water Catchment Area;* which has a water supply function, as the area is an important catchment area for Reykjavik, providing clean drinking water and thereby providing provisioning services. To assess the value of the water provisioning services two separate valuation methods were used: replacement cost and cash flow analysis
- *The Forest and Vegetation;* which provides multiple services such as: (a) provisioning services such as timber, Christmas trees, medicinal herbs, mushrooms and berries (b) support and regulating services such as carbon sequestration services and water filtration and (c) cultural and amenity services such as recreation

(Box 12 continued)

Cultural and amenity services are assessed as their own component of the study, using the contingent valuation methods, and water filtration services were evaluated as part of the water catchment component. For instance, would people state the same willingness-to-pay for this particular project if they knew it was going to be one out of three or seven projects producing ecosystem goods and services of a similar character?

- *The lakes Elliðavatn/Vífilstaðavatn* which provide (a) provisioning services such as fish harvest as well as serving as a reservoir for a hydropower plant, (b) supporting services such as maintenance of nutrition for Elliadar river (c) regulating services such as pollution dilution for the surrounding habited areas in addition to (d) cultural services such as education and recreation. In order to prevent double counting the value of fish harvest was excluded from the assessment, as most fish in the lakes for recreation purposes. Both the travel cost and contingent valuation methods have been used

Data collection in the Heiðmörk project has been completed, and the study is quickly moving through its analysis phase. However, the project is already fulfilling its objective. Awareness of the importance of ecosystem services has increased in Iceland, and its incorporation into national and local decision-making has been proposed.

Source: Some of the results of the economic valuation of ecosystem services during the Heiðmörk project are reported in the edited volume of Daviðsdottir (2010) (most of which is in English).

3.7.1 Summary and gaps for the Nordic countries

The watershed ecosystem services valuated are quite similar among the Nordic countries (Table 8). The literature review has been selective in seeking out studies that capture economic values of ecosystem services that are not directly reflected by market prices. This is one reason for provisioning services such as “raw materials” are poorly represented.

The review nevertheless shows us that the services addressed mainly are provisioning services as food and fresh water supply, regulating services like regulation of water flows and water purification, as well as cultural services as recreation, including recreational fishing, aesthetic information and opportunities for culture, art and design. Habitat services are valued mainly in terms of biodiversity and conservation in general. There are also a majority of studies performed in areas under high environmental pressure. Contingent Valuation and Choice Experiments dominate among the methods, but also deliberative or participatory valuation approaches, production function/damage function as well as Contingent Ranking, Hedonic Pricing and Travel Cost have been applied. There is a need of more studies using production and cost based methods and integrated models as well as a need of more primary valuation studies for establishing general unit values for benefit transfer related to the priority environmental goods. With a larger base of valuation studies, it would be possible to calculate the intervals and standard values by more advanced methods (e.g. quantitative meta-analysis).

A gap recognised at the EU level is the lack of valuation studies that look at the consequences of air pollution on (amongst others: watersheds) ecosystems (and hence their ES).[23] Our overview in the Nordic countries also shows that there is actually a lack of studies in this area.

There seems to be a knowledge gap in the areas of valuing regulating services, such as e.g. pollution control. Many studies are done of recreational benefits, but these benefits do not present the whole picture. More accurate studies relating to marginal benefits of reducing nitrogen and phosphorous loads would be of high importance for decision-making, since marginal costs for reducing nitrogen and phosphorus loads are more thoroughly described in the literature. There are also calls for addressing the benefits of reduced hazardous substances.

There seems to be trend in the valuation of direct uses as opposed to indirect use. Direct uses are more easily apprehended by the population and their own direct experience (fresh water, fishing, recreation, existence value for a whole system). Direct uses are more often subject to health standards (bathing, drinking water standards) and regulatory pushes to undertake valuation (e.g. WFD).

Indirect uses in environmental economics are linked to regulating and supporting services, where ecosystem services are inputs or intermediate to final services enjoyed by people. Establishing production and damage functions that are spatially and temporally explicit is a large integrated modelling challenge, and has rarely been undertaken in the context of valuation (although there are some experiences with driver-pressure-state-impact-response (DPSIR) modelling of nutrients and pollutants that could easily be re-interpreted in the light of ecosystem services). Finally some ecosystem services, e.g. related to spiritual experiences, education and knowledge advancement may not be conducive to valuation based on consumer preferences.

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[23] See http://ec.europa.eu/environment/air/pollutants/valuation/index.htm-

Table 8. Overview of watershed service valuation studies in the Nordic countries, following the TEEB categorization

	Country			
Ecosystem services	**Denmark***	**Finland***	**Norway***	**Sweden***
Total:				
Provisioning services				
1 Food and food safety	<5	<5	>5	>5
2 (Fresh)Water supply	<5	<5	<5	>5
3 Raw materials				
4 Genetic resources		<5		<5
5 Medicinal resources				
6 Ornamental resources				
Regulating services				
7 Influence on air quality				
8 Climate regulation				
9 Moderation of extreme events				
10 Regulation of water flows	<5	<5	>5	<5
11 Waste treatment/ water purification	>5	>5	>5	>5
12 Erosion prevention			<5	
13 Nutrient cycling and maintenance of soil fertility		<5		<5
14 Pollination				
15 Biological control				
Habitat services				
16 Lifecycle maintenance			<5	<5
17 Gene pool protection (conservation)			<5	<5
Cultural services				
18 Aesthetic information	>5	>5	<5	>5
19 Opportunities for recreation and tourism	>5	>5	>5	>5
20 Inspiration for culture, art and design				<5
21 Spiritual experience				<5
22 Cognitive information (education and science)				

*Number of studies identified.

4. Demonstration watershed I: Glomma River Basin

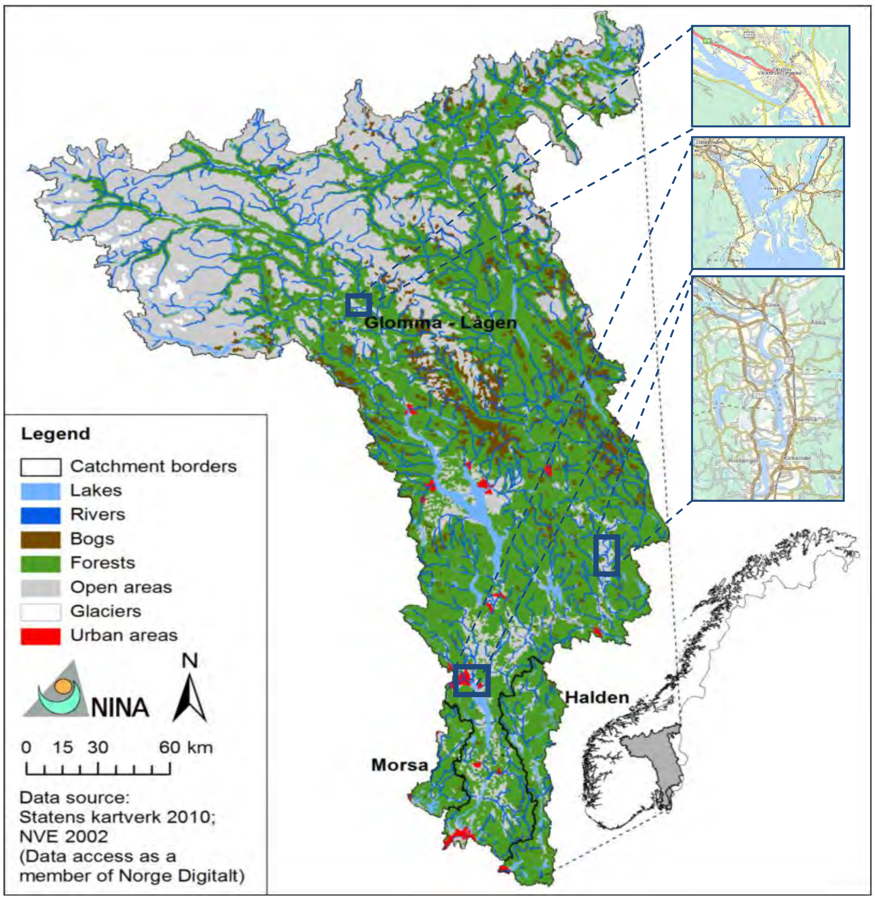
Chapter 4 provides some detail from valuation studies selected in the Glomma-Lågen Water Region. We look at approaches to valuing reduction in flood damages in this river basin, improvement in the ecological status of lake water across the three catchments of the Glomma-Lågen Water Region. A detailed look at valuation studies at the river basin and local within-river basin scale serve as a source of discussion of methodological challenges and solutions for valuation methods applied to different watershed services across different geographical scales.

The examples are (1) evaluating flood prevention versus wetland conservation in the upper catchment in Ringebu Municipality using multi-criteria analysis; (2) valuing flood damage in the mid-catchment near Grue & Åsnes municipalities in the mid-catchment using damage functions, (3) evaluating environmental flows, versus flood damage versus hydropower production in near Lake Øyeren in the lower catchment using multi-criteria analysis, (4) non-market valuation of water quality in the Morsa, Lower Glomma and Hal-den watersheds.

4.1 Watershed description

The Glomma-Lågen watershed is the largest in Norway. It is part of the Glomma Water Region under Norway's implementation of the Water Framework Directive. Glomma Water Region also includes the Morsa and Halden watersheds, which lie to the south-west and south-east of the lower Glomma respectively.

Figure 4: Glomma-Lågen watershed in Norway and three valuation case studies related to ecosystem service of flood reduction.



The Glomma-Lågen, Morsa and Halden river basins have surface areas of 41541 km², 688 km² and 1570 km², respectively. Lakes, rivers and bogs/mires cover a 12%, 9%, and 13%, respectively of total river basin area. Forest area covers, 49%, 72% and 74%, respectively of the area. Agriculture represents a modest 6%, 12% and 12% of landcover, respectively. Urban areas cover 1% or less in the three catchments (Figure 4).

For the Haldenvassdraget watershed approximately 43% of its lakes by surface area are in poor or moderate status, while more than half have not been characterized. More than half of the lake area in the river basin is at risk of not reaching good ecological status by 2015. Another 27% are possibly at risk. In total 82% of the lake areas would therefore be subject to supplementary measures under the WFD. This means that most of the river basin should also be subject to another economic evaluation of whether benefits of measures exceed costs. Eutrophication is the most important reason for lakes in the river basin not achieving good ecological status. In Morsa (Vansjø-Hobøl-vassdraget) watershed practically all lake area is in current moderate status and at risk of not reaching good ecological status by 2015. Eutrophication is Morsa's most important reason for not reaching good ecological status, even more so than for Halden watershed. There are large problems with nutrient loading and algal blooms in Halden river basin, especially in the upper parts of the catchment. Haldenvassdraget conducted a full basin wide characterization according to the WFD in 2003. The characterization shows that there is a high risk of not achieving the WFD objectives by 2015 in lakes and the adjoining fjords. Morsa has similar problems with excessive nutrient loading and blue-green algal blooms, but principally in the Vestre Vansjø and to a lesser extent in Storefjorden Lake in the lower part of the catchment.

4.2 Flood reduction

In this example we discuss valuation of reduction of flood peaks – mitigation of one type of extreme event in TEEB terminology – and how this may be associated to specific modifications of land use in the watershed. The examples are all based on secondary data, notably from the HYDRA project report series (Sælthun et al. 2000) and Barton, Berge and Janssen (2009) and Barton and Dervo (2009). The examples are laid out as follows. Valuation Step 1–3 are discussed in general for the whole watershed independently of each valuation examples, while valuation steps 4–7 are specific to each example. We return a broader discussion of aggregation across the whole watershed in Step 8 and validation in Step 9.

4.2.1 Step 1: Policy scenarios as basis for valuation

What kinds of decisions or policy can be supported through valuation of flood peak reduction as an ES?

In the TEEB report ES are referred to in a similar fashion to how one previously discussed "environmental costs or negative externalities." Valuation of foregone flood reduction services can, from a policy perspective, be addressed similarly to the "polluter pays principle". However, with flood reduction services of ecosystems the focus is shifted to "positive external effects". In watersheds with large infrastructure and land-use modifications that affect run-off, the relevant policy questions are often; how valuable is it to restore ecosystems in the catchment in places where they provide a service? Which parts of the watersheds' landcover (read ecosystems) provides a service? Where is this service enjoyed? How can such restoration be paid for?

Definition of policy scenarios of relevance for valuation is however, first about clarifying a distribution of rights to a given ES, such as flood reduction.

ES refer both to benefits following ecosystem restoration, but also to avoided costs of increased flood damage due to losses of ecosystems in the watershed. Whether reduction in streamflow peaks is perceived as an ES or a cost depends on rights associated with land-uses and regulation of watercourses. For example, a hydropower operator such as Glomma-Laagens Brukseierforening (GLB) regulate a number of reservoirs the length of the watershed; riverine landowners may refrain from placing flood walls along wetlands and floodplains; cabin owners may refrain from draining mountain boggs; forest owners may refrain from draining forest stands, from carrying out clear-cutting or making new forest roads in steep terrain. All of these actions can have an effect on flooding. Whether these land-uses are perceived as benefits by the downstream population depends on whether these downstream interests do not have rights to avoid flood damages if none of the actions upstream were to be carried out.

Hydropower regulation of reservoirs to avoid flooding is part of concessionary obligations. The effects of such regulation on flooding are relatively easy to measure and model (Sælthun et al. 2000). Perhaps because consequences of actions are easy to evaluate, it is an imposed social responsibility of the hydropower company and flood regulation services are not compensated by downstream users. In the case of land-use management of bogs, forest or riverine wetlands/floodplains it is much more difficult to predict effects on downstream flooding (Sælthun et al. 2000). When effects of land-use are hard to predict it is difficult to define legal rights that downstream interests may have on how upstream land-use is conducted. The ability to predict flooding effects of upstream land-use is a key to value flood reduction services, but also for what kinds of policy questions can be addressed.

For example, if flooding effects of specific land-uses can be predicted, this might be sufficient to establish legal rights of downstream interests to flood-reducing land-uses (as they have the right to flood reducing hydropower operation). Valuation may not be necessary to bring about a land use change if these rights can be defined (Vatn, Barton et al. 2011). Under what conditions would it be in the public interest to allocate land-use rights to the upstream land-users, and establish a scheme to compensate them for flood-reducing land-use (payments for ecosystem services)?

For flood reduction it is the case in Norway as in most of the world, we would guess that upstream land-users have full rights to land-uses because authorities have little predictive capacity on downstream impacts of land-use. Valuation of flood reduction is likely to take place in the context of PES for flood-avoiding land-use upstream. Policy relevant valuation of flood reduction depends crucially on:

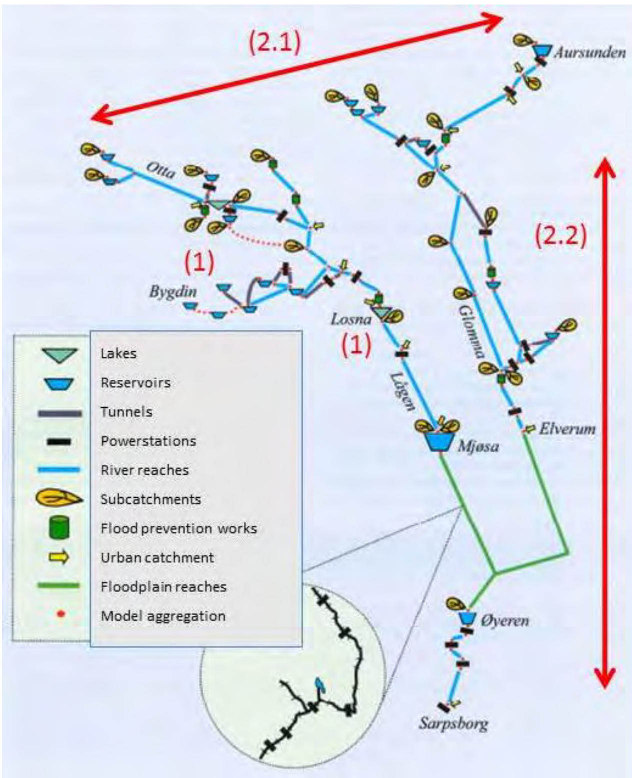
- Predictive modelling capacity of flooding effects of different land-uses (ecosystem function)
- Clear definition of land and water use rights (of rights to ES)

As we will show with the examples below, neither is clear in the case of valuing flood reduction. For this reason, authorities and individuals in

Norway – and in Nordic countries in general – do not use valuation of environmental impacts in benefit-costs analysis when determining whether hydropower concessions should be granted, in municipal land-use planning, in forestry planning, decisions regarding flood prevention measures or where to build in flood-prone areas.

Are there any new laws or regulations justifying future use of more valuation of flood reduction services in policy analysis? The most relevant regarding the evaluation of the benefits of better ecological status– ES – of water bodies is the Water Framework Directive (and its regulatory equivalents in Norway and Iceland). The WFD requires, as previously noted, an evaluation of whether it is disproportionately costly to attain the objectives of good ecological status in water bodies. Disproportionality in economic terms is in relation to the benefits of ES. Good ecological status includes flow characteristics and river morphology and would seem to address both the costs and benefits of flood mitigation measures in watersheds.

Figure 5: Priority setting issues in water-shed management that may be informed by benefit-cost analysis and valuation of flood reduction.



Source: Elkenæs et al. (2000), Hydra project.

The diagram shows a hydrological conceptualisation of the Glomma-Lågen River Basin, with the main features affecting streamflow. The Water Framework Directive requires evaluation of disproportionate costs of hydromorphological impacts due to e.g. hydropower infrastructure (1). Benefit-cost analysis could in principle also be used to prioritise across measures in the watershed (2), either across landuse management measures in the upper catchment (2.1.), or the length of the river basin (2.2) between upstream, mid-stream and downstream measures that improve good ecological status.

In some countries benefit-cost analysis is also required of flood mitigation explicitly. The Norwegian Directorate for Water and Energy (NVE) requires benefit-cost analysis in determining the dimensions of flood mitigation infrastructure. Their 1999 guidelines recommend the use of multi-criteria analysis to address environmental impacts. These guidelines have been evaluated by Barton and Dervo (2009) and are illustrated in one of the examples in this chapter.

Perhaps the greatest economic relevance of valuation of flood reduction is regarding insurance against flood damages. Flood insurance – both private and public – plays a potentially large role in risk exposure, the size of flood damage, insurance claims, the economic costs of flooding, and finally the potential economic value of ecosystem management measures in the watershed that might reduce flood risk. In Nordic countries such as Norway, the State acts as a final insurer of extreme natural hazards such as floods. This reduces the incentives of municipalities, businesses and individuals to locate to less flood exposed areas. Flood zone mapping carried out extensively in countries such as Norway (for 10, 50, 100, 200, 500 year return periods) are designed to encourage flood avoidance behaviour. Flood avoidance behaviour incurs some costs, but avoids flood damage, also determining the valuation of flood reduction benefits of ecosystem management in the watershed.

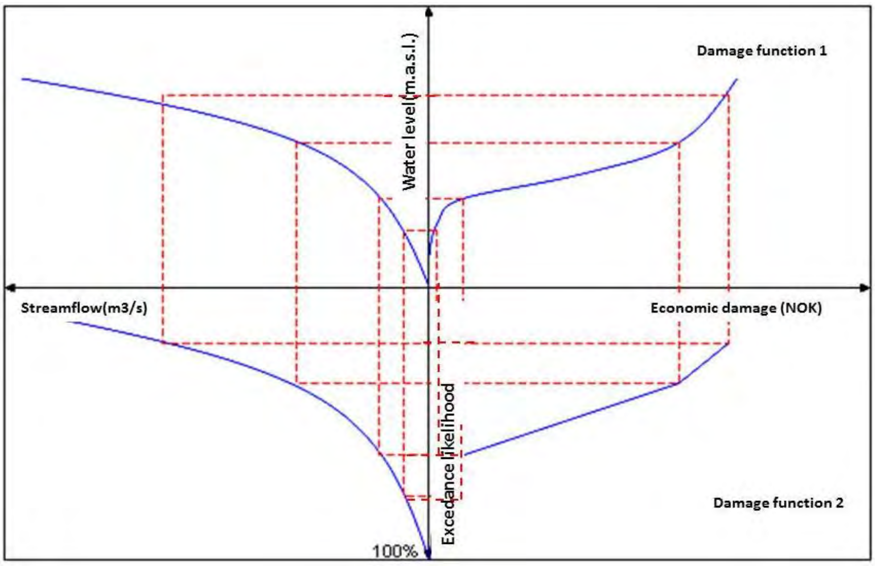
The evaluation of disproportionate costs under the WFD is intended as an economic evaluation from a social accounting standpoint. In principle this opens up for a number of possible policy applications of valuation of ES in watershed management (Figure 5). However, the examples below also illustrate large challenges for valuation particularly regarding (i) identifying the chain of watershed service providers and beneficiaries the length of the watershed (ii) predicting effects of upstream measures on downstream water levels (ecosystem function), (iii) determining both ES/benefits and disservices/costs of upstream land-uses.

4.2.2 Step 2: Definition of measures and identification of environmental change

What kind of measures and land-use are significant for run-off and therefore determine the environmental change to be valued, in this case reduction in flooding? An economic definition of "flooding" is needed. Valuation of the expected reduction in flooding damages due to management measures, can be done by comparing the area under a damage function before and after such upstream land and water use management measures are undertaken (Figure 6). A look at the figure shows that not all reductions in streamflow have impacts that are economically significant.

Systematic evaluation of all major factors affecting the flood damage function across a large river basin is rare in Nordic countries. The present case study draws heavily on the Hydra project study in the Glomma-Lågen River Basin, initiated after a 100–200 year flooding event in 1995 called "Vesleofsen". The Hydra project evaluated the incremental contribution of urban, agricultural and forest land cover and management practices to streamflow using a hydrological model of the whole river basin (see Figure 5). Local effects of urban drainage and storm flow, flood prevention walls and road and rail infrastructure was evaluated using hydraulic models. Specific studies were also undertaken of forestry and agricultural practices impacts on run-off using spatially explicit run-off models. The Hydra project was completed in 2000 after 3 years research (Sælthun et al. 2000). Different watershed management scenarios were simulated. In particular the hydrological model was run with land-use around the year 1900, which was compared to stream flows and flooding with land-use around 1995, for an event the size of "Vesleofsen".

Figure 6: An economic definition of flooding using a flood damage function



Source: adapted from Sælthun et al. (2000).

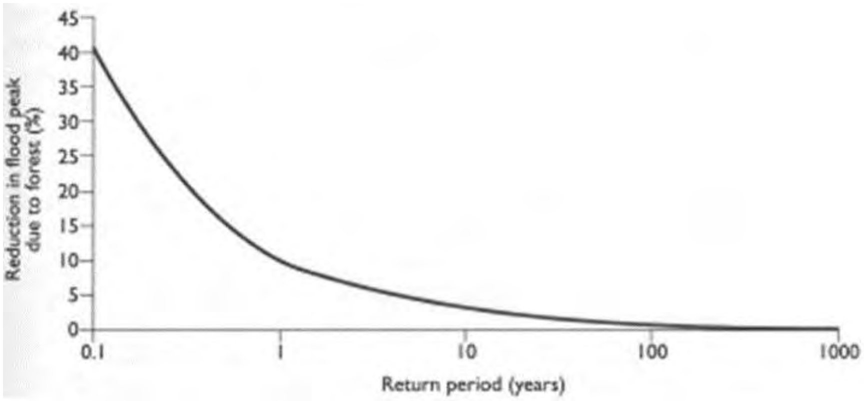
Increasing streamflow leads to increasing water level (upper right panel); above a certain water level, there is “flooding” in the sense that economic damages increase rapidly with increasing water level as the river leaves its course (upper left panel). Streamflow of a certain rate occurs with a certain frequency or “return period” which together with riverbank characteristics, including flood prevention, determines “exceedance likelihood” – the probability in a given time period that the river will exceed its “normal” course (lower left panel). Flood prevention reduces exceedence likelihoods. Exceedance likelihood multiplied by economic damages at different water levels, determine expected flood damages. The potential economic value of flood reduction due to upstream management measures (including ecosystem restoration or conservation) is the difference between the area under the flood damage function before and after flood managementmeasures are taken.

The choice of baseline is non-trivial in evaluating the value of ES. A baseline determines a “reference condition” for scenario analysis, but can very quickly be interpreted as a definition of “normality”, against which rights to environmental quality and access to resources is evaluated, and gains and losses are defined.

The choice of a year 1900 baseline was made partly because of data availability, e.g. national forestry inventory data were not available before then, and this was before the era of hydropower development of the basin. Hydra studies go on to conclude that the change in total area under farming as a percentage of the Glomma-Lågen has been so small during the past century relative to the total area of the river basin, that their impact on the course of the Vesleofsen flood was marginal.

Damming of riverine floodplains with flood walls has had small effects on water levels, but did lead to delays in the arrival of flood peaks downstream by several hours in some places (Berg et al. 1999). To the extent that this delay helped downstream emergency flood preparedness the measures could have had significant benefits. Damming floodplains had a number of negative biodiversity impacts as will be evaluated in one of the examples below.

Figure 7: Schematic representation of the reduction in flood peak that might be expected from an "average" forest for floods of different magnitudes



Source: Calder 2005.

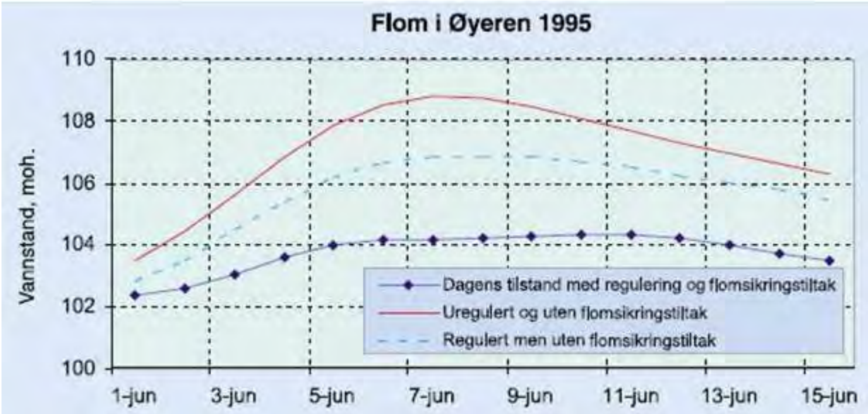
Glomma-Lågen consists of 49% forested area. Forest area changed little since 1920 when the first national forest coverage inventory was conducted. However, since 1920 forest volume per hectare has risen by 75% until 2000. Hydra project run-off modelling in sub-catchments with different forest cover indicate that no flooding effect of changing forest land use can be observed in those larger than ca. 1000–1700 km2. In sub-catchments of 3–7 km2 flooding effects can be observed for hypothetical scenarios of 100% clear cutting of the whole area. Clear cutting led to simulated run-off increasing by 57% (Rinde et al. 2000).

Findings from the Hydra project are supported by international research (Kiersch 2000, Calder 2005, Ennaanay et al. 2011). Landcover changes can have significant impacts in small catchments locally. In large watersheds flood peaks are attenuated. Vegetation cover such as forest can have a significant effect on small and medium floods (e.g. 10 year return periods), but no effect on large floods (100 year return period) (see Figure 7). Impacts of changes in land-use on flooding have been observed in watersheds of less than 1000 km2 , while impacts of forest clear-cut have been observable in catchments of less than 600 km2 (Kiersch 2000)(Ennaanay et al. 2011).

Floodplains function as storage of flood waters and have been cited as a reason for potentially large flood peak attenuation services internationally; but floodplains of Glomma-Lågen represent a relatively small surface area relative to the total catchment area (pers. com. Nils Roar Sælthun).

Forestry practices and drainage of plantations, forest roads have a larger impact on run-off than forest coverage alone (Calder 2005). Areas with forest roads and drainage in Glomma-Lågen are nevertheless so small that they were not considered to have an effect on flooding at watershed scale and under conditions of the "Vesleofsen" flood.

Figure 8: Combined effects of flood prevention measures and Reservoir regulation on local flood water levels in the Øyeren Lake. Combined measures saw a lowering of 2.5 meters relative to a situation without measures.



Source: Sælthun et al. 2000.

Local technical measures such as lowering of reservoirs prior to a flood pulse had the largest flood mitigating effect of all measures evaluated by Hydra (Figure 8).

The overall conclusion is that the geographical scale and return period of floods matters in terms of what management of land cover affects flooding. For this reason Hydra's run-off modellers may have ignored scale effects in hydrology when extrapolating their findings to the whole Glomma-Lågen River Basin. They suggest a hypothetical 18%–24% reduction in flood peaks with a 100% loss of the forest cover in the river basin (Rinde et al. 2000). These findings have important bearing on the transferability of valuation results for avoided flooding between catchments of different sizes (Step 9 validation).

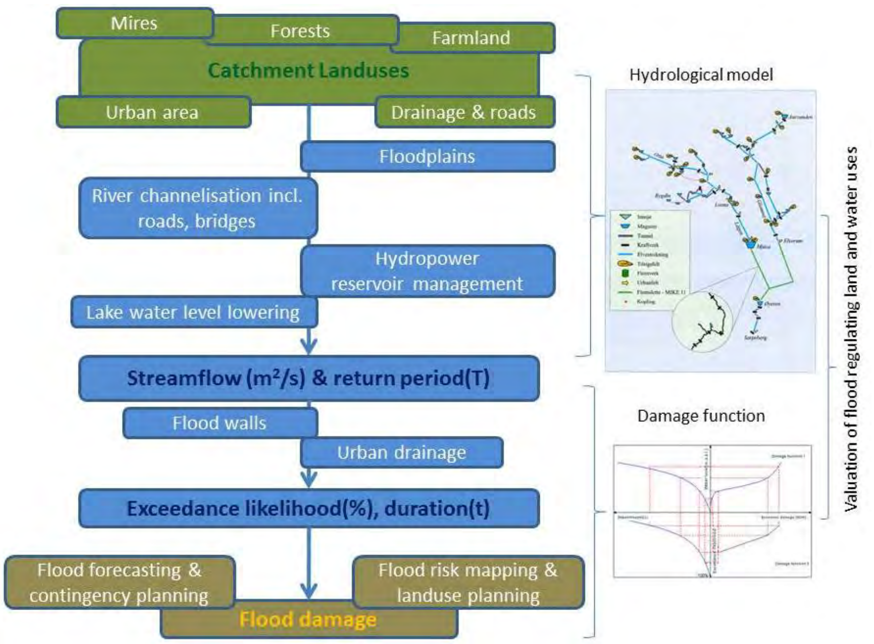
Recent studies of unregulated catchments in the whole of Norway have demonstrated some effects of forest and bogs on summer low flows in winter and summer, depending on the country region of study (Engeland og Hisdal 2009). Percentage of glaciers of total area is one of the most important determinants of low flows in Norway's remaining unregulated catchments. For flood peaks the percentage of lakes and steepness of river courses have been shown to be significant. Spatial regression techniques have not uncovered effects of forest or mires on flood peaks in unregulated catchments (Skaugen og Væringstad 2005). Why study only unregulated catchments? – it is easier to control for effects of landcover without considering the myriad different operations of reservoirs in regulated catchments. However, there is a selection bias in this data as unregulated catchments tend to be less populated and have less landcover. They are potentially of less interest for valuation of ES in watersheds.

Most Nordic countries are particular in having a lot of open mountain areas with little vegetation and snow and ice cover in the winter. Compared to tropical countries with many valuation studies on ES of forests and wetlands, Nordic countries have relatively little "biology". However, snow cover has significant interactions with forest cover in what could be called an ecosystem function and might be called an ES. Studies in Sweden have shown that presence of fir trees leads to high "sublimation' of snow – evaporation in cold weather – than other land uses (Lundberg og Koivusalo 2003). This in turn leads to less melt water in the spring. Meltwater can contribute to flooding under some circumstances or available streamflow for hydropower generation in others.

Spatially explicit hydrological modelling makes it evident that flood reduction as an ES requires seasonal, catchment and event specific knowledge. It also raises the question of what types of landcover are classed as ecosystems or not. What types of landcover can be managed or not also becomes a question relevant for policy and by extension for valuation. The percentage of lakes, glaciers and steepness of rivers are all significant determinants of streamflow and they are all characteristics of ecosystems. But in what situations are they policy relevant indicators of ES? The question suggests superficially that only biological "components' of ecosystems are adequate indicators of ES? However, from an economic point of view, the relevant components of ecosystems are those that can be modified by human beings and are subject to public and private decision-making. For example, glaciers are also landcover, not modified by land management (except in ski resorts in the Swiss Alps!), but are modified by anthropogenic climate change. In other words, what is a relevant indicator of ES also depends on the scale at which geographical policy is being assessed.

What kind of data is needed to predict flood reduction services of ecosystems at the watershed scale? Figure 9 summarises different types of information we have discussed needed to determine flood damage as a function of catchment land uses.

Figure 9: Factors determining economic value of flood reduction service in a watershed



Flood reduction services are provided by combinations of human modification of watershed hydrology, preventive and avoidance measures.

4.2.3 Steps 4–7: Identification of goods and services, beneficiaries and economic values

A number of indirect measures of flood reduction benefits are relatively easy to gather data on. Economic estimates of flood damage are available based on insurance claims, as will be discussed below. Insurance covers residual risk after a number of costly actions have been taken in the watershed. Cost estimates can be obtained for avoided local flood contingency/emergency measures during flood episodes. Costs of flood walls, and of upgrading urban drainage systems are partial and indirect estimates of the benefits of avoiding flood damage. Reservoir management to reduce flooding or address environmental concerns has opportunity costs in terms of foregone income from power generation. Flood walls and river channelization in the upper catchment also incur costs, affect ecosystems locally, and may have local effects on flooding. Economic valuation of the flood reduction services of land-use should ideally add the prevention, mitigation and avoidance costs of these flood management measures to the estimates of avoided flood damages using a damage function (Figure 9). In practice a hydrological model integrating the effects of different measures on flooding is needed in order to avoid double counting.

An important step in avoiding double counting in economic valuation is a clear identification of beneficiaries and losers of different changes in streamflow. ES casts a new light on the term “user interests” which has been familiar in Nordic water resource management.[24] ES encompasses a specification of the “classical” understanding of user interests, not only to the water body itself, but now also to conflicts of interest with upstream users of ecosystems. This means that “ES” will be defined differently according to whose interests are at stake. Unique classifications of ES of regulating services – such as flood reduction – and supporting services – such as habitat conservation – are particularly difficult because of upstream-downstream, on-site and off-site conflicts of interest.

If we are a stakeholder concerned with nature conservation, we want economic valuation to demonstrate which human interventions in nature reduce natural flooding processes, even while reducing flood risk. If we represent a business interest such as hydropower, we wish economic valuation do demonstrate how reservoir regulation contributes to flood prevention, while generating carbon free electricity. If we represent forest owners we wish to demonstrate how forestry practices can reduce peaks in run-off, while storing carbon. If we are recreational fishermen, we may wish economic valuation to demonstrate the importance of vegetation cover and glaciers to stable low flows during summer.

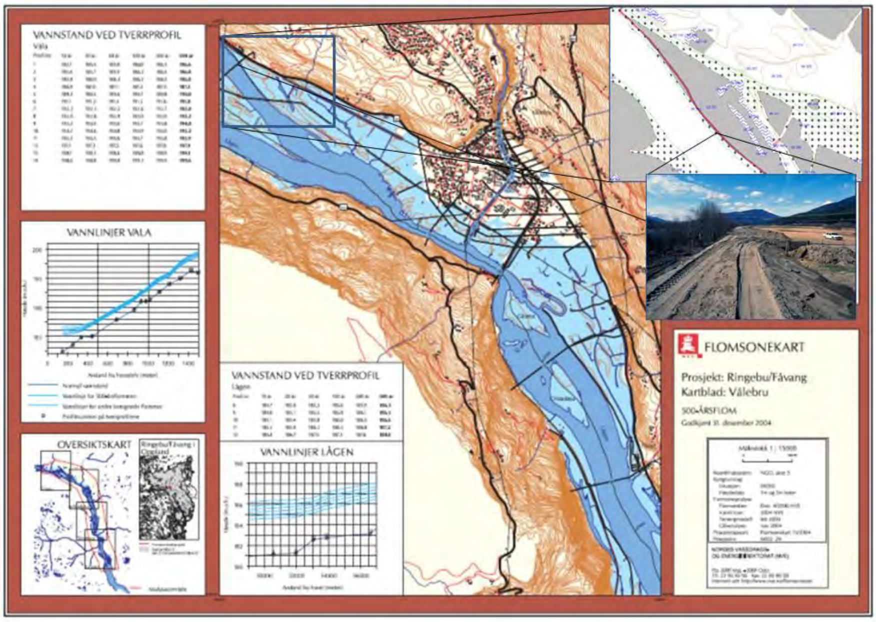
Flooding is a statistically defined concept as we have seen above. But in economic terms it depends on perspective and context. Land and water uses that provide benefits to some interests downstream provide disservices to others. Organisms and water users may also have adapted to and even depend on more or less natural flooding cycles. Whether we are dealing with an “ES” or an “ecosystem cost or disservice” imposed by upstream /off-site interests depends on “who, where and what is being done and done to, and the rights they have to do it’. As we will see from the following three cases some interests prefer low stream flows, other high stream flows during given times of the year.

4.2.4 Case 1 Balancing flood prevention versus wetland conservation in the upper river basin

Barton and Dervo (2009) demonstrate the use of flood damage functions in the context of benefit- cost analysis and multiple criteria analysis of flood protection measures.

[24] See e.g. http://www.vannportalen.no

Figure 10: Flood risk map for Ringebu municipality for a 500 year flood



Source: adapted from www.nve.no. Inlay shows a red line representing the Skarvvollene flood prevention wall constructed between the Lågen River and river flats, depicted in the inlayed photo.

They used available data from the Skarvvollene flood protection works on river flats along the Lågen River in Ringebu Municipality. The report also evaluates the current benefit-cost analysis guidelines for flood protection works used by the Norwegian Water Resources and Energy Directorate in the context of the EU Water Framework Directive (WFD). Barton and Dervo (2009) argue that multiple criteria analysis (MCA) can be employed as a complement to benefit-cost analysis in the assessment of "disproportionate costs" under the WFD. MCA is particularly useful in evaluating trade-offs between priced and non-priced hydro-morphological impacts of flood mitigation projects. It is also a framework for documenting both expert and local opinion on non-priced impacts and their relative values.

Table 9 shows a comparison of the costs of flood wall construction and maintenance versus the expected losses of crop value (hay production). Based on market-priced impacts, the flood wall has negative net benefits. Qualitative scoring of impacts in the multi-criteria analysis was based on a subjective evaluation by researchers using official guidelines. This shows that a conflict of interests exists principally between local interests in removing a mosquito nuisance (a regulating disservice) and to a lesser extent protecting farmland (provisioning service), versus conservation of river flat wetland habitat and its biodiversity (supporting service). Impacts of the flood wall on downstream flooding were considered negligible and not included in the analysis. The flood wall was built by the municipality revealing an implicit willingness to pay by local government to avoid the mosquito nuisance to local inhabitants of at least 1.75 million NOK (the difference between the priced impacts and project cost). If local residents valued the loss of habitat at all, this implies that willingness to pay for getting rid of the perceived mosquito nuisance was even higher. With the help of sensitivity analysis of impact weights in a multi-criteria analysis, Barton and Dervo (2009) go on to show that the mosquito nuisance was valued (weighted) at least 4-5 times higher than wetland habitat loss for other biodiversity, tipping local net-benefits in favour of the flood wall. In summary this case of valuation demonstrates that the definition of what constitutes an “ES” in Ringebu municipality depends on the perspective of conflicting multiple local, regional and national interests.

Table 9. A benefit-cost and multi-criteria assessment of floodwalls at Skarvvollene

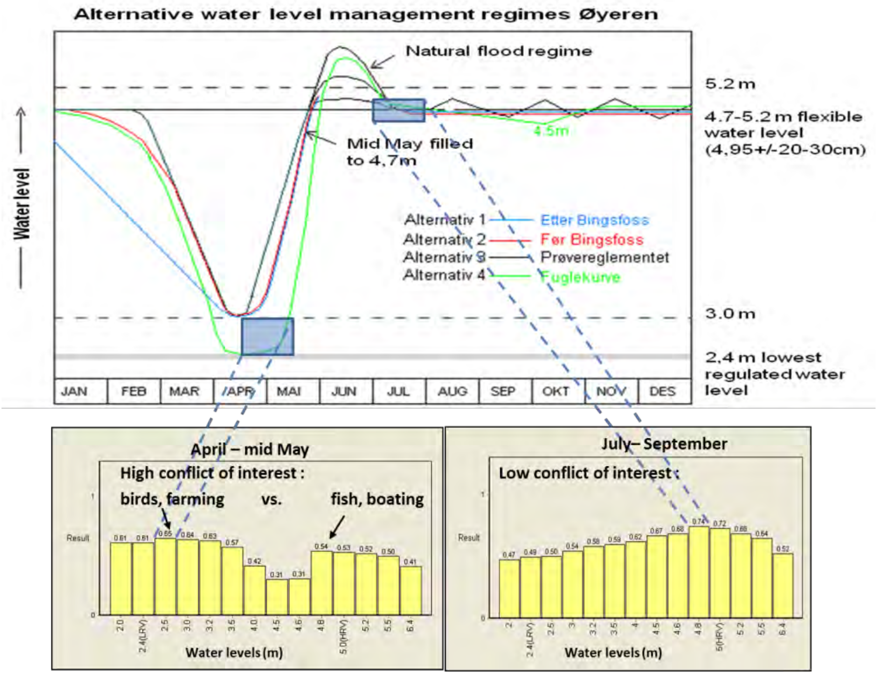
Criteria	Sub-criteria	Unit	Impact	
			Alternative 1 (flood wall)	**Alternative 0 (no flood wall – floodplain)**
Net present value project	Investment and maintenance flood walls	NOK	-2 020 000	0
			-2 800 000	0
NPV priced impacts	Actual and potential crop value	NOK	0	-465 000
Unpriced neutral and positive impacts	Mosquito nuisance	score	+1,5	0
	Avoided erosion and sedimentation of farmland	score	+1	0
	Recreational opportunities	score	+0,5	0
	Development of urban or commercial land	score	0	0
Unpriced negative impacts	Rare species	Red list species	4 species	0
	Rare nature types	% reduction of area in municipality	-30%	0
	Spawning area for fish	score	-0,5	0
	Additional risk of increased erosion	score	-1	0
	Risk loss of bird habitat	score	14/18	6/18

Source: adapted from Barton and Dervo (2009). NOK estimates are in net present values with a 40 year time horizon and 4% discount rate, and flood prevention of flood prevention with a 10 year return period. Qualitative scoring of impacts by Barton and Dervo based on NVE (2001) guidelines: +/- 0,5–1,5 small to large local importance; +/-2,0–2,5 small-large regional importance; +/-3 national importance; NVE. 2001. “Brukerveileder for Nytte- kostnadsanalyse av sikringstiltak i vassdrag, Vassdragsavdelingen, seksjon for vassdragsteknikk. NVE september 2000. Oppdatert juni 2001.”

4.2.5 Case 2 Balancing environmental flows versus flood prevention in the mid-river basin

Using a “pressure-impact multi-criteria environmental flow analysis” (PIMCEFA), Barton et al. (2010) evaluate the impacts of water levels in the Øyeren Lake and wetlands on different user interests, including trade-offs with opportunity costs to hydropower up and downstream of the Øyeren. The study looked particularly at water level recommendations provided by an earlier study for low flows in April–May versus high flows in July–September (upper panel Figure 11).

Figure 11: Comparison of alternative regimes with PIMCEFA multi-criteria optimal water level.



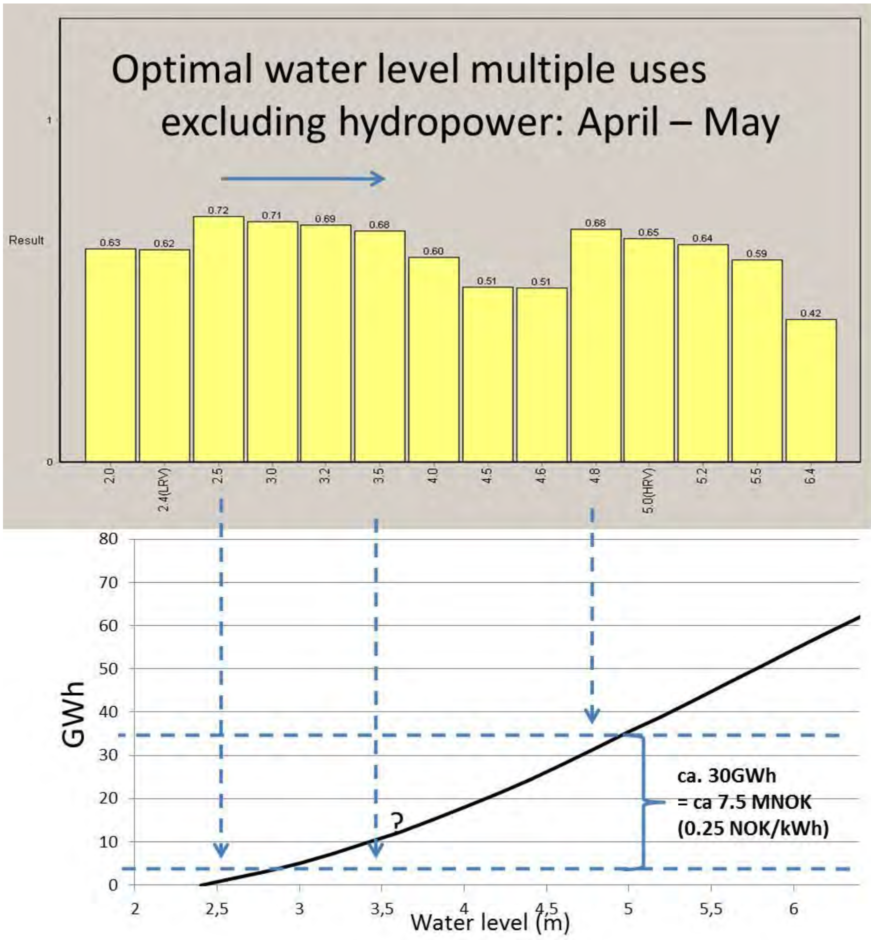
Aggregate weighted scores of alternative water levels for April–May and July–September as determined in DEFINITE (lower panels). The optimal water levels determined independently using PIMCEFA coincide with most of the scenarios identified in Berge et al. (2002) in Barton et al. 2010.

A multi-criteria impact matrix was evaluated by thematic experts for impacts on indicator species of wetland birds, fish, as well as recreational boating and farming on the banks of the lake. Pressure-impact curves linking water levels to impacts on these interests were developed by experts for the two periods and compiled in a multi-criteria analysis (MCA) software. MCA provided a normalised score (vertical axis) for different water levels (horizontal axis) for the two periods (lower panel Figure 11). These scores were then compared with calculated opportunity costs to hydropower production of maintaining different water levels in these two periods.

The analysis shows high water levels around 4.8 meters being desirable for all water user interests in July–September (right lower panel). In April–mid May wetland birds and farming have optimal water levels around 2.5 meters, while boating and wetland fish have optimal water levels of 4.8 meters or more. High or low water levels depend on a combination of hydropower regulation and springtime run-off from a combination of land-uses in the river basin. Ecosystem services of lower run-off and water levels for wading wetland birds and farmers (who need access to fields in the flood plain for ploughing) are disservices to spawning wetland fish and recreational boating (who want to put their boats on the water). Different effects of land-use serve different user interests at different places and times.

Water levels are regulated by hydropower concessions to a large extent. The ES of maintaining water levels at 4.8 meters for fishing and boating would have an estimated power loss of around 30 GWh, at an opportunity cost of near 7.5 million NOK every spring. Concession requirements of maintaining water level around 3.0 meters – favouring farming and wetland bird habitat – has meant a cost of around 5 GWh lost production relative to a situation with no regulation.

Figure 12: Estimation of hydropower production at different water levels and comparison with optimal water levels for other users



Source: Barton, Berge and Janssen (2010).

Optimal water level scores shown in the upper panel were evaluated in DEFINITE not including hydropower (all other interests). The higher the water level that needs to be maintained in April–May, the higher the generation losses to hydropower. High or low water levels depend on a combination of hydropower regulation and run-off from a combination of landuses in the river basin.

How would we value the service of higher water levels for boating and fish habitat? A cost-based valuation estimate of around 7.5 million NOK would be an overvaluation for several reasons:

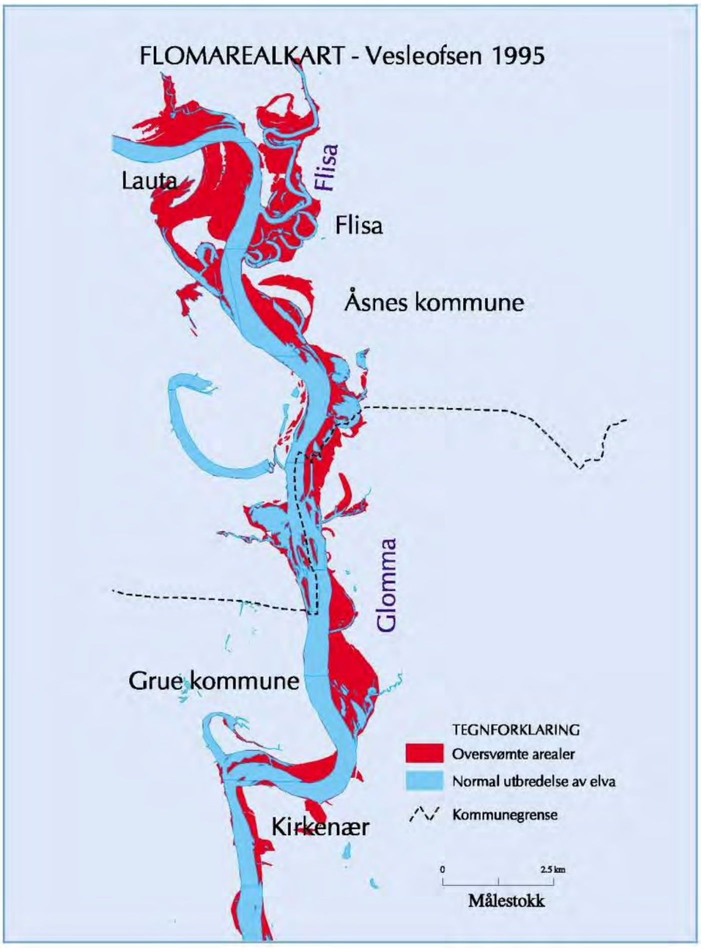
- Based on revealed preferences of authorities, concessionary rights have been set at lower water levels, prioritising farming and wetland bird habitat, and implicitly placing little or no value on recreational boating. It could also be argued that valuation of the potential service to boating and fish habitat of higher water levels should be conducted using hydropowers concessionary rights to a water level at 3.0m as a baseline, rather than a situation with no reservoir regulation <2.5m. Boaters and fish conservationists would have to compensate the hydropower company with the money equivalent of roughly 25 GWh every year for the hydropower company to be willing to consider a water level of 4.8 meters. The decision-relevant value of the "water level service" depends on an interpretation of rights
- Maintaining low water levels in spring happens to be a flood prevention precaution which coincides with hydropower interests. Higher water levels in spring may mean higher risk of flood damage. The total social value of higher water levels may be negative, even though it is positive for boating specifically

In this case all the "action" or "service" is being provided by hydropower reservoir regulation in normal years. It is difficult to identify the contribution of ecosystems upstream to water levels, except when run-off exceeds the regulating capacity of reservoirs.

Case 3. Valuing flood damage mitigation downstream

A final example of valuation of flood reduction services is drawn from the Hydra project. Sælthun et al. (2000) used damage claims for buildings and infrastructure in the municipalities of Åsnes and Grue (Figure 13) to the Norwegian Natural Perils Pool after the Vesleofsen flood of 1995 and a smaller flood of 1966.

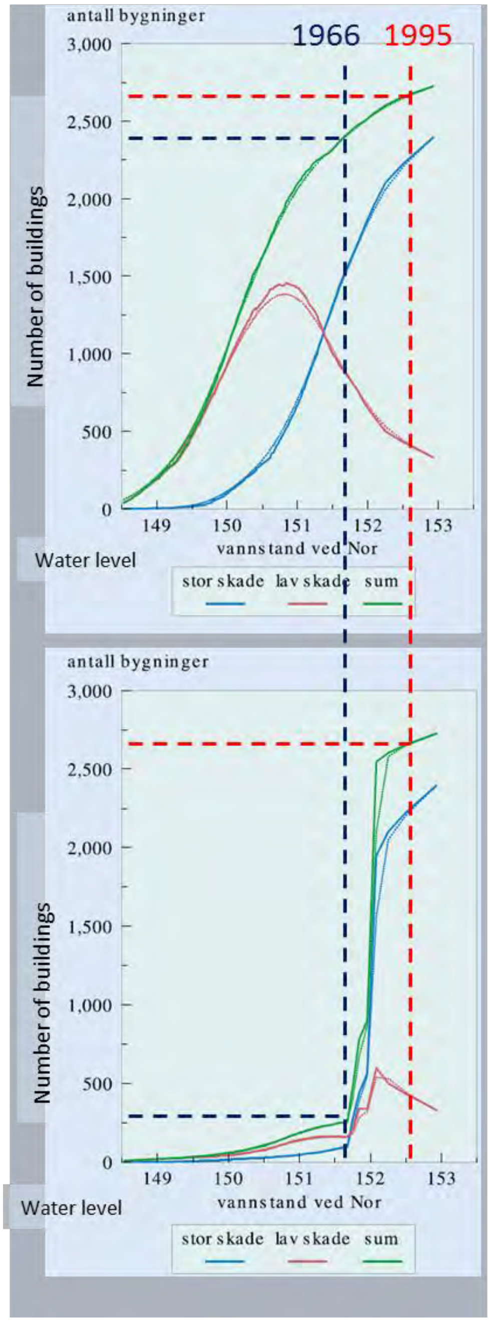
Figur 13: Flood area map for Åsnes and Grue on the Glomma River



Source: Sælthun et al. 2000, Hydra project.

They modelled damages of different water levels with flood prevention walls versus without flood prevention (Figure 14). Flood prevention walls up to a water level of about 151.5 meters above the watermark, reduces the area under the damage function. Recall that the area under the damage function represents the reduction in expected economic damage over different flood return periods smaller than the 1995 flood. Expected flood damages without flood prevention walls were around 50 million NOK. With flood prevention walls designed to protect against a 100-year flood, damages were estimated at roughly 5 million NOK. Economic benefits of flood walls of this kind are around 45 million NOK. If the costs of flood prevention walls were less than this amount they would be economically viable.

Figure 14: No. of buildings damaged on the vertical axis against different water levels on the horizontal axis, with flood prevention walls (lower panel) versus without flood prevention (upper panel)



Upstream land-use measures with equivalent flood reduction impacts have this potential "ES" value, but as mentioned earlier the challenges lies in linking the damage function to land-use and run-off.

A further challenge is taking into account how people react to floods in flood prone areas. Near Kirkenær village in Grue municipality the Vesleofsen flood exceeded the design capacity of flood walls. Expected damages of maximum water levels were around 900 million NOK. Actual flood damage was around 140 million NOK due to temporary emergency measures erected on top of existing flood walls (Sælthun et al. 2000).

In valuing flood damage one must also account for adaptation and avoidance behaviour which reduces flood damage (at some smaller cost). Flood contingency planning reduces the costs of emergency measures and increases their effectiveness, for example using flood zone mapping. In other words, valuation of ES of reducing flood risk based on a mechanical use of damage functions may over-estimate the economic value of flood walls, and by extension of ecosystem services. The correct valuation estimate would adjust the value of avoided flood damages by subtracting the cost of flood contingency planning, prevention, avoidance and mitigation measures (see Figure 9 above).

4.2.6 Step 8: Value aggregation – demonstrating value

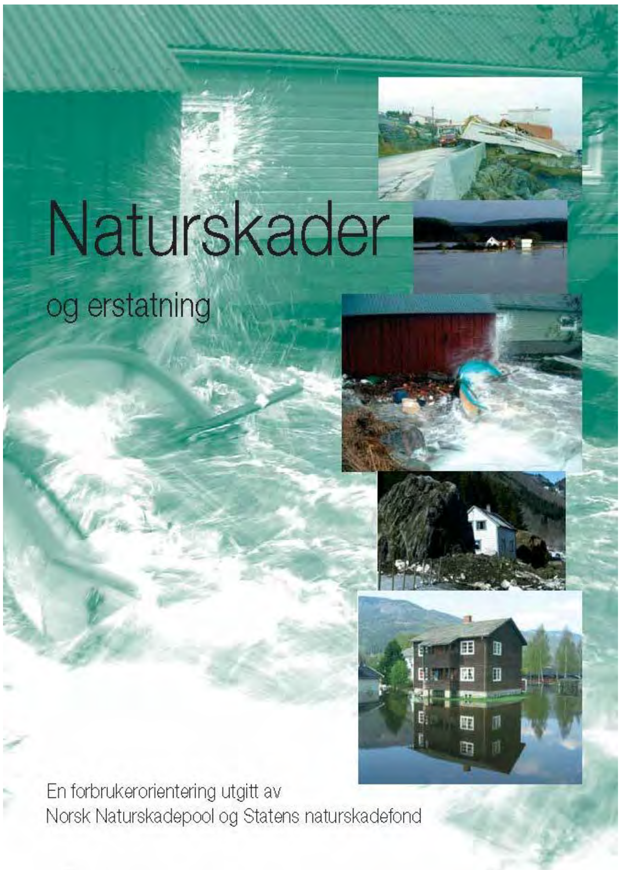
How many people, households or user interests are affected by flood prevention measures? What is the total economic value of flood reduction services to society, whether provided by management of ecosystems or technical measures?

The Norwegian Natural Perils Pool (Norsk naturskadepool) keeps statistics of annual flood damage claims. The Vesleofsen flood in South Eastern Norway in 1995 caused 6900 damage claims for a total 940 million NOK. As of October 2011 more than 3400 flood damage cases had been presented with claims of 389 millioner kroner to the Norwegian Natural Perils Pool (Source: Norsk Naturskadepool). Starting with this aggregate figure we need to ask how much could have been avoided with better land-use management in the watershed, technical measures and flood contingency planning.

It was outside the scope of this study to determine whether similar figures exist for Sweden, Finland, Denmark and Iceland.[25]

[25] As part of the UK National Ecosystem Assessment (UK NEA) it was estimated that 5 million properties are exposed to a flooding likelihood of 0,5%–1,3% annually (Morris og Camino 2011). Flooding causes annual damages in the order of £ 1,4 billion per year. An additional £ 1 billion is spent on flood prevention measures. Morris and Camino admit that these aggregate figures do not represent the potential value of flood reduction services of ecosystem restoration and management.

Figure 15: Norway's flood damages claim manual



Until October 2011 more than 3400 flood damage cases had been presented with claims of 389 millioner kroner. How much of this damage would have been avoided with better landuse management in the watershed?.

Source: Norsk Naturskadepool.

4.2.7 Step 9.1: Validation of valuation assumptions and estimates

Value transfer is discussed in the introduction. Economists must also be weary of transferring damage functions, which may be specific to ecosystem functions / biophysical processes in space, seasonally and temporally (Table 10). Infrastructure costs are generally transferable, whereas damage functions for buildings are partially transferable. For agriculture, damages are locaction, season and annually specific and not transferable according to Hydra (2000) findings. Damage costs functions must be adjusted over time for effects of adaptation to flood risk based on e.g. flood risk maps, as these become integrated into municipal land use regulations.

Table 10. Transferability of damage functions

Damage function	Spatial transferability	Seasonal transferability	Temporal transferability
Buildings	Partial	Yes	Partial
Farming	No	No	No
Roads, railway	Yes	Yes	Yes
Flood prevention	Yes	Yes	Yes

Source: adapted from Hydra (2000).

4.2.8 Step 9.2 Evaluation of demonstration and policy relevance

Aggregation of values to river basin level or higher, as here, may have an awareness raising or "demonstration" effect on policy. We have tried to argue with three site specific cases tied to land and water use decisions, that aggregation is sensitive to the geographical scale of the analysis and runs the risk of double counting benefits unless trade-offs/conflicts between user interests are properly modelled. With the examples, we have tried to demonstrate multiple trade-offs between provisioning, cultural and supporting ES due to the regulation of wetland ecosystem functions (indicated by streamflow and water level). We have shown that the value of this regulating service is (often) indirectly derived from other services. As such the value of regulating services of flood reduction depends on valuation of provisioning and cultural services that can be directly valued using market prices. Local trade-offs and interdependencies between ES mean that they are generally non-additive for a particular wetland or catchment land-use. The practice of citing Total Economic Values (TEV) for specific ecosystems or biomes – exemplified by amongst others the TEEB "Ecological and Economic Foundations" study (Kumar 2010) should therefore be treated with much caution as a guide to policy.

4.3 Run-off and pollution regulation

4.3.1 Valuation of improved water quality

In this section we will give a brief description of a case study valuing improved water quality in Glomma River Basin. In the next sections we will discuss which ES that were actually valued and if values for other ES related to run-off and polluting regulation should and could be added to these values.

4.3.2 Step 1: Policy scenarios as basis for valuation

Glomma river basin and Morsa water area in particular has been in the focus for several years due to the heavy pollution of the water and the many user interests in the area. Thus, for several years, even before the

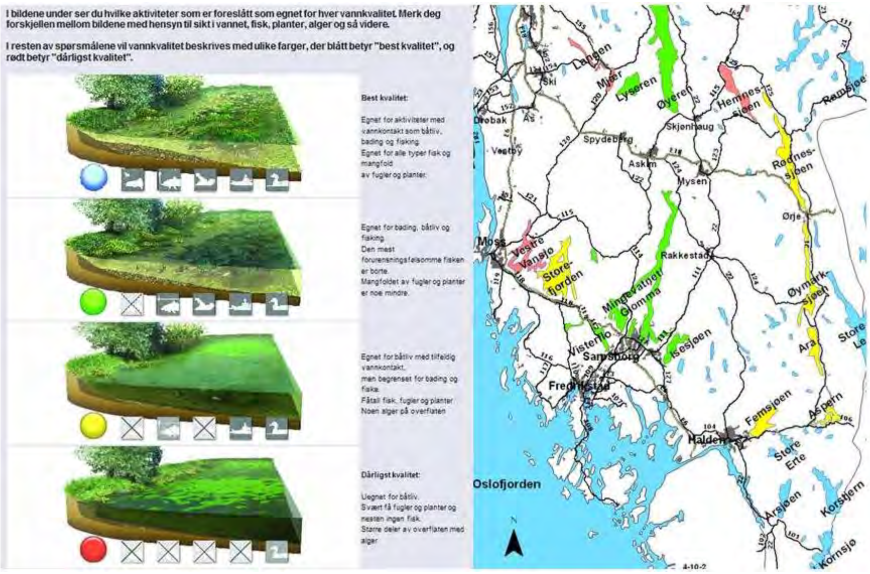
WFD was implemented, there have been efforts to reduce the eutrophication in the water area. However, the scenario without WFD implementation would be uncertain.

Since WFD is implemented in Norway, the scenario most relevant to value is that the water area Morsa (and water region Glomma) reaches the goal of (at least) good ecological status.

4.3.3 Steps 2–3: Definition of measures and identification of environmental change

In the Management plan for water region Glomma/Indre Oslofjord and the appendices with "programs of measures", several measures are identified, and (to a certain degree) costs and effects and cost-effectiveness of the measures are estimated/calculated/assessed. Measures related to agriculture, and household wastewater are the most important. The management plan stresses that the estimated costs and effects are uncertain, but the plan concludes that it will be possible to attain good ecological status in Morsa and the entire Glomma water region with a few exceptions.

Figure 16: Water quality ladder depicting water body status used in CV scenario, and portrayal of current "status quo" water body status in a map tool in the web-survey of the Morsa, lower Glomma and Halden catchments. Green in the water quality ladder represents "good ecological status"



Source: Barton et al. 2009 and Hime and Bateman 2008.

With the WFD the scenario relevant to value and hence the environmental change to be valued is the change from "present status" to "good ecological status". It is also relevant to value the change from present and "good ecological status", respectively, to "very good ecological status".

However, with the last 30 years efforts which have not given more than "moderate ecological status" in Morsa, it may seem unrealistic to achieve more than good ecological status in the foreseen future. Hence, the environmental change most relevant is from "present" to "good (Figure 16).

4.3.4 Steps 4–6: Identification of goods and services, beneficiaries and economic values

According to Barton et al. (2009) a total of 160 952 households live within the 27 municipalities included in the study area. Urban areas include the coastal towns of Moss, Sarpsborg, Fredrikstad and Halden. The population is located mainly along the coast and the outskirts of Oslo. In the context of willingness to pay (WTP) for lake recreation, households constitute the main water user.

All municipal water supply in the study area is from surface water. Storefjorden in the Morsa catchment is an example of a source of intermunicipal supply. A description of "pressures" in the two catchments shows which sectors are users of lakes as sinks. In the Haldenvassdraget water region, agriculture is the most important source of water pollution in one third of the lake area. Dispersed and municipal sewage discharge make up the other pollution pressures with the widest influence on lake surface area.

In the Morsa water region agriculture is a significant pollution pressure on almost the entire lake surface area. Dispersed and municipal discharges constitute a moderate pressure in about half of the lake area.

4.3.5 Step 7: Value elicitation/ demonstration

Watercourse Morsa has been valued in three different valuation studies during the last 20 years; Magnussen (1992) conducted a CV study of improved water quality due to reduced eutrophication in this water course separately and combined with the other watercourses in county Østfold in 1992. In 1995 water course Morsa was valued again using CV with discrete choice questions in order to value improved water quality and test for benefit transfer validity between this and a similarly polluted watercourse (Orrevassdraget) in southwestern Norway (Magnussen et al. 1995; Bergland et al. 1995, 2000). Both these valuation studies valued a bundle of benefits resulting from an improvement in water quality measured according to the four or five water quality classes respectively, used by the water management authorities at the time. Both studies described which benefits would result from the improvement, explained verbally and by the aid of pictures and symbols for drinking water quality, fishing possibilities, swimming and boating, and use values as well as non-use values were estimated.

Morsa was valued again by Barton et al. (2009) and this time the water quality improvements valued were connected to the water quality classes according to the Water Framework Directive (WFD). Since fulfilment of the goals in WFD is the present goals for Norwegian water management, the latter study will be used to give estimates of values for watershed ecosystem services in the following sections.

The questionnaire used for valuation of Morsa in Barton et al. (2009) was composed of questions that were common to all the water quality case studies in the Aquamoney project and some questions specific to the Norwegian case study. A number of illustrations used in the survey were common to all the water quality case studies in Aquamoney (Hime and Bateman 2008) (see also the Odense case in chapter 5).

An internet survey was conducted on 1113 households in Østfold and southern municipalities of Akershus county in the summer of 2008. The survey focused on households' recreational use of water bodies and their willingness to pay (WTP) for improvements in lake ecological status. The valuation methods used aimed at capturing recreational values and non-use values. The largest lakes in Østfold in three different catchments (Morsa, Glomma and Halden) were considered.

Two pilot river basins, Haldenvassdraget and Vansjø/Hobølvassdraget in the Glomma Water Region were selected as a focus of the case study. Additionally, WTP for improvements in lakes and coastal waters bordering these river basins, in order to assess the relative importance of the pilot river basins to water bodies that may be substitute sites for recreation.

The summary in Barton et al. (2009) reports that WTP for improvements in water quality from the situation in 2008 ("at present" at the time of the survey) and to "good ecological status" is estimated to NOK 1070–2000 per household per year for the lakes Vansjø and Storefjorden, depending on the valuation method used.

The report aims at measuring the distance decay and spatial extent of willingness to pay. Barton et al. find that WTP drops by 25–72 NOK/kilometre depending on which valuation method was used for improvements from current status to good ecological status or better. However, this distance decay effect seems to be driven by zero responses, and should be interpreted and used with care. For the lakes in lower Morsa catchment this means that people living 30–60 km away from the lake are willing to pay something for improved water quality in these lakes.

Barton et al. find that people's WTP depends on the number of lakes improved (one versus two lakes) only in special cases. Recreational values of lakes seem to predominate over non-use values. The households' WTP is sensitive to the size of the improvement in water quality for some lakes. There are large differences in WTP for lakes in adjacent catchments, such as Morsa, Glomma and Halden.

4.3.6 Step 8: Value aggregation – demonstrating value

Barton et al. aggregate WTP for improvements in the lakes Vestre Vansjø and Storefjorden and find that aggregate WTP based on contingent valuation results is around 30 million NOK per year, while aggregate WTP based on choice experiments is around 113 million NOK per year. (This comparison is based on not excluding the percentage of households protesting to the CV-question because there is no protest option in the choice experiment).

They suggest a conservative estimate for benefit-cost analysis use might exclude the households who protest the WTP questions. Omitting protesters, the aggregated WTP is estimated to approximately 21 million NOK per year for the improvement, using the CV scenario.

4.3.7 Which ES are valued and should others be added?

The aim of this report is to give estimates for values of ES in watersheds. However, since there are no studies valuing watershed ES directly, we had to use valuation studies which do not discuss the environmental improvement valued in terms of ES.

In order to state valuation estimates for ES from watersheds, we therefore have to consider which ESs are actually valued – and which are not valued, and could/should be added.

If we look at the list of ES in the TEEB report (Kumar 2010), we can see that several of the watershed related ES are totally or partly related to run-off and pollution regulation. That is the case for provision of food, since polluted water can make the habitat for fish and other species unsuitable (for instance is sea trout dependent on water of a certain quality) or the food can be unsuitable for human consumption (for instance due to hazardous substances in shellfish). Provision of water for drinking for humans and husbandry, agricultural purposes and industrial use is an ES whose quantity and quality is dependent on a certain water quality. Other provisional services, like provision of raw materials, genetic resources, medicinal resources and ornamental resources may depend on water quality standard as well – for instance genetic resources may be lost if species get extinct due to unsuitable living conditions. However, these last mentioned ES may be hard to specify for one or two lakes – if there are not particular vulnerable species or habitats present.

Of the regulating services, particularly water purification is highly related to run-off and pollution regulation. (These services may be nested concepts due to the wide definition of water purification). But also other regulating services may be related.

Of habitat services both lifecycle maintenance (especially nursery services) and maintenance of genetic diversity (gene pool protection) is related.

And finally, the cultural services; first and foremost opportunities for recreation and tourism, but also inspiration for culture art and design are connected to water quality status.

The case valuation study in Glomma River Basin focused on the recreational values; most of the explanatory text, pictures and symbols gave information about improved recreational use of the water bodies. Thus, the ES "opportunities for recreation" is definitely covered by the value estimates in the case study. Further, a certain non-use value was expressed in the valuation study – illustrated by preserving "wildlife". This was both a use value (bird watching) and non-use value. This means that a non-use value, sometimes called "the testament of the water (shed)" is covered through the valuation case study. Still, it is not given from the valuation case study which parts of the non-use ES which are really "covered" in the non-use values derived. And it is an open question whether more of this non-use part of ES values should be added to the derived value estimate.

Another open question is whether the value of provisioning of water for human drinking, and for agricultural use should be added. Residents in the urban areas surrounding river basin Vansjø and Morsavassdraget, get their drinking water from surface water in the river basin. The explanations, pictures and symbols did not mention drinking water use and drinking water quality, probably because the water purification equipment for drinking water which is currently in place in the area already gives the residents drinking water of acceptable quality.

The water purification service is also an important service. In case of the lakes and rivers in River Basin Glomma, the situation is that the water's natural purification capacity is exceeded, with the resulting reduced water quality and reduced quality of other ES.

This brief overview of ES illustrates some important points regarding practical use of the ES concept – and particularly use of "old" valuation studies for environmental goods in order to derive value estimates for ES related to the good in question.

The ES concept is a potentially very useful concept because it focuses on and highlights that "ordinary" ecosystems in watersheds supply us (humans) with a variety of goods and services which are important for us in our everyday life. However, this is contrasted with some other concepts and frames which are important for instance for watershed management. The water framework directive which sets the frames within which to manage our water resources in EU (and Norway and Iceland) focuses its goal on ecological status – which at the outset is not related to people's use or benefits derived from these water bodies at all. Thus the goals for good ecological status are decided without any considerations of people's goods or benefits from the water bodies. On the other hand, the WFD also stresses that the users should be consulted and included for instance in the economic analysis.

We have seen that the valuation study designed in order to value the benefits from improved water quality according to WFD, captures only one or some ES, and further that it is a bit uncertain exactly which and how large parts of which ES are actually valued.

The first estimates of values for ES often attempted to value the total ("the stock") of one or several ES. For instance, one could attempt to put a value on the amount of drinking water supplied from lake Storesjøen and Vansjø, for instance by considering the amount of drinking water supplied from the water bodies (in m^3) and the value of drinking water per m^3. The amounts can easily be calculated and the value could be estimated either as the amount of money people actually pay per m^3 for drinking water in the area, or by the amount of money people pay for bottled drinking water.

However, the methods economists use in order to value environmental goods are developed to measure the value of a marginal change in the quality or quantity of the environmental good. This means that if we wanted to value and add the value of drinking water in River Basin Glomma, we need to consider what the change in water quality will mean in terms of improved drinking water quality. And this brings up an interesting question: in our case study the water course is a drinking water source and thus the water quality for drinking water is relevant. However, in this case; due to long-term pollution in the water course; strict purification equipment is installed, and currently a shift in water quality will not imply any change in drinking water quality facing the consumers. How should we then treat the value of improved lake water quality? In the long term improved lake water quality could reduce the need for purification of raw water for drinking purposes, and thus in the long term this could be valued. However, even though we cannot easily put a monetary value on this, most people would agree that it has a value that we have water courses with water quality which is "more suitable" or "closer" to drinking water quality. This stresses the need for valuation of ES (also from watersheds) to include both monetary values and other assessments of values.

The water purification service of the lake as a sink is also interesting. Definitely, this service is very important, and from a point of "no pollution" this service "saves us" from spending money on measures to reduce pollution – which has a monetary value of "reduced costs to polluting reducing measures". In economics, this is illustrated with a figure when the "damage cost function" (in most cases) does not necessarily result in damage costs for the first unit of pollution, but when the "cleaning capacity" of the lake as a sink is exceeded. This point where the cleaning capacity is exceeded will vary between recipients, and with respect to non-use values, "testament of the watershed" etc it may be rather low. As soon as this cleaning capacity of the lake is exceeded, however, increasing the pollution means that other goods and services are reduced in quality and perhaps quantity – and therefore also the value. This illustrates the point that several of the listed ES are linked, and that if we want to increase the values taken out for instance of food or drinking water or use of purification functions, this may influence other goods and services and reduce their value. Therefore we have to consider trade-offs between different ecosystem goods and services.

In our case study area, a change in water quality from present to good ecological status will not result in any "free purification capacity".

5. Demonstration watershed II: Odense River Basin

Chapter 5 provides a summary of a valuation study in the Odense River Basin, looking at the improvement in the ecological status of river water quality. A detailed look at the choice experiment valuation methodology at the river basin scale serves as a source of discussion of methodological challenges and solutions for valuation methods applied to different watershed services across different geographical scales.

5.1 Watershed description

5.1.1 Location of the Odense River

Odense River is located in Denmark, at the island Fyn (Funen), between Jylland and Sjælland. The catchment area of Odense River basin is 1046 km^2. There are about 246 000 inhabitants in the area (Dubgaard et al. 2007). There is one major city, Odense city, with a population of ca. 180 000 located in the catchment area; otherwise the area is mostly rural. Figure 17 gives an overview of the area.

Figure 17: Overview of Odense River Basin (ORB)

5.1.2 Characteristics of Odense River Basin (ORB)

Odense River basin (ORB) consists of 1100 km of open water courses and 2 600 lakes and ponds (> 100 m^2). 21 lakes occupy an area of more than 3 ha each. These water bodies are subject to varying pressures to their environmental state (Hasler et al. 2009).

5.1.3 Land use in the area

71 percent of the area is agricultural land, including approximately 70 000 livestock units (LU) (Fyns amt, undated). There are approximately 1,800 registered farms in ORB. Approximately half the farms are livestock farms, dominated by pigs (59%) and cattle (37%). The livestock density is 0.9 LU/ha (Dubgaard et al. 2007). The dominating crops are cereals (63% of cultivated land), while 10% is grassland.

Urban areas account for approximately 16% of the area, woodland 10% and semi-natural areas (meadows, bogs/fens/swamp forest, dry grassland, lakes and wetlands) for 6%. Artificial drainage has been established on an estimated 55% of the cultivated land in ORB. The semi-natural areas have undergone major physical changes and many of them have disappeared in recent decades. Restoration of these areas is one of the measures to obtain good quality of the water-bodies in the basin (Dubgaard et al. 2007).

Approximately 90% of the population in the river basin discharges their wastewater to a municipal wastewater treatment plant. The remaining 10% of the population live outside the towns in areas without access to sewerage.

Dubgaard et al (2007) gives a further characterization of water use in Odense River Basin, which is used in the following description.

Most of the abstraction of water in ORB by human activities is accounted for by waterworks. Abstraction of water by farms and market gardens is more modest. The water requirements of the agricultural sector in Fyn (Funen) are considerable less than in the sandy areas west in Denmark (Dubgaard et al. 2007). In addition to the above mentioned the remaining abstraction is by industrial wells, minor waterworks and institutional wells.

Approximately 38 million m^3 groundwater is abstracted for the drinking water supply in county Funen. Approximately 11 million m^3 is abstracted for industrial purposes, crop irrigation. The total amount of water abstracted corresponds to approximately 25% of the mean summer runoff in the watercourse of county Funen, or more than half of the amount of water that flows in the watercourses in dry summers. A considerable part of the abstracted drinking water is “returned” to the watercourse in the form of retreated wastewater, but not necessarily in the area from where it was abstracted. Within ORB large amounts of groundwater are abstracted, in some catchments amounting to over 50% of the median minimum water flow in the associated watercourses.

5.2 Run-off and pollution regulation

5.2.1 Valuation of improved water quality

In this section we give a brief description of a case study valuing improved water quality in Odense River Basin. In the next section we discuss which ES were actually valued and if values for other ES related to run-off and pollution regulation should and could be added to these values.

Step 1: Policy scenarios as basis for valuation

In the proposed management plan for river basin Odense, several measures are identified, and costs and effects and cost-effectiveness of the measures are assessed. Measures related to agriculture and household wastewater are the most important. The management plan stresses that the estimated costs and effects are uncertain, but the plan concludes that it will be possible to attain good ecological status in ORB. However, this plan is still a proposal and not final, and thus all conclusions must be treated with care.

As noted in previous chapters, with the implementation of WFD in EU, the aim is to achieve at least "good ecological quality" in all water bodies. Thus the scenario, and hence the environmental change, most relevant to value is the change from "present status" to "good ecological status". It will also be of interest to value the change from present and/or from "good ecological status" to "very good ecological status".

Steps 2–3: Definition of measures and identification of environmental change

The environmental change valued is the change from present status (see Figure 18) to good ecological status. In Aquamoney these changes are valued (Hasler et al. 2009). In order to make people understand what this change implies, however, one has to explain in ways that is meaningful to the respondents in the choice experiment survey.

This is often done by the use of symbols for different uses, pictures and explaining texts. In Aquamoney drawings were used together with symbols to illustrate uses of water with differing quality (see water quality ladder in Figure 16):

- Fishing symbol (1): This symbol shows that the water quality is good enough even for very pollution sensitive fish species
- Fishing symbol (2) with another fish: This symbol shows that the water is suitable for fish which are less sensitive to pollution

- Swimming symbol: This symbol shows that the water is suitable for swimming (corresponding to “blue flag”)
- Rowing boat symbol: This symbol shows that the water is suitable for rowing or sailing
- Duck/bird symbol: This symbol shows that the water is suitable as a habitat for birds and that there are good opportunities for seeing birds

The text explains that when the water quality is reduced, the habitats for plants and animals and the possibilities to perform different recreation activities are reduced.

Pictures are used to explain different water qualities, and these pictures are used in order to illustrate the water quality at different stretches of river Odense.

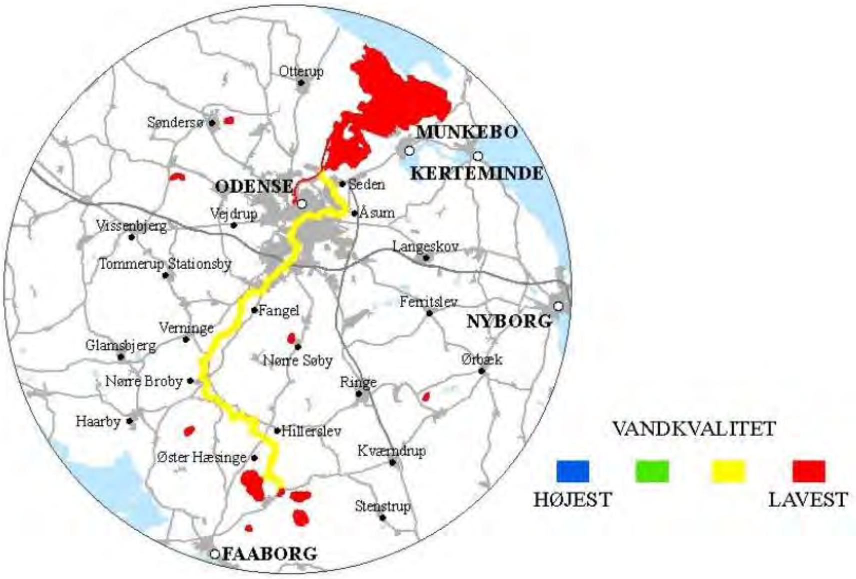
Thereafter the drawings of four water quality levels are shown. The description for each level describes which activities the water quality is suitable for, as described for the respective symbols. And the level of diversity of birds, fish and plants is described for each quality.

The water quality can differ from red (lowest water quality), yellow (next to lowest), green (next to best) and blue (best water quality).

The environmental change is “from the present situation”. In Aquamoney a map showing the present situation for Odense River was shown, and it was explained that the present level is next to lowest (“yellow”) quality. It is explained that water quality may vary a little from place to place in the river for several reasons; for instance discharges from buildings and roads, erosion of nutrients from agriculture, and physical factors etc. in the river. However, generally, Odense River is in the “yellow state”.

In the text leading up to the WTP question, it is said (author’s translation from Danish): “that the environmental authorities suggest improving the water quality in a smaller part of Odense River. These improvements involve costs. These costs will be covered by an extra amount of money, which will be charged as a part of the yearly water bill. If this suggestion is carried out, all users of water and all contributors to the pollution of the river (industry, agriculture and private households) should pay this yearly amount.”

Figure 18: Illustration of water quality in Odense River Basin



Source: Hasler et al. 2009.

Steps 4–6: Identification of goods and services, beneficiaries and economic values

Ecosystem Services – as described, quantified, and valued in the Aquamoney report for ORB (Hasler et at. 2009)

The case study's main objective was to estimate the benefits of improvements of the ecological status of Odense River according to the WFD. The survey focused on households' recreational use of water bodies and their WTP for improvements of the ecological status of Odense River. However, the case study report gives description and assessment of several other ES (goods and services) as well.

Hasler et al. (2009) gives a short characterization of water uses and water users which gives information about identified water goods and services in the ORB.

Households

- Consume drinking water from groundwater in the area
- Discharge wastewater
- Use the water bodies in the basin for recreational purposes

The expenses for consumption of drinking water are calculated in Hasler et al. (2009) on the basis of a distribution of the total production costs in relation to the household sector's share of the consumption. The expenses for drinking water also include a charge to cover groundwater mapping by the counties, of which the households' contribution is expected to be ca. DKK 2 million. Hasler et al. have not calculated expenses for the actual pollution-limiting measures to protect the groundwater, as these measures are part of the current planning of the future groundwater protection initiatives.

There were no available calculations for the amount of waste water discharges for different consumer groups. Hasler et al. therefore estimated the households' expenses for waste water for the same amount as their consumption of water. The expenses for wastewater include a part of the wastewater levy, which is determined by actual discharges of BOD5, N and P.

The levy was estimated to just over DKK 10 million for ORB, and the households pay just under half of this amount. Household expenses for wastewater disposal comprise almost two thirds of the expenses for the actual services to this tax on water consumption of DKK 5 per m^3 which, together with VAT, comprises one third of the total costs. The cost of one m^3 of water is typically DKK 4–6. On average households pay 4000 DKK per year for water consumption and wastewater disposal (Hasler et al. 2009).

Industry/services

- Use drinking water for consumption
- Discharge wastewater
- Fynsværket combined heat and power plant uses fjord water for cooling (the heated water is later disposed into the lower reach of River Odense)
- A small number of enterprises have their own water supply
- A couple of small enterprises have their own wastewater outfall

Additionally to the abovementioned expenses, a minor fee for the protection of groundwater against soil contamination is included. In Hasler et al. a rough estimate has been made for the expenses for remediation contaminated sites that are paid by members of the public. This cost has been ascribed to industry/services, even though a minor share might be defrayed by households.

These expenses are placed in relation to the industry/services production value. This measure is an attempt to determine the proportion of the sector's total production costs comprised by water and wastewater services, and hence how important water use is as a production factor. It should be noted that industry's expenses for complying with various discharge criteria are not included. Such expenses are not calculated in Denmark as they are often process-integrated and it is thus quite arbitrary how much is ascribed to environmental requirements and what is operation-related improvement in production.

The relative expenses for industry alone are somewhat higher since water and wastewater together account for approximately 0.6% of the production value, compared to 0.1% for the service sectors alone.

Agriculture and market gardens

- Water consumption for livestock
- Water consumption for crop irrigation
- Discharges of nutrients

Agriculture is a central sector in the analysis of the existing water use because it accounts for the greatest proportion of nutrient loading of surface water and groundwater in the basin. The environmental pressure is an unintentional side effect of the intensive production. The calculations of agricultural expenses associated with water use include an estimate of the annual expenses associated with pollution-limiting activities. These encompass expenses associated with the implementation of the nationwide Action Plan on the Aquatic Environment II.

Agriculture use of water is a necessary input to both livestock and crop production. The sector primarily uses water abstracted for the public waterworks in livestock production. Field irrigation is primarily based on individual abstraction wells, and accounts for approximately 50% if combined water consumption by agriculture and market gardens in ORB.

In addition, drainage measures have been or are being carried out in the form of the drying-out of wetlands, watercourse regulation, drainage and watercourse maintenance in order to optimize agricultural production and maximize the size of the area suitable for cultivation. At the same time, these measures enhance pressure on the environment due to a reduction in the maturation capacity of the soil. Expenses for these drainage measures are paid for by agriculture with respect to wetland drainage (pump and dyke associations), and by counties and municipalities with respect to watercourse maintenance.

Expenses associated with wastewater are estimated on the basis of just under 2 000 farms in the basin. The latter are assumed to be connected to an emptying scheme for sewage sludge from the individual mechanical treatment facilities, while at the same time paying the wastewater levy of DKK 3.8 per m^3. These expenses each correspond to approximately DKK 1 million per year. The expenses for the pesticide tax have been calculated from the proportion of arable land in ORB relative to that in Funen county as a whole, and the total tax proceeds.

Agriculture's own expenses for their agri-environmental measures account for just below 15% of the sector's total expenses immediately related to water use in ORB. Expenses for environment-related green taxes (tax on pesticides) account for 21% of the total expenses. Relatively, the expenses for water use comprise a higher proportion of agricultural market garden production value than is the case with industry/services. The percentage is still relatively small, though.

The consumptive non-market use values related to the water bodies in the river basin, which are the focus for valuation within the case study of ORB, comprise recreation, both bathing waters and fishing waters. Both the coast and the Odense River have good locations for anglers.

Step 7: Value elicitation/ demonstration

The Aquamoney case study in Odense River Basin gives an economic valuation of the benefits of attaining "good and very good ecological status" in ORB.

We cite Hasler et al.'s (2009) summary of the results of the survey: Willingness to pay per household per year, including sensitivity to scope*:* The sensitivity to scope is tested by asking of the WTP for improvements of the whole river versus one stretch of the river. The stretch is located outside Odense city, and is 15 km out of the total of 60 km. With the contingent valuation (CV) method the mean willingness to pay (WTP) for an improvement in Odense River to good ecological status is estimated for a short improvement (15 km out of Odense River which is approx. 60 km long) to be 323 DKK (43 EURO) per household per year. For the large improvement (the whole river) the equivalent is estimated to be 479 DKK (64 EURO) per household per year.

Hence, households' WTP is sensitive to the magnitude of the improvement, i.e. whether the whole river is improved or only a minor part. However, the internal scope is much stronger than the external, which is tested by introducing the short and the large improvement first in two split samples, and subsequently asking about the long/short improvement. The respondents answering the large improvement first and then the small have a significantly lower WTP for the smallest improvement, while the difference is not significant for the other part of the sample receiving the smallest improvement first. This is a classic test of the validity of the valuation method, i.e. that people's WTP is sensitive to the size or the quality of the good valued.

With the CE method the mean WTP for obtaining "good" valuing the river in three stretches is estimated to be 1053 DKK (141 EURO) per household per year for the whole river (329 + 467 + 257 DKK). To obtain "very good" quality of the water the mean WTP is estimated to be 1430 DKK (192 EURO) per household per year (582 + 545 + 303 DKK). Valuing the whole river at once gives a mean WTP for "good" status, 430 DKK (58 EURO) per household per year. Estimated WTP is approximately the same for achieving a better water quality (WTP to obtain "very good status" is 423 DKK (57 EURO) per household per year. These results do not show scope sensitivity as the respondents are willing to pay approximately the same amount of money for 1/3 of the river as for the whole river. Lacking WTP sensitivity to scope in terms of the size of an ecosystem could be explained by respondents only using a particular location within the ecosystem, so that the marginal value of protection of ecosystem area not used by the respondent is perceived to be low or zero. Localised ecosystem use can explain lacking scope effects and can also be a challenge for estimates of average WTP per river length or wetland area.

Distance decay and spatial extent of willingness to pay. Estimated with the CVM the WTP drops by 1.53 DKK/kilometre for the short improvement and 2.31 DKK/kilometre for the large improvement. This means that for the short improvement the radius of households affected by the improvement is 144 km, while it is 212 km for the large improvement. The larger distance for the large improvement despite the faster drop, is caused by the higher mean WTP. A summary of WTP results are shown in the table 11 below.

Table 11. The Aquamoney case study in Odense River Basin (ORB) gives an economic valuation of the benefits of attaining good and very good ecological status in ORB. WTP derived using different methods and approaches

	Mean WTP (Euro Mean WTP (Euro) per household per year) per household per year	
	Large improvement (60 km)	Small improvement (15 km)
CV method (Improvement to good ecological status)	64	43
CE method (Improvement to good ecological status)	141 (river in 3 sequences) 58 (river as a whole)	
CE method (Improvement to "very good" ecological status)	192 (river in 3 sequences) 57 (river as a whole)	

(Source: Hasler et al. 2009).

Step 8: Value aggregation – demonstrating value

Total willingness to pay: The total economic value (TEV) is calculated using the estimated distance decay function. Alternatively, aggregation could have been done within the administrative/political region. For the CV, the TEV for the short improvement to obtain good quality, is estimated to be between 138–150 million DKK per year (18–20 million EURO). For the large improvement the TEV is estimated to be between 200–223 million DKK per year (27–30 million EURO). For the CE, the TEV is more than 3 times as large as for the CV to obtain a good quality, 489 million DKK per year (66 million EURO). To obtain a very good quality (using CE) the TEV is 664 million DKK per year (89 million EURO). We do not go into details about the methodological challenges in the methods in general or the studies in particular here(see appendix 1 for some further discussion). However, we note that the results from CE are considerably higher than results from CV. For practical purposes, these estimates may be used as an upper and lower limit for people's WTP, and though the differences are considerable, they still narrow down the possible values of the ES considerably.

5.2.2 Which ES are valued and others be added?

The valuation case study from Odense is very similar to the one from River Basin Haldenvassdraget, and the ecosystem services related to pollution regulation are similar as well. Therefore, the discussion of which ES are included in the valuation estimates is very similar to the one for RB Glomma in section 4.3.2, and is therefore not repeated here. There is one difference, however. Drinking water for human use is supplied from ground water in the ORB area, not from surface water. Therefore the improvements in water quality in the lakes and rivers will not affect the ES "human drinking water" in this case.

In other respects, the river basins have much in common; for instance the water purification capacity is exceeded in ORB as well, and even if the water quality is improved, this will not result in any new capacity for the river basin to receive "new" pollution which could be purified for "free".

5.2.3 Valuation of run-off and polluting regulating ES

The improvements in water quality from present (2008) status to "good" or "very good" ecological status according to the WFD definitions will increase the value of several polluttionrelated ecosystem services. For the Aquamoney case study see Hasler et al. (2009).

The main pollution regulation related ES in Odense River Basin are increased opportunities for recreation, possibly purification services and certainly non-use ES, such as preservation of wildlife and habitats (biodiversity), genetic pool etc.

Of these ES, recreation opportunities and a certain part of non-use ES were valued to (conservatively) at 138–223 mill DKK per year according to Hasler et al. (2009).

The natural purification capacity in watersheds is also important. However, although this ES is certainly of some value, one very quickly comes in a position when "using" this purification capacity must be traded-off with reduced quality of the ecosystem services.

6. Discussion

Chapter 6 discusses the data gaps uncovered in the review of Nordic valuation studies, followed by a discussion different challenges to valuation methodology in the context of ES, challenges to increase the policy relevance of valuation of ES, and to dissemination of "ES" in public debate. This discussion provides additional support for recommendations for further research.

6.1 Gap analysis for watershed services in Nordic countries

Our literature review shows that the watershed ES valued are quite similar across the Nordic countries. The services addressed are mainly provisioning services as food and fresh water supply, regulating services like regulation of water flows and water purification, as well as cultural services like aesthetic information and opportunities for recreation and tourism. Contingent valuation and choice experiments dominate among the methods applied. However the travel cost method has also been used frequently, particularly for valuation of recreational fishing in lakes and rivers. Application of deliberative or participatory valuation approaches and production function/damage functions are more rarely seen. In order to broaden the methodological approaches at hand and our perspectives on values, use of such methods could increase our knowledge of how people value ES. Further, more studies are needed that use production and cost based methods and integrated models as well as a need for more primary benefit-based valuation studies. A larger base of valuation studies, would give us more information about values for specific watersheds and ES, and would enable us to improve value transfer.

Many studies have valued increased recreational benefits due to improved water quality. Examples of further data needs mentioned in the Nordic studies are more accurate studies relating to marginal benefits of reducing nitrogen and phosphorous loads as this would be of great importance for decision-making. Marginal costs for reducing nitrogen and phosphorus loads are more thoroughly described in the literature. More studies are needed to address the benefits of reduced hazardous substances.

The WFD art. 4 states that derogations from attaining the objective of "good ecological status" in water bodies can be given only if the costs of measures are disproportionate. In order to decide whether the costs of measures are disproportionate, one also needs to consider the benefits. The family of ES concepts is a framework for defining the benefits to society of improvements in ecological status. There is a need to apply studies of environmental benefits like those presented in this report in the decision making processes and come up with examples on how this could be used to define disproportionate costs in the WFD.

Using valuation for instrument design such as scaling and targeting incentive levels to particular land-use management practices or develop instruments for cost recovery of water services in the WFD is may be of the areas where valuation studies can give the most concrete input. However, it is also the area which requires the highest level of reliability and accuracy. More studies and good examples are therefore needed.

One of the reasons that there seems to be a knowledge gap in the areas of valuing some of the ES, is the lack of appropriate ES indicators and the need for more scientific knowledge and dose-response relations.

Most of the former valuation studies value bundles of goods and services resulting from (changes in) specific watersheds or water bodies rather than individually specified ES, with the exception of studies carried out for valuation of recreational fishing in particular. Therefore it is in general a need to carry out more primary studies for all ES with the more specific aim to value the individually specified services (or a bundle of specified services).

6.2 Methodological challenges

The literature review of Nordic valuation studies and the detailed discussion of valuation estimates from two demonstration case study watersheds generated several methodological issues which we discuss briefly here.

Despite a growing body of valuation literature, we feel much of the review work thus far has focused on identifying and demonstrating ES values. We have not evaluated this question in depth, but suspect that few of the studies have had an impact on policy ("capturing values" in the TEEB terminology). In the light of policy relevance, we discuss the limitations of average per hectare values estimated for different ecosystems. A particular problem facing state-of-the-art valuation studies is whether to focus on estimating marginal values from changes in individual final ES "end-points" (such as water quality suitability for particular water uses), or value bundles of ES associated with large non-marginal changes in whole ecosystems. The former is advantageous for benefits transfer, but partial in nature, the latter is comprehensive, but confounds value transfer because of trade-offs between user interests and resulting context dependence of bundles of ES. Part of this confounding is due to difficulties in distinguishing intermediate and final ES using the MEA nomenclature, resulting in the danger of double counting benefits and values. Problems distinguishing intermediate and final ecosystem services is in part due to the way geographical scope and resolution co-define ES in watersheds. Valuation has focused on provisioning and cultural services that are directly enjoyed by people on-site in a natural area (ecosystem). We then argue that regulating and supporting services have not been the subject of economic valuation because they require dose-response/ damage functions /production function modelling. Modelling of regulating services such as flood reduction and pollution control needs to be spatially explicit if it is to address economic interests and their locations, and in turn be policy relevant. We finish with a few words regarding dissemination of the "ES" concept in Nordic countries.

6.2.1 Policy relevance – from awareness raising to policy support

One objective of the VALUESHED review was to find Nordic examples of watershed service values instead of frequently cited examples from international studies. One such example – the value of the Catskills watershed for water purification – has been frequently cited and important in the policy framing of research on the value of watershed services (Sagoff, 2002). Sagoff cautions the use of simple parables that "undeveloped nature provides services spontaneously and that human manipulation, intervention, or transformation cannot improve and therefore only diminish service" (Box 14).

Box 13. The Catskills parable

"to get benefit of nature's services (...).is to depend not so much on nature directly, but on technologies that lift the cup of nature to the lip of consumption" (Sagoff, 2002).

Many cite a decision by New York City to spend over $ 1 billion to purchase land in the Catskills watershed to secure or restore the capacity of the natural ecosystem to purify the City's water supply, rather than building a new filtration plant (Sagoff, 2002). Sagoff discusses how the choice faced by New York City was defined by specific regulations and the regulatory approach EPA adopted towards water treatment and watershed management. A new regulation in 1989 required surface water systems serving more than 10 000 people either filter water or petition for a "filtration avoidance determination" from the EPA. Despite meeting high safety standards and quality of drinking water the City of New York undertook a number of measures, including dam and pipe renovations, waste-treatment and septic-system improvement and farm-operation enhancements. By 1997, however, the City had purchased only 19200 acres of the 355 000 acres it had signed a Memorandum of understanding to purchase with the EPA (and upon which estimates in the literature have been based). In the end the ecosystem service of water purification was (1) defined by a regulatory technical standard, (2) complied with using technological solutions which were deemed more cost-effective than restoring natural land uses in the Catskills watershed.

In order to move from awareness raising to decision support – from parable to policy – the question to be asked before starting a valuation study is quite simply; "what is its policy purpose? "

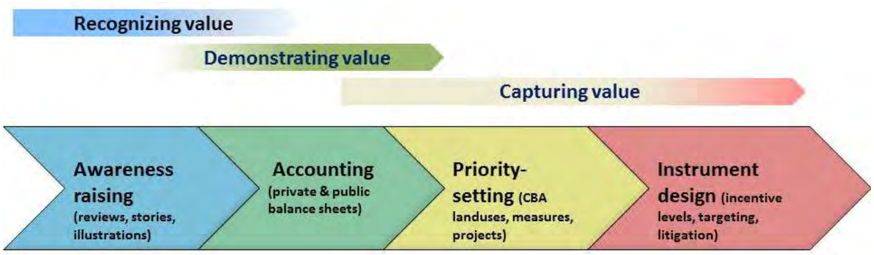
A common policy starting point for justifying economic valuation of ES is that markets' profit from nature is privatized, while the cost of destroying it is socialized. Economic valuation methods will help the public take the cost of destroying the nature into account in benefit-cost analysis of projects and policies. Valuation practitioners such as ourselves readily admit that it is impossible to put an exact price on environmental quality. Nevertheless we believe that approximate valuation can be useful.

In the context of the Water Framework Directive, valuation of watershed services would seem to hold out the promise of supporting the categorization and selection of measures, justification of derogations to the objective of "good ecological status", and benefit-cost analysis of trade-offs between different measures and different stakeholders (Brouwer, Barton et al. 2009). The WFD mandates cost recovery of water services, which if socially defined would also imply a role for economic valuation in determining the size of water charges that internalize ecosystem damages due to water use.

This simple framing of the problem needs to be nuanced in terms of *what kinds of policy* we can expect valuation methods to inform depending on *how approximate* they are.

Valuation studies are required to be increasingly reliable and accurate as their purpose progresses beyond recognising and demonstrating value to capturing value in policy. Beyond using valuation studies as information for framing policy debate through raising awareness, valuation of ES can be used in (i) accounting, (ii) priority-setting and (iii) instrument design (Figure 19).

Figure 19: Different policy contexts of economic valuation of ecosystem services



6.2.2 From average values to incremental values of policy change

The TEEB study (Kumar 2011) collected over 1300 original values from 160 valuation studies in an Ecosystem Services and Valuation Database (ESVD) (after a screening of many hundreds from a number of databases.[26] In Table 3 we extracted information for the biomes addressed in TEEB relevant for Nordic countries and watershed services. There were no observations for polar and high mountain systems with respect to the flood and pollution reduction services we chose to focus on in this review.[27]

Values cited from the review are average per hectare values. Despite this level of aggregation some noteworthy features of the values summarised in the table are:

- At the global level of this review, there were no more than a handful of valuation studies on the effects of land-use on economic activities in watersheds. Most of these were from wetlands and tropical forests (included here for comparison as the most studied of the terrestrial biomes)
- High values are associated with wetlands (which are generally found downstream), relative to forests (which play a role in the upstream part of the catchment)
- There is an enormous variation in values between studies. Whether water flows are "extreme" or "regulated"(low and stable?), has very different values
- All studies are based on the assumption that ES valued by a downstream beneficiary can be associated to a surface area of the biome, in order to calculate average $/ha values
- The accuracy of the individual valuation studies are not reported, only the value ranges across studies. The large variation in average values would nevertheless increase if for example a 95% confidence level was required

The TEEB studies meta-analysis of ES values was restrictive. For example it only included valuation studies for which it was possible to infer a per hectare value to a particular biome in the study. This would have excluded a number of valuation studies of for example changes in water quality suitability which did not have any information on the catchment area in which the study took place. It could also exclude valuation stud-

[26] http://www.fsd.nl/esp/77395/5/0/30

[27] Freshwater storage in ice caps is mentioned in the TEEB report, but there are no valuation references.

ies conducted for water bodies where there was no identifiable boundary to the ecosystem, e.g. coastal waters and shores of large lakes.

The selection criteria chosen by the TEEB meta-analysis highlights a fundamental problem in ES valuation, that of associating the value of a change in environmental quality or resource availability at the valuation location to the location and area of ecosystem that regulates or supports this change. Where such an association can be supported by the information in the available studies, $/ha values derived are average values across the area of the ecosystem/biome. Average values assume a linear relationship between the area of the ecosystem and ES values. Brander et al. (2010) show that studies focusing on marginal loss of wetlands services produced higher willingness to pay estimates than studies focusing on average values. If this is also true that ES values are not linear in ecosystem area, what policy applications are average values relevant for?

6.2.3 Valuing final ES is not the same as valuing ecosystems themselves

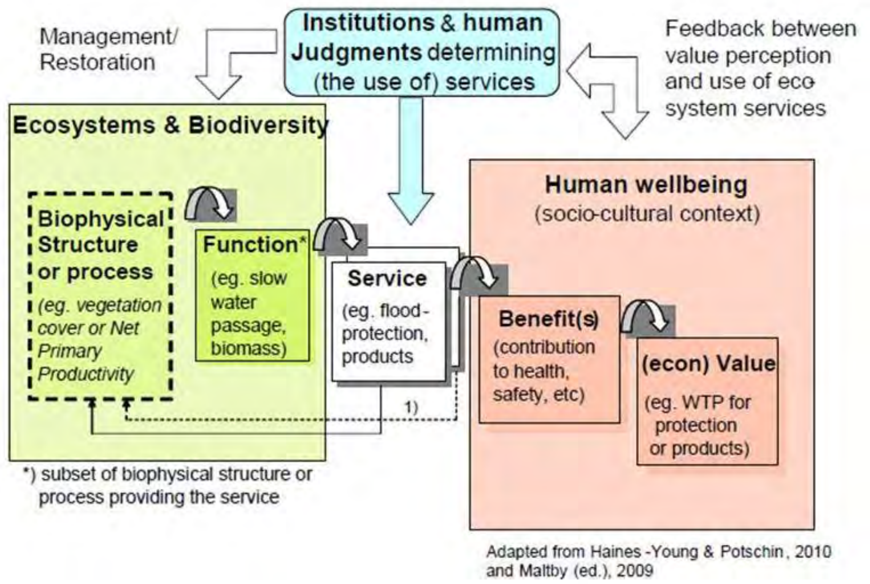
The case studies from Morsa and Odense River Basins showed that even for water quality improvements due to reduced eutrophication which is one of the most widely valued in watersheds, we do not retrieve precise valuation estimates for the specific ES in question. In this case, the problem may lie as much in a poor definition of ES indicators, as in the endpoint addressed by the valuation study. In the case of the AQUAMONEY study a conscious decision was taken to value changes in an ES outcome: "ecological status" of lake and river water. In order to value the ecosystem component providing this water quality, biophysical modelling of DPSIR chains is required. For example, Barton et al. 2008 try to link contingent valuation estimates of water quality in Vansjø to a lake water quality model and a runoff model of management of cultivated land and artificial wetlands (modified ecosystems). Without this modelling the water quality valuation endpoint cannot be associated with ES of the lake and ecosystems in the catchment.

More specific studies for specific ES may even reduce the problems we faced regarding dividing valuation estimates to different ES and the dangers of double counting. However, as we discussed earlier, some of these problems may not be easily overcome even by more studies, but may be due to lack of correspondence between how ES are defined (or not precisely defined) and the concepts implicit in economics and economic valuation, for example with respect to how intermediate and final services and how valuation methods like CV encompass both use and non-use values, while these are separated in the classification of ES. Further, the ES classification operates with several services that would be part of "non-use values" when valuating using the CV method – and it is as open question whether researchers and respondents would be able to separate non-use values related to different non-use services from each other.

6.2.4 *Distinguishing intermediate, final ES , benefits and values*

Following the MEA (2005), the TEEB study distinguishes between ecosystem functions, ecosystem services, benefits and values. However, the well-known TEEB cascade model in Figure 20 is not a rigorous definition of ES, but rather conceptual model for explaining the link between ecosystems and human welfare (pers. com. Roy Haines-Young).

Figure 20: The pathway from ecosystem structure and processes to well-being

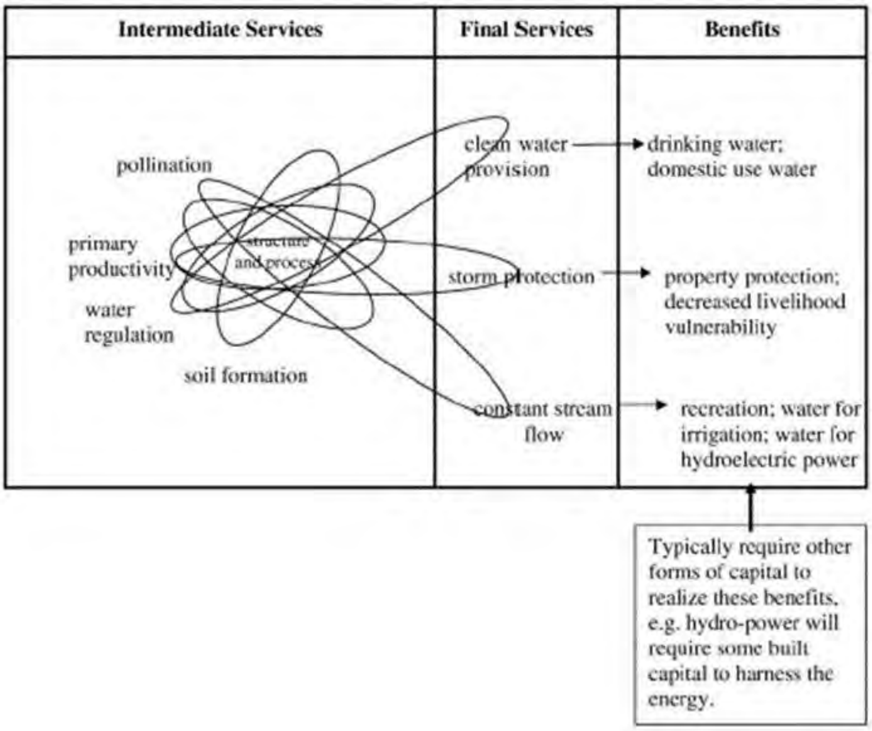


Source: adapted from Haines-Young & Potschin (2010) and Maltby(ed, 2009) in Kumar(2011).

Several authors use the term "intermediate and final ecosystem services" where TEEB refers to ecosystem function and ecosystem services (Boyd and Banzhaf 2007; Fisher, Turner et al. 2009; Bateman, Mace et al. 2010).

Fisher et al. (2009) illustrates that the concept of intermediate services is comparable to TEEBs notion of ecosystem function. Final ES such as provision of clean water is distinguished from benefits in a clear economic sense. In order for final ES to be benefits, man-made capital and human labour are required as inputs – for example in final treatment and distribution of drinking water to peoples' taps, or hydropower through generation and distribution through the power grid to home appliances (Figure 21).

Figure 21: Intermediate, final services and benefits. Benefits from ecosystem services are derived by using human capital and labour inputs



Source: Fisher et al. 2009.

Boyd and Banzhaf (2007) have also pointed out that valuation studies often do not distinguish between "intermediate" and "final" ES, and that final ES, cannot be valued directly without accounting for other inputs. For accounting purposes Boyd and Banzhaf (2007) suggest that the conceptual confusion between intermediate and final regulating services, and difficulties in identifying these as separate from inputs, makes ecosystem "stocks" a more robust surrogate indicator of the economic importance of ES.

An additional message that arises from our review of Nordic valuation studies is that benefits enjoyed from regulating services are in many cases co-determined by legal health and safety standards. Such regulations determine what uses are allowed or advised to do, and so determined thresholds for when an ES – i.a. increases in environmental quality – actually provide a benefit. For the regulating services focused on in this report – reducing flooding and pollution from land-uses – the definition of where benefits are measured, and how they are co-produced, is important for the discussion of the roles of different valuation methods in the rest of the report.

Table 12 exemplifies how the benefits of a number of watershed services are in many cases defined as benefits through regulation, rather than pre-existing consumer preferences. The benefits of watershed services are only enjoyed with the help of a number of capital and labour inputs. In some cases, labour and capital inputs are substitutes for the ES – such as drinking water treatment or flood prevention.

Table 12 Differentiating ecosystem function, services and benefits

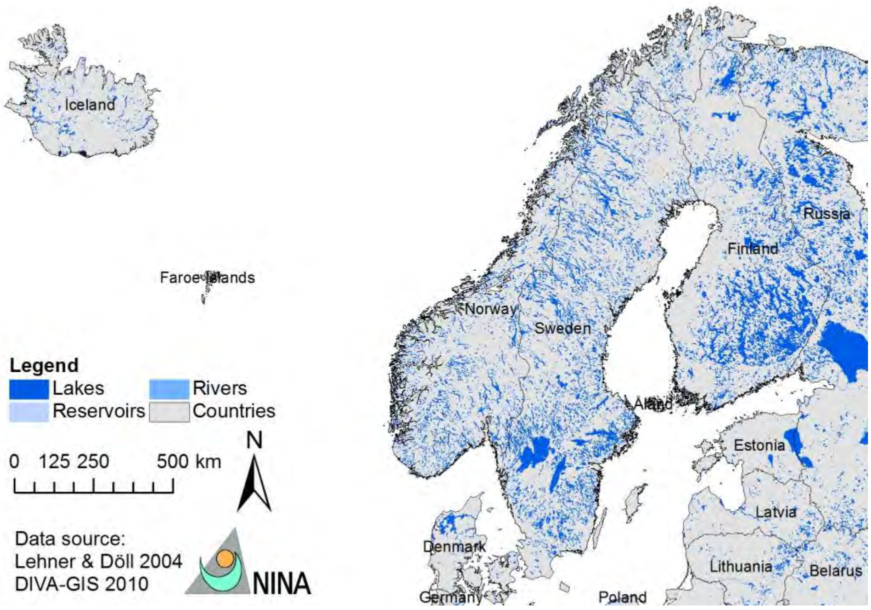
Ecosystem "stock" indicators	Intermediate ecosystem services? (=ecosystem functions?)	Final ecosystem service? (=outcomes?)	Health, safety & insurance regulations	Labour & capital inputs	Benefits (increases in...)	
Forest and other vegetation cover configuretion in watershed; Wetland location in hydrological cycle	Moderation of extreme events (i.a. floods, drought)	Reduced probability of peak streamflow levels	Flood damage liability River regulation concessions	Flood prevention measures	Avoided damages building, infrastructure, farming	Biodiversity conservation downstream?
		Reduced probability of low streamflow levels	River regulation concessions	Run-of-river hydropower plants & el.grid	Electricity availability	
				Travel and equipment	Angling days Boating days Biodiversity conservation	
	Waste treatment / water purification	Increased quality of raw water intake for drinking	Drinking water quality standards	Drinking water treatment	Drinking water availability	
		Increased Quality of surface water	Bathing water standards	Travel and equipment	Bathing, boating days	
		Increased quality of water column	Dietary health advisories	Travel and equipment	Angling days Reduced health risk	
	Erosion prevention	Reduced concentration article bound nutrients in wetlands	Drinking and bathing water quality standards	Drinking water treatment	Drinking water availability	
		Reduced sedimentation of reservoirs			Avoided maintenance cost	
		Reduced natural capping of contaminated sediments	Dietary health advisories		Dietary health risk	

6.2.5 Tradeoffs in the spatial resolution of ecosystem service valuation

In this section we discuss the spatial resolution at which measurement of ecosystem services and economic valuation is meaningful for different types of decision support in watershed management. When funds available to do valuation studies are fixed there is a trade-off between (i) the needs for high temporal and spatial resolution, (ii) the need for geographical scale and spatial aggregation and (iii) the need for different levels of reliability and accuracy with different "policy-purposes" of valuation. We briefly discuss the possible relevance of economic valuation at different scales and resolutions.

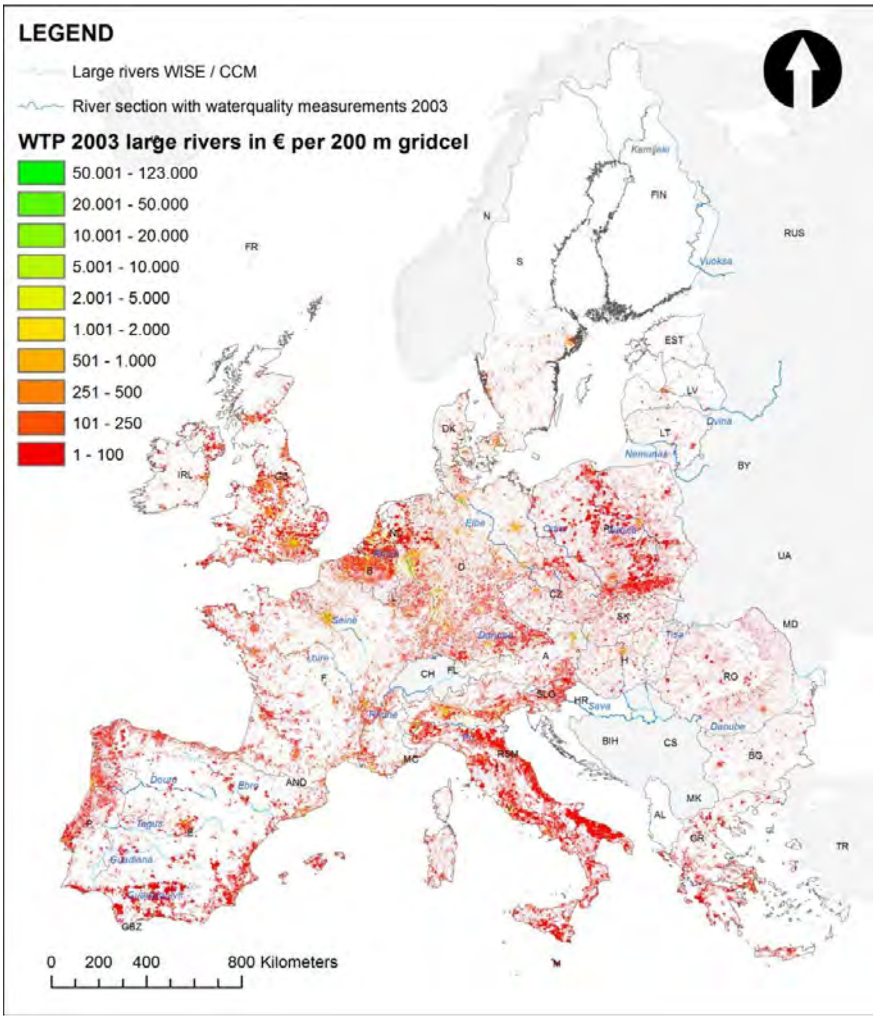
Figure 22 portrays wetlands in the Nordic countries at a high spatial scale, but also high resolution. The map shows that freshwater wetlands (lakes, artificial reservoirs, rivers) are a defining and common feature of Nordic landscapes. High resolution makes it possible to determine the relative importance of wetlands in different countries and water regions. Average estimates for the "total economic value" of wetlands could be applied using "value transfer methods" to derive aggregate wetland values for each water region.

Figure 22: Wetlands in Nordic countries



Despite higher resolution at this scale, we are still not in a position to identify trade-offs between different types of user interests in any particular wetland and watershed landscape. For example, it is difficult to distinguish between natural lakes, artificial reservoirs and rivers which often have different constellations of user interests. High scale combined with high resolution implies a lot of spatial information, and higher study costs, unless "averaging" assumptions across wetland types are made using for example meta-analysis based value transfers. A combination of high scale and high resolution might be relevant for national accounting of physical natural capital.

Figure 23: Value map for water quality improvements in the EU

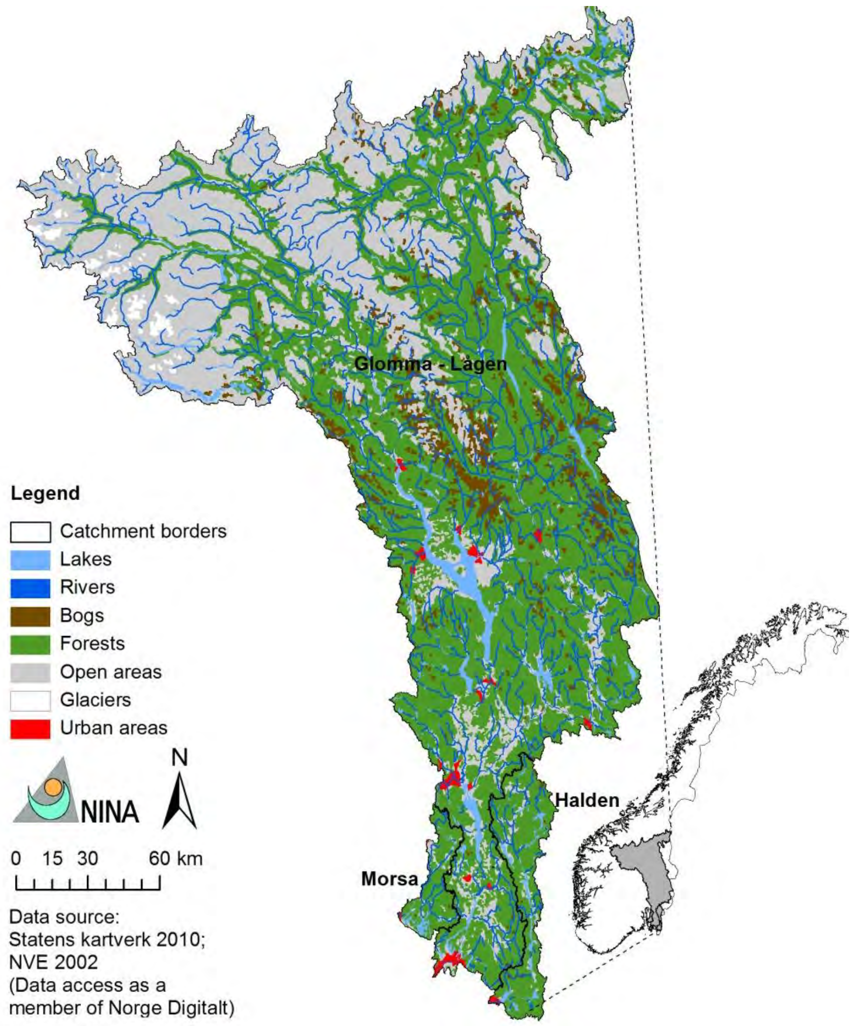


(Source: Brouwer et al. 2009).

Figure 23 shows the results of a value transfer for rivers in the EU based on results from the AQUAMONEY project (Brouwer et al. 2009). Spatially explicit contingent valuation was used in several Northern European countries to derive distance decay functions for willingness to pay to improve rivers and lakes to "good ecological status". Results were then transferred to rivers in the whole of the EU. Valuation estimates appear to be driven in large part by population densities in proximity to rivers (due to distance decay of WTP).

Comparing the economic valuation of rivers in a country such as Finland or Sweden, with the geographical importance of wetlands in the previous map, we can see that value transfer maps at high scale, despite the high resolution, have an awareness raising function. Another possible application might be to national accounting. In terms of priority setting and policy instrument design the scale is too aggregate for needs related to land and water management in a watershed. A hypothetical future application could conceivably be in the context of scaling budget allocations to different water region authorities based (in part) on the economic importance of wetland natural capital.

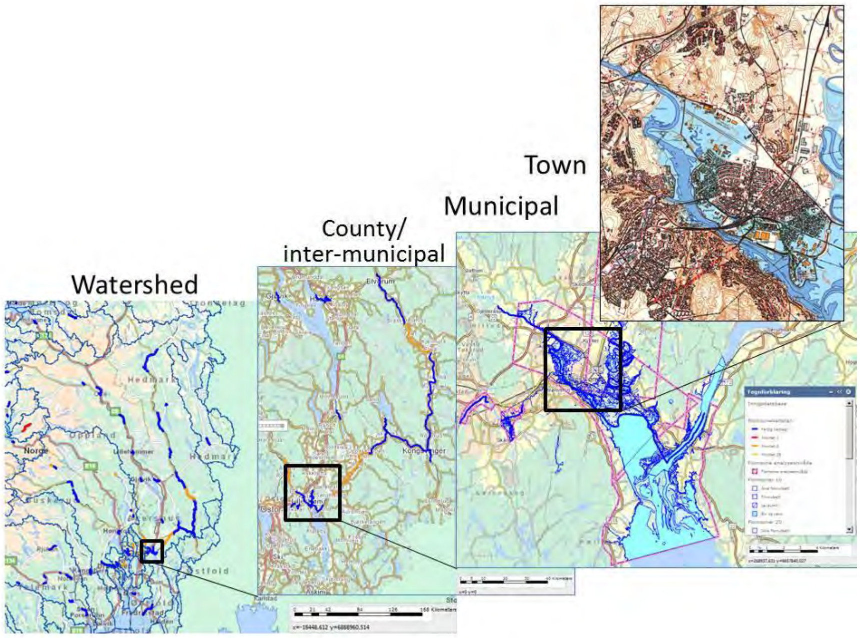
Figure 24: Landcover at watershed level. Glomma-Lågen, Morsa and Halden river basins



Kareiva et al. 2011 discuss using the economic value of each type of land cover in providing flood peak and water pollution reduction to prioritise management measures of different land use types and even for targeting of incentives. Figure 24 shows an example of a land cover map for a large river basin (Glomma-Lågen, Norway) and two neighbouring catchments (Morsa and Halden). As exemplified by the Norwegian demonstration cases in this report, some land cover types have been shown to correlate with run-off. However, at this scale land cover classes are general (agricultural land, forests and bogs/mires) and have a relatively low resolution, making distinctions between land uses for priority-setting purposes difficult, particularly in the smaller catchments such as Morsa and Halden. There is a trade-off between scale and resolution also in economic valuation.

Figure 25 shows a decrease in geographical scale in order to increase resolution and focus on particular land uses prone to flood risk. In order to value differences in flood risk of different types of land and infrastructure, a municipal, town or lower geographical scale is needed. Flood scenarios at this high resolution need to be linked to hydrology of water routing through the river basin at a larger scale and lower resolutions, and then linked to land management in the catchment at lower scale and higher resolution. This need for rescaling models of ecosystem functions between where watershed services are enjoyed and where they are provided, is a major challenge for valuation of regulating services such as flood and pollution reduction. The data collection challenge is smaller for smaller catchments, but in principle the same.

Figure 25: Measuring flood risk at different spatial resolutions compared to administrative boundaries



Watershed, county and municipal resolution maps are from the Glomma-Lågen river basin as mapped in NVE Atlas with the flood zone theme. Highest resolution map shows the town of Lillestrøm and is taken from flood zone scenario maps produced by NVE (in this case flooded areas in a 500 year flood in light blue).

Based on the different mapping examples above, summarises some conceptual challenges in economic valuation when it relies on biophysical modelling of ecosystem function to link land and water management decisions to ecosystem services. TEEBs three economic valuation steps – recognizing, demonstrating and capturing value – are depicted in relation to some broad policy applications of valuation. Information costs of valuation increase with increasing spatial scale, resolution and need for accuracy required of policy in these settings.

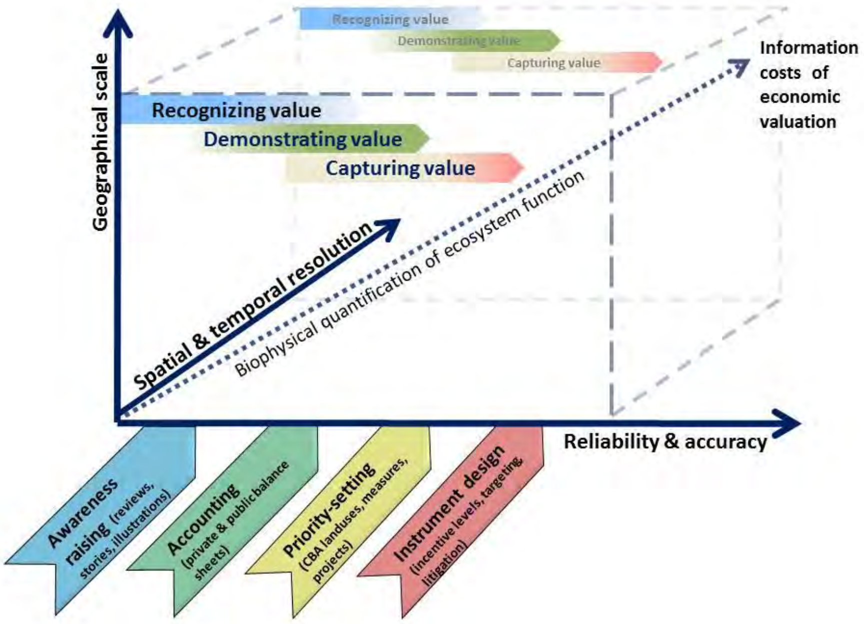
Valuation may be used for *awareness-raising* in different decision arenas. TEEBs reviews of valuation studies, best-practice case study and "valuation success stories" fulfil this role. Valuation asks the "coarse grain" question "*is the ecosystem service's value significantly greater than zero?"* The requirements for reliability and accuracy are low relative to other policy contexts.

Accounting requires higher reliability and accuracy, also depending on at what scale aggregation is taking place (national, private company). Valuation answers the relatively coarse grain question, "*is the aggregate value of flows of ecosystem services from natural capital increasing or decreasing?"*

In *priority-setting* such as cost-benefit analysis, valuation is meant to answer a relatively "fine grain" question such as, "*are the ecosystem services provided by one land use significantly more valuable than by another land use?" Also, "are the benefits disproportionately greater than the costs of the project or policy measures; are the net benefits significantly greater than zero"?*

Finally, the greatest level of reliability and accuracy is required when using valuation for *instrument design*, such as scaling and targeting incentive levels to particular land use management practices, or calculating *litigation* claims for natural resource damages in a court of law. Economic valuation is required to answer the question, "*what is the absolute value of opportunity costs to landowners, or of benefits from the targeted land use? What is the absolute value of interim damages to plaintiffs due to some act of negligence by the accused?"*

Figure 26: TEEBs three economic valuation steps – recognizing, demonstrating and capturing value – in relation to different policy contexts of valuation



Information costs of valuation increase with increasing spatial scale, resolution and need for accuracy and reliability because of increased requirements of biophysical quantification of ecosystem services. Source: own elaboration.

Demonstrating and then capturing value has increasing information costs, which stem as much from modelling ecosystem function, as from gathering data on ecosystem service demand using the economic valuation methods. Costs are increasing with increasing spatial scale needed for aggregation of economic valuation estimates e.g. across a whole watershed. Costs are also increasing in the spatial and temporal resolution required (where are the land and water users located exactly, and at what time of year does their land use provide an ecosystem service?). With a particular decision at hand – raise awareness, account, prioritise, or instrument design – and a fixed budget for gathering information, there are some hard trade-offs to be made between spatial scale and spatial and temporal resolution of valuation estimates. A tiered modelling approach is recommended by the Natural Capitals project depending on data availability (Tallis and Polasky 2011). An evaluation of what policy questions each modelling tier can answer is also called for.

6.2.6 *Public dissemination regarding ecosystem services from watersheds*

The expectation is that "ecosystem services" will make ecosystems more visible in decision-making of individuals, organisations, businesses and governments. In this section we briefly discuss the challenges of communicating the benefits and economic valuation of ecosystem services from watersheds. This is a particular challenge because "ecosystem services" introduces a language that in several ways overlaps existing popular and scientific terminology, and has so far been explained and illustrated mainly using abstract and conceptual diagrams.

Prior scientific concepts perhaps familiar to economists might include "external costs and benefits"; "environmental costs and benefits"; "public goods", " natural amenities" and "environmental goods" , "biodiversity" and "natural capital", "natural systems", "natural assets". Prior popular concepts perhaps more familiar in the population might include "user interests"; "nature's own value"; "environmental quality", "natural resources", or simply a myriad of names of the species and places that mean something to people.

The Millenium Ecosystem Assessment's shared vision, conceptual framework and synthesis of knowledge illustrated the many ways in which natural systems are vital assets critical for human well-being (Kareiva, Tallis et al. 2011). "Ecosystem services" can encompass all the concepts above. In seeking an all encompassing concept in English, Nordic languages can be challenged in translating this to a concept that can have popular understanding, political recognition and scientific meaning. In Norwegian for example, the translation "økosystemtjenester" makes associations with "favours" or "service sector of the economy". Institutes such as NINA in Norway are discussing whether to use a well known popular concept such as "naturens goder", but provide it with added content.

In this report we have not used popular conceptual illustrations, perhaps because we are communicating with our traditional policy and science audience. However, as the brief discussion above shows, in the communication of ecosystems and their services with the public, much more thought will have to be put into illustrations – both anecdotal, iconic, artistic, photo, mapped – in defining ecosystem services in a way that connects ecosystems to popular and commercial interests. In fact it will be interesting to see to what extent interests will be redefined in this process, extending perhaps beyond users' private and site-specific interests to other interests and the wider society.

Figure 27: Iconic illustrations of ecosystem services used in the TEEB

(Copyright Jan Sasse).

The process may be similar to what happened when "biological diversity" brought into the public eye in the process leading up to the Convention on Biological Diversity in the 90's. A sign that biological diversity became an accepted expression beyond science was perhaps when its contraction "biodiversity" became widely used shorthand (in what media arenas?). We might similarly expect ecosystem services to become "ecoservices" when the concept has been mainstreamed into public debate – "økosystemtjenester" to become "økotjenester" in e.g. Norwegian. Will a future "ecoservices" concept have the same place in people's awareness as "biodiversity" does today? What role does "biodiversity" play today in public discourse? From science writers' point of view, we definitely see a need for national TEEB follow-ups in Nordic countries to decide on a consistently used concept in local Nordic languages if the local debate within science is to take hold in policy and popular awareness.

7. Conclusions and recommendations

In this chapter we focus our conclusions on general recommendations to policy makers in using (or not using) valuation results in different contexts, data gaps and recommendations for further research, and some recommendations for national TEEB follow-ups in Nordic countries, based on our material from the literature review and case studies.

7.1 Conclusions

Caveats and sample selection bias

Our review has sought studies that address non-market values of final ecosystem services. Regulating and habitat supporting services are difficult to classify as final services for economic valuation. The TEEB classification of regulating and habitat services is classified by some economists as intermediate services, and as such are not directly enjoyed or easily valued by people using non-market valuation methods, although such values may be included as an important part of people's non-use values. Other regulating services are poorly covered in our review because we focused on aquatic systems, or impacts of land-use on aquatic systems (e.g. pollination and biological control).

Coverage of valuation studies

Our literature review shows that the watershed ES valued are quite similar across the Nordic countries. The services addressed are mainly provisioning services as food and fresh water supply, some examples of regulating services like regulation of water flows and water purification, as well as cultural services like aesthetic information and opportunities for recreation and tourism. Most valuation studies value a bundles of goods and services resulting from (changes in) specific watersheds or water bodies, rather than individually specified ES, with the exception of studies carried out for valuation of recreational fishing in particular. Contingent valuation and choice experiments dominate among the methods applied. Despite some examples reviewed in this report, valuation studies of regulating and supporting/habitat services seem to be under-represented

Values of watershed services in Nordic watersheds – flood reduction

Examples from the Glomma-Lågen watershed show that the value of regulating services can be indirectly derived from valuation of property at risk. Flood damages for many hundreds of millions of kroner happen annually. Establishing the link between flood risk and the condition of ecosystems in the watershed is nevertheless a complex biophysical modelling task. The value of flood reduction services provided by upstream ecosystems is more difficult to identify the larger the watershed, the larger the storm event, and the more regulated the watershed is by man-made infrastructure (reservoirs, transfers, channeling). The value of flood damage reduction depends on a combination of preventive, avoiding and mitigation actions throughout the catchment, and in particular in the downstream areas at risk of flooding. Empirical analysis and model simulation in Norway have been able to show only a limited and local role of historical landuse changes on the hydrological cycle. Non-market valuation studies have not been carried out, perhaps because flood damages are captured in the insurance market. Psychological well-being relative to flood risk exposure and mitigating behavior may be a policy relevant valuation study.

Values of watershed services in Nordic watersheds – water pollution reduction

A number of stated preference studies of water quality, in particular related to eutrophication, have been conducted in the Nordic countries. Contingent valuation and choice experiment studies have either focused on improving bundles of goods and services through hypothetical management measures of "whole watersheds", or focused on valuing incremental changes in suitability for specific water uses, using different variations of a water quality ladder. Valuation studies looking at definitions of "good ecological status" under the Water Framework Directive, while designed to be directly policy relevant, are not necessarily useful for finding per hectare values for ecosystems, or for benefits transfer to other policy contexts. Linking the values of water quality in lakes, wetlands and rivers to landuse management in the watershed is possible by linking biophysical models of landuse pressure – water body quality – use suitability, but has seldom been undertaken. Uncertainty of biophysical modelling can be as large as that of economic valuation estimates. Aggregation of values of water quality improvements and defining "the extent of a market" is possible with valuation studies that evaluate "distance decay" of willingness to pay depending on how far respondents live from water bodies. Research findings are mixed on the strength of "distance decay" for use values of water bodies.

Aggregation of ecosystem service values

Aggregation of value of flood reduction damages from case studies to the whole watershed were not attempted, because site specific flood damage modeling is required. The reliability of transferring economic damage functions is limited, in particular for buildings and agriculture. Local trade-offs and interdependencies between ES mean that they are generally non-additive for a particular wetland or catchment land-use.

7.2 Recommendations

Policy framing of valuation

Based on our review we argue that valuation studies framed to address economic analysis of a particular policy such as the Water Framework Directive are responding to a different policy need than studies aiming at calculating average per hectare values of ecosystems. Commissioned valuation studies must start by addressing what *kinds of policy* they are aimed at informing as a function of *how reliable and accurate* the valuation method has been found to be relative to policy requirements. Beyond using valuation studies as information for framing policy debate through raising awareness, it should be made clear whether specific studies of valuation of ES are to be used for (i) accounting, (ii) priority-setting or (iii) instrument design. Valuation studies are required to be increasingly reliable and accurate as their purpose progresses beyond recognising and demonstrating value to capturing value in policy.

Valuation of alternative land-uses and management practices

Associating values of water quality to states of the ecosystem involves combining pressure-state-impact modelling of run-off from land and water uses to status of water bodies. Watershed management in Nordic countries is seldom about how many hectares of land to allocate to "natural" ecosystems such as forests, versus agriculture. It is much more about identifying and targeting the most cost-effective agricultural practices and run-off mitigation measures, and determining whether the aggregate benefits to downstream users exceed the total costs of a programme of measures. The focus on valuing ecosystems contribution to human well-being while laudable, must avoid a focus on trying to isolate the value of "natural" ecosystems if this is at the expense of tried and tested methods such as cost-effectiveness analysis.

Valuing distributional impacts

Modelling of regulating services such as flood reduction and pollution control needs to be spatially explicit if it is to address economic interests and their locations, and in turn be policy relevant. Different interests live and use different "hectares" of an ecosystem. Watershed management – particularly in the Nordic countries – is seldom about large reallocations of land and water use rights to different types of users and landcover (with different average per hectare landcover/ecosystem values). In most cases it is about prescriptions for marginally different management techniques of existing land and water use rights. Average values of ES do not address income distributional issues, except at the level of differences between ecosystems – average land-users in one ecosystem can be identified as having different income levels from average land-users in another ecosystem. Average per hectare ES values "hide" conflicts of interest between different users using the same ecosystem and trade-offs between them. We therefore argue that calculation of average per hectare ES values may be useful for awareness raising and accounting at aggregate levels. However, it is not useful for the part of policy addressing priority-setting and instrument design.

Economic valuation for instrument design

Priority setting between alternative land-uses, projects, and measures is at its core identifying how land and water use values differ between interests at specific locations. Economic valuation is useful for instrument design if it can help predict how similar incentive levels would lead to different behaviour of different interests at different locations; or how to target incentives differentially across different interests and locations in order to achieve similar behaviour.

Valuing transaction costs of establishing rights to and regulation of ecosystem services

Ecosystem services – what functions of natural systems are perceived as beneficial by people – is in part defined by policy regulation. Particularly in high income countries such as the Nordic, health and safety standards probably play a large part in how we perceive nature. For example, new and higher standards for quality of drinking water increase the costs of achieving those standards, and induce higher value on natural and technical treatment processes. Insurance policy defines what kind of behaviour is negligent, for example with regards to actions taken before, during and after floods. Stricter definitions of negligent behaviour in the face of flood risk (e.g. building on a site with high flooding probability; not taking action to minimise damage) defines property at risk and the potential value of flood risk reduction. Regulations define what rights private interests have to ecosystem services, to public measures to provide these services, and to compensation if these are lost. Establishing these rights through regulation is costly (establishing regulations, monitoring and enforcement). However, valuation of ES seldom – if ever – address the transaction costs of defining ES as rights (Vatn, Barton et al. 2011).

Valuation of ES can be used to assess whether benefits exceed costs of ecosystem management. Whether there are grounds for establishing regulation and rights to the ES, as a basis for payments for ecosystem services schemes, depends on the net benefits (of the ecosystem service minus ecosystem management) compared to transaction costs of establishing rights to ES. It is open to debate whether the costs of establishing, monitoring and enforcing these rights and schemes are borne mainly by the public, rather than private interests participating in a PES scheme (Vatn, Barton et al. 2011). Policy relevant valuation research therefore includes research on transaction costs of payments of ES.

7.2.1 Recommendations for further research

At present there are insufficient valuation studies to carry out statistical meta-analysis in Nordic countries. However, in time and with a larger base of valuation studies, it would be possible to calculate uncertainty intervals and standard values of similar ES across countries. Such studies might be useful for demonstrating values of nature and for awareness raising.

Valuation studies across Nordic populations that are representative at national and county/regional level – for generic hypothetical policies – have been useful for scoping benefits of ecosystem management in studies as diverse as hydropower regulation and marine sediment remediation. Such a study has been carried out with NMC funding for recreational fisheries (Toivonen, Appelblad et al. 2000), and could be repeated for other cultural services.

A complementary approach would fund site and project, policy, and measure specific valuation studies of populations within particular watersheds. The dearth of valuation studies for particularly regulating services in watersheds suggests a need for more studies using production function and damage function approaches to be carried out.

Methodological issues that have not been widely evaluated in existing valuation studies – but which are crucial for accounting, priority-setting and instrument design – are the spatial patterns of ES values and their dependence on distance, direction, scale and resolution.

The European Environment Agency has recently proposed an experimental framework for ecosystem capital accounting in Europe. Amongst accounting measures of relevance to watersheds "ecosystem capital water accounts" have been proposed, including "net ecosystem accessible fresh water surplus", and "landscape green infrastructure accounts" including measures of "rivers ecosystem potential" (EEA 2011). Monetary accounting tables would quantify ecosystem capital depreciation, amongst others in these measures. Future research could demonstrate possibilities and limitations in scaling existing water body and watershed specific valuation studies using damage function and cost-based approaches for purposes of ecosystem capital accounting.

Developing visualizations and illustrations of ecosystem services will help promote public awareness as a supplement to economic valuation, but may well provide economists with tools they can use in non-market valuation studies as well.

7.2.2 *Specific recommendations for TEEB follow-ups in Nordic countries*

Priority setting and instrument design

Throughout the review we have argued for specific policy framing of valuation studies. The Water Framework Directive is the main policy tool to manage water resources in EU (and Norway and Iceland). The WFD focuses its goal on ecological status of water bodies – which at the outset is not related to people's use or benefits derived from these water bodies at all. Thus the goals for good ecological status have been determined with only indirect considerations of people's goods or benefits from the water bodies. On the other hand, the WFD stresses that the users should be consulted and included for instance in economic analysis.

Nordic countries could sponsor research on the design of economic instruments of WFD "programmes of measures" for watershed management (such as payments of ES) – assessing their ecological effectiveness, benefits of derived ES, technical and transaction costs of implementation, distributional impacts and legitimacy, institutional and political barriers and opportunities for implementation.

In particular regarding valuation, Nordic countries could conduct comparative primary valuation studies to further demonstrate the use of Guidelines (like the ones developed in AQUAMONEY) for using economic valuation under the Water Framework Directive. In particular, case studies could evaluate what kinds of valuation methods and accuracy were relevant for

- justification of derogations to the objective of "good ecological status"
- extending cost-effectiveness analysis to benefit-cost analysis to prioritise between different measures determining "full social cost" recovery of water services

Policy impact of valuation studies

Despite the recent focus on ES brought on by the TEEB study the last five years, natural resources and environmental quality have been the subject of economic valuation studies for almost three decades in the Nordic countries. Nevertheless, the policy impact of the numerous valuation studies that have been conducted has never been reviewed. Within which policy sectors have valuation studies been used as decision-support, and for what types of decisions? Has lacking policy impact been due to a lack of awareness, of conceptual definition of goods and services, has it been due to characteristics of the valuation methods themselves, or differences in management traditions across Nordic countries? Can the advent of the "ES" concept be expected to be a "game changer" in this regard?

Ecosystem services in Nordic languages
Comparative analysis of valuation of ES would be facilitated by a common terminology in Nordic languages.

Other ecosystem reviews
For practical reasons VALUESHED limited its focus mainly to ES values in or from wetlands. We largely ignored other interconnecting ecosystems in Nordic countries (e.g. forests, coastal wetlands and open sea ecosystems), and addressed only cursorily the interdependencies of valuation estimates between ecosystems (e.g. off-site ES of forests). Addressing other ecosystems in specific reviews would be a complement and follow-up to Nordic TEEB.

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9. Sammendrag

Valueshed-rapporten diskuterer innledningsvis en definisjon av økosystemtjenester fra nedbørfelt – «nedbørfeltstjenester» (Kapittel 2). Vi gjennomfører en literaturgjennomgang av verdsettingsstudier i Norden og sammenligner denne kunnskapen med lignende litteraturstudier som er gjort internasjonalt (kapittel 3). Vi ser i detalj på økonomisk verdsettingsstudier som er gjennomført i nedbørfelt i hhv. Norge og Danmark (kapittel 4 og 5). I kapittel 6 diskuterer vi metodiske utfordringer med verdsetting av økosystemtjenester i nedbørfelt. Kapittel 7 legger frem konklusjoner og anbefalinger om videre studier. Valueshed-rapporten inneholder også tilleggsmateriale – vi presenterer en trinnvis metode for verdsetting basert på veilederen fra EU-prosjektet AQUAMONEY (appendiks 1). I appendiks 2 diskuterer vi kort ulike økonomisk verdsettingsmetoder for lesere som er ukjent med denne litteraturen.

9.1 Hovedfunn

Litteraturstudien viser at økosystemtjenester (ØT) i nedbørfelt har blitt verdsatt på noenlunde samme måte i alle denordiske land. De fleste verdsettingsstudiene har sett på forsyningstjenester, som for eksempel mat og vann, men også kulturelle tjenester, som for eksempel fritidsbruk. På tross av noen eksempler på regulerende og støttende ØT som diskuteres i denne rapporten, er disse ØT underrepresentert i nordiske verdsettingsstudier. Disse ØT er bl.a. vanskelige å klassifisere som slutttjenester som kan verdsettes direkte. Regulerende tjenester er også underrepresentert i vår gjennomgang fordi vi fokuserte på verdsettingsstudier av vannforekomster, og i mindre grad på hvilke konsekvenser endret arealbruk har på den økologisk statusen for vannforekomstene.

Å etablere en sammenheng mellom flomrisiko og tilstanden til økosystemene i et nedbørfelt er en kompleks biofysisk modelleringsoppgave. Romlige egenskaper ved nedbørfeltet spiller en rolle. Verdien av flomdemping som en tjeneste fra arealbruk i nedbørfeltet er vanskeligere å anslå i store nedbørfelt, i forbindelse med store nedbørsepisoder, og i vassdrag som er mer regulert av menneskelig infrastruktur (vannmagasiner, overføringer, kanaliseringer). Den økonomiske verdien av flomdemping avhenger av en kombinasjon av preventive, unnvikende og avbøtende tiltak på tvers av nedbørfeltet, spesielt i flomutsatte områder nedstrøms. Aggregering av verdien av flomdempingstjenester for hele nedbørfelt ble ikke utprøvd i denne studien fordi det ville krevd stedsspesifikk hydrologisk- og flomsone-modellering. Påliteligheten ved å overføre skadefunksjoner for flom er begrenset, spesielt for bygninger og jordbruk. Lokale avveininger og avhengighet mellom ulike ØT, betyr at de stort sett ikke kan legges sammen for en bestemt vannforekomst eller arealbruk i nedbørfeltet.

Et betydelig antall verdsettingsstudier basert på spørreundersøkelser er gjennomført i Norden, spesielt for eutrofiering av vannforekomster og fritidsfiske. Betinget verdsettingsstudier og valgeksperimenter har verdsatt enten «knipper» av goder og tjenester som skyldes hypotetiske tiltak i «hele nedbørfelt", eller fokusert på verdien for spesifikke vannbrukere av å øke vennkvalitet målt med en eller annen form for «vannkvalitetsstige". Verdsettingsstudier som fokuserer på endringer i «økologisk status» i forbindelse med EUs vanndirektiv er utformet for å være direkte tiltaksrelevante, men er ikke anvendbare til beregninger av verdien per hektar for økosystemer i nedbørfeltet. Sistnevnte krever avrennings- og forurensningsmodellering for å kunne tilegne bestemt arealbruk i nedbørfeltet en ØT-verdi i forhold til bruk av vannforekomstene. Aggregering av verdien av vannkvalitetsforbedringer i en vannforekomst til et helt nedbørfelt er mulig dersom man blant annet har anslag for hvordan betalingsvilligheten endrer seg med økende distanse fra husstanden til vannforekomstene som blir verdsatt. Studier har i liten grad sett på disse romlige aspektene ved verdsettingsestimater, og der man har sett på såkalt «distance decay» har resultatene så langt vært blandet.

Basert på vår litteraturgjennomgang ser vi at verdsettingsstudier som er utarbeidet for å vurdere tiltak i forbindelse med vanndirektivet, løser en annen oppgave enn studier som beregner gjennomsnittsverdier per hektar av ulike økosystemtjenester. Verdsettingsstudier som igangsettes i fremtiden bør klarlegge hva slags politiske valg de skal informere, og hvilken nøyaktighet og pålitelighet som kreves i den sammenhengen. Beveger man seg utover folkeopplysning bør det gjøres klart om verdsetting brukes til (i) nasjonalregnskap, (ii) prioritering av tiltak med nytte-kostnadsanalyser, eller (iii) utforming av økonomisk virkemidler. Økende pålitelighet og nøyaktighet kreves når man beveger seg fra å demonstrere verdi av natur på generelt grunnlag til å anvende verdier i konkrete beslutningssammenhenger.

Å knytte verdiene av endringer i vannkvalitet til status av økosystemer i nedbørfeltet krever en kombinasjon av modeller som relaterer avrenning fra arealbruk til vannkvalitet og egnethet for brukere av vannforekomster. Arealplanlegging og vassdragsforvaltning i nordiske land dreier seg sjelden om omdisponering av vesentlige arealer fra et formål til et annet, for eksempel fra skogbruk til jordbruk. Det vil i større grad dreie seg om innretting av tiltak i jord- og skogbruk som øker ressurseffektivitet samtidig som det minimaliserer nedstrømseffekter. Med vanndirektivet vil det også i større grad dreie seg om å vurdere om de samlede kostnadene ved en tiltakspakke overstiger nytten av tiltakspakken i form av bedret økologisk status i vannforekomsten. Verdsetting av økosystemers bidrag til menneskelig velferd er ventet å bidra til mer naturvern, men man må vokte seg for å overfokusere på å «isolere» verdien av «naturlige» økosystemer. Selv der verdsetting av økosystemtjenester ikke er mulig vil kostnadseffektivitetsanalysse være et viktig bidrag til beslutningsstøtte.

Modellering av regulerende tjenester som flomdemping og forurensningsreduksjon må være romlig eksplisitt hvis den skal være kunne brukes til å vurdere avveininger mellom økonomiske interesser, det vil si for at den skal være politisk relevant. Ulike interessenter bor og bruker ulike «hektarer» av et økosystem. Gjennomsnittelige per hektar-verdier av ØT tar ikke hensyn til fordelingsmessige konsekvenser av tiltak, annet enn på et overordnet nivå mellom økosystemer – mellom typer arealbruk. Gjennomsnittelige ØT-verdier per hektar kan bidra til å «gjemme» arealbrukskonflikter mellom interessenter på samme arealtyper – innen de samme økosystemene. Beregning av «per hektar ØT-verdier» kan ha en nyttig bevisstgjøringseffekt, men utover det synes ikke denne tilnærmingen å bidra mye til prioriteringsspørsmål eller utforming av virkemidler. Økonomisk verdsetting er et nyttig verktøy dersom det for eksempel kan hjelpe til å (i) anslå om nytten av tiltak betydelig overstiger kostnadene (ii) anslå verdien av naturskader i rettsaker (iii) eller predikere hvordan økonomiske insentiver slår ut i ulik arealbruk på tvers av arealbrukere, eller om den kan brukes til å anbefale differensierte insentivert.

9.2 Anbefalinger for videre arbeid

- Gjennomføre en konsekvensvurdering av verdsettingsstudier som er utført over de tre siste tiår i Norden og vurdere hvilke forhold som har ført til anvendelse av verdsettingsestimatene i politikkutforming
- Gjennomføre komparative verdsettingsstudier mellom nordiske land for videre uttesting av veiledere for anvendelse av økonomisk verdsetting i EUs vanndirektiv
- Gjennomføre verdsettingsstudier som er representative for den nordiske befolkningen på nasjonalt og lokalt nivå – med generiske tiltaksbeskrivelser – og for ulike kulturelle tjenester etter mønster fra tidligere NMR-finansierte prosjekt for fritidsfiske (Toivonen m.fl. 2000)
- Finansiere ytterligere steds- og politikkspesifikke verdsettingsstudier i utvalgte nedbørfelt i Norden, spesielt av regulerende tjenester med anvendelse av egnede metoder som for eksempel produksjonsfunksjons- og skadefunksjonsmetodene
- Initiere verdsettingsstudier som eksplisitt vurderer romligheten i ØT-verdier og hvordan de avhenger av avstand, retning, romlig skala og oppløsning. Vurdere implikasjoner for anvendelse av verdsettingsestimater i nasjonalregnskap, nytte-kostnadsanalyser og virkemiddelutforming

- Demonstrere muligheter og begrensninger i å skalere opp verdsettingsestimater fra vannforekomst og vassdragsnivå til anvendelse i kapitalregnskap for økosystem (som ledd i grønt nasjonalregnskap)
- Støtte forskning om utforming av økonomiske virkemidler under vanndirektivets forvaltningsplaner for nedbørfelt (for eksempel betaling for ØT), der man vurderer økologisk effektivitet og nytte av resulterende endringer i ØT, tekniske så vel som transaksjonskostnader ved implementering, fordelingsmessige hensyn og legitimitet, institusjonelle barrierer og muligheter for implementering
- Støtte utvikling av nordiske visualiseringer og illustrasjoner av ØT – på nordiske språk – for å øke offentlig bevissthet om ØT
- Støtte litteraturstudier lignende Valueshed for andre nordiske økosystemer (f.eks. skog, fjell, kyst og åpent hav), der man også vurderer gjensidig avhengigehet av økonomiske verdier av ØT mellom økosystemer (for eksempel regionale verdier av skog), som supplement og oppfølging til nordiske TEEB

10. Appendix 1 Valuation methods

10.1 Benefit –based valuation methods

10.1.1 Introduction

Private goods are both excludable and rival in consumption, while public goods are typically neither. Since the consumption of private goods by one individual reduces the amount of the good available for consumption by others, and people can be effectively excluded from using the good, private goods can be valued through market prices. Non-market valuation methods are needed for economic assessment of impacts on public goods like environmental quality, for example water quality. Non-market valuation methods try to elicit individuals' (or households') preferences for public goods through their behaviour in markets for private goods which are related to the public goods (i.e. revealed preferences – RP), or their behaviour in constructed, hypothetical markets (i.e. stated preferences – SP) for the public goods.

In this chapter we give a brief description of available non-market valuation methods and how they can be adapted to assess the value of ecosystem services from watersheds.

For a detailed discussion of the concept of Total Economic Value (TEV) in the context of ecosystem services see the TEEB report (Kumar et al. 2010). The alternative to new primary valuation studies is to transfer values from previous valuation studies. This practice is most often referred to as "*benefit transfer*", but since damage costs can also be transferred a more general term is "value transfer". While benefit transfer is less costly and faster than conducting an original study, the resulting values are more uncertain. Thus, we will also review benefit transfer techniques.

10.1.2 Non-market valuation methods

The welfare loss to households due to e.g. damages to ecosystem goods and services can be estimated based on individuals' willingness to pay (WTP) to avoid these damages, which is termed the Total Economic Value (TEV) of the change in this public good. The TEV can be divided into: (i) use value, motivated by individuals' actual use of the public good, and (ii) non-use values motivated by the wish to preserve the option for future use (i.e. option value), the wish to preserve the existence of the good (i.e. existence value), and being able to deliver the good to future generations (i.e. bequest value).

The different non-market valuation methods can be used to estimate some, or all, parts of the TEV. Thus, the economic valuation of environmental impacts would typically be based on individual preferences, either observed behaviour (revealed preferences) towards some marketed good with a connection to the non-market good of interest; or stated preferences expressed in surveys about the change in non-marketed goods.

10.2 Benefit –based valuation methods

This appendix summarizes methods that measure benefit-based preferences and the welfare provided by nature. These methods can either measure stated preferences or revealed preferences.

Revealed Preference techniques can be divided into direct and indirect methods. *Direct* methods include the use of *market prices* to value productivity gains in e.g. commercial fish catches due to reduced pollution. This approach would in its simplest form use an assessment of the physical effects based on reported reduced damages from fishermen or some general dose-response relationship between e.g. how much less severe the pollution is, when it occurs and how long it lasts; and the annual gain of different fish catches. The observed market prices of the affected fish species are then multiplied by the magnitude of the physical or biological response to obtain a monetary measure of damage. Thus, neither behavioural adaptations nor price responses are taken into account. Simple multiplication provides an accurate estimate of economic behaviour and value – in this case changes in gross revenue – only if economic agents are limited in the ways in which they can adapt to the environmental effect, if the effect is small enough to have little or no impact on relative prices, and if the market prices are not distorted by market imperfections. This combination of circumstances is rather unlikely. Thus, this approach should be used with great care and clearly stated assumptions. Indirect methods include travel costs and hedonic pricing. Production function approaches are discussed in a separate section below as they are broadly relevant for valuation of ecosystem services as inputs to both market and non-market goods.

The *Travel Cost (TC)* method has been widely used to measure the economic value of recreational activities. TC method relies on the assumption that people make repeated trips to recreational sites until the marginal utility derived from a trip equals the marginal costs of a trip. The marginal costs are travel costs in terms of time cost and transportation cost. These travel costs can be regarded as a directly revealed preference for recreation and an indirectly revealed preference for nature.

The TC method assumes that the demand for trips to a specific site is dependent on travel costs, income, characteristics of the site, availability and prices of substitutes, etc. In this way a demand curve for the site is derived.

The costs of travelling to a recreational site (e.g. watershed, beach, river, national park) together with participation rates, visitor attributes, and information about substitute sites are used to derive a measure for the use value of the recreational activity at the site.

Travel can be used to infer the demand for recreation, only if it is a necessary part of the visit, or in economic terms is a *weak complement*. TC models builds on a set of strict assumptions, which are seldom fulfilled, and the results are sensitive to the specification of the TC model, the choice of functional forms, treatment of travel time and substitute sites etc. However, the TC method can be relatively cheap to use (compared to SP methods), and give reasonably reliable estimates for *use* values of natural resources (e.g. recreational use values of beaches, rivers, national parks etc) for the *current* quality of a site.

Most applications of the Travel Cost Method (TCM) have been to value recreational sites. If the "market" for visits to a site is geographically extensive, then visitors from different origins bear different travel costs depending on their proximity to the site. The resulting differences in total cost, and the differences in the rates of visits that they induce, provide a basis for estimating a demand curve for the site (Boardman et al. 2001). TCM measures revealed preferences for natural sites in terms of willingness to pay for site visits. As such, it includes the consumers' surplus. The TCM has often been applied for water recreation, fishing, wetland visitations and hunting.

10.2.1 Hedonic Pricing Method

Hedonic Pricing Method (HPM) is based on the idea that market goods are often traded at prices in which natural amenities are internalised. For example, the price of a house or a summerhouse in beautiful surroundings overlooking the sea is likely to be higher than the price of the same kind of house without a seaview.

The HPM starts with a regression of house prices against all their valuable characteristics.

A model of the factors affection house prices can be written as follows:

- Price (house) = f (architecture, size, view, contents, amenities, local taxes, neighbourhood attributes, etc.)

From this function one can calculate the willingness to pay for a marginal change in each of these explanatory variables. This is the implicit price of the amenity under investigation. From these implicit prices, the demand curve for a specific amenity can be derived. The demand curve is then used for estimating the economic value of an amenity such as natural beauty.

The HPM has often been used to measure the (negative) values of noxious facilities and for the value of environmental goods such as air quality improvement or noise reduction. This approach has not been widely used to value watershed related ecosystem services.

HP refers to the estimation of implicit prices for individual attributes of a market commodity. Some environmental goods and services can be viewed as attributes of a market commodity, such as residential property. For example, proximity to noisy streets, noisy airports and polluted waterways; smell from hog operations, factories, sewage treatment plants and waste disposal sites; exposure to polluted air, and access to parks or scenic vistas are purchased along with residential property. Part of the variation in property prices is due to differences in these amenities. Another application (termed Hedonic Wage models) has been to analyse wages for different jobs that entail different levels of mortality risks, to reveal how much people must be compensated in higher wages to take on a job with a higher occupational mortality risk. These wage and risk differentials can be combined to estimate the Value of a Statistical Life (VSL), which can be viewed as the economic value of preventing a fatality.

HP data can be quite costly to collect, as there is often no database of residential properties, that have detailed data on attributes of the property and its environment, including public goods, which determine the property price. The HP function is very sensitive to the specification and functional form, and it is often difficult to find a measure for the environmental amenity where data exist, and which the bidders for residential properties can recognize marginal changes in and has complete information about at the time they bid for the property. However, as an approximation one could use the assessed value (or market prices where they exist, corrected for market imperfections), conduct field surveys to register characteristics that could have a potential effect on property prices, and e.g. use distance to the river as a proxy for flood risk.

The HPM measures revealed preferences and it includes the consumers' surplus as it measures the total area underneath the derived demand curve. The validity of the method may however be questioned because the shape of the hedonic price function is not known. It is also possible that there are several amenities that influence the price of a house in opposite directions. There may, for example, be a positive influence of a river nearby, but at the same time two noxious facilities which supply jobs. It is also possible that the house market is distorted due to governmental interventions (Pearce and Markandya, 1989). Since the number of explanatory variables can be numerous, one runs the risk of not including an important variable or encountering multi-collinearity and thus drawing spurious conclusions about the value of an amenity.

HPM has a very large data requirement because both primary data (characteristics of the surroundings) and secondary data (market transactions) need to be collected. The value of a house depends on many factors: there are social factors, such as employment opportunities, taxes and accessibility. Data needs to be gathered for all these factors. This makes HPM less suited as a tool for decision support.

Although HPM can be used to value amenities such as natural beauty like sea view, this may not be enough to capture the total economic value of a natural site. Beauty is only one attribute of a natural site. The HPM was not developed to determine the total value of nature, but to determine the value of amenities as e.g. natural beauty only)

Stated Preference Methods

While (indirect) RP methods are based on actual behaviour in a market for goods related to the non-market good in question (and thus the value of the non-market good is elicited based on sets of strict assumptions about this relationship), SP methods measure the value of the environmental good in question by constructing a hypothetical market for the good. The hypothetical nature is the main argument against SP methods. SP methods have the advantages of being able to measure the TEV including both use and non-use value, derive the "correct" Hicksian welfare measure, and can measure *future changes* in the quality or quantity of water and other public goods.

The SP methods can be divided into direct and indirect approaches. The direct Contingent Valuation (CV) method is by far the most used method, but over the past few years the indirect approaches of Choice Experiments (CE)/Choice Modelling have gained popularity. The main difference between these two approaches is that while the CV method typically is a two-alternative (referendum) approach, CE employs a series of questions/choices between two alternatives (and often a status quo alternative) described by different characteristics/attributes including the cost of providing the alternative. CE is designed to elicit preferences and values for different attributes of the public good, while CV values the good "as a whole".

10.2.2 Contingent Valuation Method

A CV survey constructs scenarios that offer different possible future government actions. Under the simplest and most commonly used CV question format, the respondent is offered a binary choice between two alternatives, one being the status quo policy, the other alternative policy having a cost greater than maintaining the status quo. The respondent is told that the government will impose the stated cost (e.g. increased taxes, higher prices associated with regulation, or user fees) if the alternative to the status quo is provided. The respondent then provides a "favour/not favour answer" with respect to the alternative policy (versus the status quo): What the alternative policy will provide, how it will be provided, and how much it will cost, and how it will be charged for (i.e. payment vehicle), have to be clearly specified. This way of eliciting willingness to pay is termed binary *discrete choice (DC)*. In such a closed-ended version of CV, respondents can also be asked to value multiple discrete choices in double- and multiple DC WTP questions. Another elicitation method is *open-ended* questions where respondents are asked directly about the most they would be willing to pay to get the alternative policy. A payment card with amounts ranging from zero to some expected upper amount are often used as a visual aid. Then the data could be treated statistically as interval data; i.e. if you say "yes" to pay 50 US $ as the highest amount, but say "no" to 100, we know that the respondent has a WTP within this range. One of the main challenges in a CV study is to describe the change in the non-market good the new policy/program will provide in a way that is understandable to the respondent and at the same time scientifically correct. Another challenge is to find a realistic and fair payment vehicle, which does not create protest behaviour, in terms of people stating zero WTP even if they have a positive WTP for the change in the non-market good (e.g. due to strong dislike of taxes).

Concerns raised by CV critics over the *reliability* of the CV approach led the US National Oceanic and Atmospheric Administration (NOAA) to convene a panel of eminent experts co-chaired by Nobel Prize winners Kenneth Arrow and Robert Solow to examine the issue. In January 1993, the Panel, after lengthy public hearing and reviewing many written submissions issued a report which conclude that "CV studies can produce estimates reliable enough to be the starting point for a judicial or administrative determination of natural resources damages – including lost passive use value" (Arrow et al. 1993). The Panel suggested guidelines for use in Natural Resource Damage Assessment (NRDA) legal cases to help ensure the reliability of CV surveys on passive use values including the use of in-person interviews, a binary discrete choice question, a careful description of the good and its substitutes, and several different tests should be included in the report on survey results. Since the Panel has issued the report, many empirical tests have been conducted and several key theoretical issues have been clarified.

The simplest test corresponds to a well-known economic maxim, the higher the cost the lower the demand. This price sensitivity test can easily be tested in the binary DC format, by observing whether the percentage favouring the project falls as the randomly assigned cost of the project increases, which rarely fails in empirical applications. The test that has attracted the most attention in recent years is whether WTP estimates from CV studies increase in a plausible manner with the quantity or scope of the good being provided. CV critics often argue that insensitivity to scope results from what they term "warm-glow", by which they mean getting moral satisfaction from the act of paying for the good independent of the characteristics of the actual environmental good. There have now been a considerable number of tests of the scope insensitivity hypothesis, and a review of the empirical evidence suggests that the hypothesis is rejected in a large majority of the tests performed (Carson 1997). Thus, most CV studies seem to pass this test.

Producing a good CV survey instrument requires substantial development work; typically including focus groups, in-depth one-to-one interviews, pre-tests and pilot studies to help determine whether people find the good and scenario presented plausible and understandable. The task of translating technical material into a form understood by the general public is often a difficult one. Adding to the high costs of CV surveys is the recommended mode of survey administration being in-person interviews (Arrow et al. 1993). Mail, internet and telephone surveys are dramatically cheaper, but mail and internet surveys suffer from sample selection bias (i.e. those returning the survey are typically more interested in the issue than those who do not) and phone surveys have severe drawbacks if the good is complicated or visual aids are needed.

CV results can be quite sensitive to the treatment of potential outliers. Open-ended survey questions typically elicit a large number of so-called protest zeros and a small number of extremely high responses. In discrete choice CV questions, econometric modelling assumptions can often have a substantial influence on the estimated mean and median WTP. Any careful analysis will involve a series of judgmental decisions about how to handle specific issues involving the data, and these decisions should be clearly noted. The reliability of estimates and validity of results depend on the design and implementation.

CV is the most widely applied valuation method and has been used for valuation of a variety of watershed services, as is discussed in several chapters of this report.

Choice Experiments (CEs) have been employed in the marketing, transportation and psychology literature for some time, and arose from conjoint analysis, which is commonly used in marketing and transportation research. CE differ from typical conjoint methods in that individuals are asked to choose from alternative bundles of attributes instead of ranking or rating them. Under the CE approach respondents are asked to pick their most favoured out of a set of three or more alternatives, and are typically given multiple sets of choice questions. Because CE are based on attributes, they allow the researcher to value attributes as well as situational changes. Furthermore, in the case of damage to a particular attribute, compensating amounts of other goods (rather than compensation based on money) can be calculated. This is one of the approaches that can be used in Natural Resource Damage Assessments (NRDAs). An attribute-based approach is necessary to measure the type or amount of other "goods" that are required for compensation (Bennet and Blamey 2001). This approach can provide substantially more information about a range of possible alternative policies as well as reduce the sample size needed compared to CV. However, survey design issues with the CE approach are often much more complex due to the number of goods that must be described and the statistical methods that must be employed.

10.3 Production function approaches

The production-function approaches (PFA) are based on the fact that natural resources, processes and qualities are used as "factors of production" or "inputs" to manufactured goods and services (Boardman et al. 2001). Production-function approaches aim at valuing natural qualities by exploiting the relationship between environmental attributes and the output level of an economic activity, assessing change in productivity or effect on production. As such this valuation method, has potential in addressing the valuation of "intermediate" ecosystem services, in particular regulating and supporting services.

The underlying assumption is that when an environmental attribute enters a firm`s production function, economic impacts of the environmental changes may be measured by the effect on production. Such effects can be valuated at market (or shadow adjusted) output prices.

PFAs have been widely used, particularly to evaluate the impacts of environmental quality changes (e.g. acid rain or water pollution) upon agriculture (e.g. Adams et al. 1986) and fisheries (e.g. Kahn 1991). Other examples of application include analysis of the impacts of water diversion (Barbier 1998), and the valuation of the protection benefits provided by coastal wetlands against hurricane damage (Farber 1987).

A PFA consists of a two-step procedure. The first step is aimed at identifying the physical impacts of environmental changes on a production activity. The second step consists of valuing these changes in terms of the corresponding change in the activity's output. Particularly the first stage requires co-operation between natural scientists, economists and other researchers, in order to determine the nature of the linkages between production and the environment (Barbier, 1998) – in essence quantification of an ecosystem service contribution to final output.

If Y is the activity's output, ENV_j the environmental variable(s) of interest, and X_i (i=1……N) other inputs, the production function of a representative firm might look like:

- $Y = f(X_i, ENV_j)$ (1)

If $\delta Y/\delta ENV$ is positive, then an increase in ENV_j (e.g. water quality) will increase output.

When Y is a marketed good, and the observable price is not affected by relevant market-failures, this price (or a shadow adjusted price) can be used to estimate the value of a change in ENV. Alternatively, this value can be estimated by looking at the changes in marketed inputs (X_i)

required to maintain a given level of output. It is worth noting that this applies to manufacture and provision of both final goods and services. As such "production" is a rather restrictive name for the valuation method when applied to ecosystem services.

All the production function approaches require the (non-trivial) identification of the physical relationship f(.) between environmental variable (ecosystem service indicator ENV_j) and the output level of Y. ENV_j can be understood as the indicator of an intermediate ecosystem service into production/provision of a good/service that is demanded. Considering that many (most?) ecosystem services can only be enjoyed using some human effort, input or technology (X_i), this model defines most ecosystem services as "intermediate" in the sense of Boyd and Banzhaf (2007). Linking the indicator of ecosystem service ENV_j to measurable biophysical components of ecosystems (ES_k) that may be subject to management and policy is an extension of the production function method to include ecosystem function modeling:

- (2) $ENV_j= s(ES_k)$

Quantifying the functional relationship requires the non-trivial modeling of the ecosystem function s(.).

Various quantitative methods have been used to estimate the economic costs (or benefits) of environmental changes affecting production activities. Following Hanley and Spash (1993), these methods can be classified as follows:

- "traditional" type models (or "historical approach"). Its main advantage is that the informational requirements regarding valuation are relatively modest. The monetary value of the environmental change is estimated by multiplying the output change by the current output price. The main caveat of this method is that it ignores possible price changes. Although prices may be unaffected by marginal environmental changes, significant and widespread changes in environmental conditions could entail not-negligible price effects, so that the assumption of constant price could provide seriously biased welfare measures
- optimization models; require extensive data sets, but provide more detailed information, and allow indirect effects to be considered. Quadratic programming models allow treating both price and quantities and endogenous variables. However, because of their normative nature assuming that agents are economically rational optimizers, discrepancies may emerge between the model solutions and reality, and identifying the source of such discrepancies may prove difficult
- econometric models. Econometric models do not adopt a normative approach, but, by using observable data, and their variations over

space or time (or both), they try to get factual evidence about the interrelationships of interest. "This applied work is objective in the sense that the results can be rigorously examined using accepted scientific and statistical methods, although ideological bias can be expected both in the selection of questions investigated and in the inferences drawn from factual evidence" (Hanley and Spash,1993, p.106)

A number of more general problems may also arise when applying a production function approach (PFA). Following Barbier (1998) these potential drawbacks may be summarized as follows:

- The *price* of the output can be heavily distorted, i.e. it may fail to provide a reliable proxy of the output's "true" economic value. Besides market failures, prices may be distorted by fiscal policies (taxation or subsidization)
- Public regulatory policies (or the absence of appropriate regulations) defining *use rights* may influence the values imputed to the environmental input (ENV_j). For example, when considering the impacts of an environmental change, say a change in a coastal wetland supporting an offshore fishery, if the latter is subject to open-access conditions, "rents in the fishery would be dissipated, and price would be equated to average and not marginal costs. As a consequence, producer surplus is zero and only consumer surplus determines the value of increased wetland area" (Barbier, 1998, p.8)
- Applications of production function approaches may be most straightforward in the case of a natural resource supporting only one economic activity (single-use resources) than in the case of *multiple-use resources*. Ecosystems typically contain many "natural resources" and multiple users. When a natural resource supports many different users, "applications of the production function approach may be slightly problematic […] and assumptions concerning the ecological relationships among these various multiple uses must be carefully constructed to avoid problems of double counting and trade-offs between the different values" (Barbier, 1998, p.8)
- For some valuation problems, choosing whether to incorporate *intertemporal aspects* of environment can be very important" (Barbier, 1998, p.9). For example, in their study aimed at estimating the value of estuarine wetlands and mangroves in supporting offshore fishery in the state of Campeche (Mexico), Barbier and Strand (1998) adopted and compared, a "static valuation approach", and a "dynamic valuation approach". The former valuation exercise assumes that fish stocks are always constant. The latter attempts to model the impact of a change in coastal wetland area on the growth function of the intertemporal fish harvesting process

In general, the production function approach will often underestimate the benefits being estimated (Boardman et al. 2001). Production function approaches allow one to determine the value of the production capacity of nature. For example, a reduction of water quality may cause the fish population to decline and consequently reduce the income from fisheries. At the same time the poorer water quality may cause a rise in the production costs of drinking water (i.e. increased treatment costs). The total economic value of nature does, however, comprise more attributes than fish production and water purification, and therefore the production function approach can only capture part of the total economic value of nature. This approach cannot capture non-use values of nature.

10.4 Cost-based valuation methods

These are second best methods for valuing ecosystem services when non-market valuation and production function approaches are not possible. Consequently they are very common for valuing regulating and supporting services. Broadly speaking the cost-based methods observe the real costs, including expenditures and use of time, that interests/users make in adapting to changes in ecosystems (environmental and resource change). These valuation methods have been in use for some time due to their simplicity (Hufschmidt, James et al. 1983). Adaptation can take place in preparation of a change, during the change, and after the change, providing three slightly different cost-based approaches, prevention, avoidance and mitigation costs. For the sake of exposition we use examples from flood risk and water quality deterioration discussed in this report:

Opportunity costs of landuse are often discussed as a separate cost-based valuation method. Generally, opportunity costs are the same as foregone net income from not choosing other options. In ecosystem service valuation, opportunity costs of conservation refer to the net income landowners forego from productive activities when conserving or restoring natural systems and their ecosystem functions. These are typically costs incurred by "upstream" landowners in managing land and water so that e.g. benefits of flood reduction and water quality can be enjoyed downstream. If costs are incurred in conservation they are "preventive". If costs are incurred in restoration they are "mitigative". Different authors use different classifications, but the essence lies in an adequate description of the decision alternatives and their costs and benefits relative to a baseline or status quo.

Prevention costs – in expectation of flooding downstream inhabitants may construct flood walls, making capital investments, using labour and other inputs, and requiring maintenance expenses that are all valued at market prices. In expectation of worsening water quality, households and municipalities may invest in additional water treatment equipment.

Costs for contingency planning for natural hazard emergencies can also come under this heading.

Avoidance costs – during a flood or due to water quality deterioration residents may also incur costs by moving away and avoiding negative impacts, if prevention has not been effective, and avoidance costs are lower than staying in place. Avoidance may also be an ex ante strategy, if respondents have sufficient information about risks (e.g. through flood zone mapping).

Mitigation costs – during and after a flooding event or a water quality deterioration event such as an algal bloom, users may incur costs to reduce the negative consequences of what has taken place.

Damage costs. Conversely, when land and water upstream is not managed with downstream interests in mind, or simply for natural hazard reasons, these downstream interests may suffer damages. Damage costs have legal and insurance interpretations relative to what is admissable damage. Perception of damage may also depend on what is "normal" and what perceptions are about rights to e.g. water quality or flood risk avoidance by downstream users. Damage costs are reduced by prevention, avoidance, mitigation actions.

The cost-based methods are overlapping and interdependent and should perhaps for that reason be seen as a single approach. For welfare economists cost-based methods are considered inferior because they are conservative estimates of welfare that do not describe consumer surplus. For example, from an observation of flood wall construction by residents we can only assume that the expected benefits to residents behind the flood wall are at least as great as the costs of building the wall. Cost-based methods also assume that the agent incurring the cost is also the agent with interests at stake, and so the action reflects preferences directly. Preventing risk to health and property often involves public works (flood walls, waste and drinking water treatment plants) which are not directly undertaken by households or businesses at risk. Evaluating whether costs are a good conservative proxy for welfare therefore involves evaluating how well preventive or mitigative measures by public bodies or organisations represent interests of those at risk.

10.5 Deliberative valuation methods

The TEEB "approach" has opened up the consideration of "value" to include both monetary and non-monetary expressions of human welfare (or conversely biophysical-cultural and non-biophysical-cultural). Kumar (2010) discusses deliberative methods that are aimed at systematically documenting non-monetary values. This large class of qualitative social science research methods goes beyond the scope of this report. We briefly discuss multi-criteria analysis (MCA) as one of these methods that has been used both as a qualitative deliberative method, and as a method for indirect monetary valuation of e.g. flood risk remediation. MCA is a "bridging method" between individual-focused monetary estimates of impact, biophysical modelled impacts and group-based deliberative descriptions of impact. A schematic description of steps in multi-criteria analysis applied to evaluation of streamflows and flooding is prvided in Barton et al. (2010).

MCA has been used to compare multiple impacts on ecosystem services that are evaluated biophysically, but not monetised, with impacts on business and households that can be monetised. Multi-criteria analysis is particularly convenient for evaluating multiple trade-offs between multiple uses of ecosystem services. As such it addresses some of the methodological limitations of the production function approach. MCA used in this way is also subject to the non-trivial task of quantifying the multiple relationships – damage functions – between indicators of ecosystem service and user interests. To address this Berge, Barton et al. (2010) use deliberation with experts to determine "pressure-impact curves" describing the functional relationship between an ecosystem service indicator, such as water level, and multiple user interests and key species indicators (PIMCEFA – " pressure-impact multi-criteria environmental flow analysis"). Barton et al. (2010) used the method to evaluate trade-offs between hydropower generation income, wetland habitat quality indicators, and other wetland user interests.

10.6 Valuation and value transfer issues

Value Transfer (VT) 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" is now most commonly used, while earlier studies often used the term Benefit Transfer (BT). In this report we will generally use "value transfer" as the most general term, but occasionally the term benefit transfer will be used, for example if the literature quoted or referred to, used BT. This section builds on Magnussen and Navrud (2008).

Lack of time and money to do new primary studies drive, and hopefully justify, the use of VT. However, VT implies increased uncertainty and potential errors of the value estimates. We need to know how we can test for such errors, what transfer errors are acceptable, and how they can be reduced.

There are three basic requirements for value transfer: i) Database with primary valuation studies; ii) Criteria for assessment of the quality of primary valuation studies, and iii) Methods for Value Transfer.

There are different approaches to value transfer, (and also different ways of categorizing the approaches; the listing below builds on Navrud (2008)).

Unit Value Transfer

- Simple (naïve) unit transfer
  a) Use value: Consumer surplus/activity day
  b) Non-use value: WTP/household/year
- Unit transfer with income adjustments
- International transfer: Purchase Power Parity (PPP) -adjusted exchange rates (which accounts for the different levels of costs of living in different countries)

Function Transfer

- Value Function Transfer (from one or a few similar studies)
- Meta-analysis (from many studies with different scope in terms of size of the environmental change (and different baselines and availability of substitute sites)

Unit value transfer with income adjustment

Adjusted value estimate B_p' at the policy site:

- $B_{p'} = B_s (Y_p / Y_s)^{ß}$
- B_s primary value estimate (e.g. WTP) from study site,
- Y_s, Y_p income levels at the study and policy site, resp.
- *ß* income elasticity of WTP for environmental good

GDP per capita (i.e wealth in society) or respondents' income (i.e. individual wealth) can be used in order to make adjustments for income.

Value Function (VF) Transfer

- *VF:* $WTP_{ij} = b_0 + b_1 G_j + b_2 H_{ij} + e$
- WTP_{ij} = willingness to pay of household i at site j,
- G_j = set of characteristics of environmental good at site j,
- H_{ij} = set of characteristics of household i at site j

Meta Analytic (MA) Transfer:

- MA: $WTP_s = b_0 + b_1 G_j + b_2 H_{ij} + b_2 C_s + e$
- WTP_s = mean willingness to pay/household of study s
- C_s = set of methodological characteristics of study s
- n = number of studies
  (but also several estimates from each study)

In the unit value approach the unit value at the study site is assumed to be representative for the policy site; either without or with adjustment for differences in income levels between the two sites (by using GDP per capita or purchase power parity indices).

Brander et al. (2010) note that unit value transfer means that mean unit values estimated at the study site for an environmental good or service is transferred to the policy site as the "policy site unit value".

Since unit values most commonly are expressed as values per household or values per unit of envionmental good (e.g. area) the aggregated value measure is calculated by multiplying the unit value with the relevant population or the units of environmental good (e.g. area of the valued ecosystem).

In adjusted unit transfer one makes adjustments to the transferred unit values in order to reflect differences in site and population characteristics.

In the function transfer approach a value function is estimated at the study site and transferred to the policy site, or a value function is estimated from several study sites using meta-analysis. Then values for the independent variables at the policy site are used in the function to calculate WTP at the policy site.

A value function from a CV survey expresses WTP as a function of the characteristics of site, good and respondent. Meta-analysis will typically also include methodological characteristics of the different studies as a variable, because methodological differences often influence results.

So far, there is no single universally adopted methodology used for VT. Rather there are different approaches used in different studies.

10.6.1 Transfer errors

Sources of transfer errors

Transfer errors arise when estimates from study sites are adapted to policy sites. These errors are related to the degree of correspondence between the study site and the policy site.

As Brander et al. (2008) put it: "Assume there is an underlying meta-valuation function that links the values of a resource (such as a lake) or an activity (such as swimming or recreational fishing) with characteristics of the markets and sites, across space and over time. Further, hypothesize that a primary research project samples from this meta-function. The meta-valuation function may be constructed as an envelope of a set of study site functions that relates site values to characteristics or attributes associated with each site, including market characteristics, physical site characteristics, spatial characteristics, and time (Rosenberger and Phipps 2002). The degree that any of these sets of factors affects value transfer accuracy is an empirical question. However; the greater the correspondence (or similarity) of the policy site with the study site is, the smaller the expected error (Boyle et al. 1992; Desvousges et al. 1992)".

Three general sources of error in the values estimated using value transfer (Brander et al. 2008):

- Errors associated with estimating the original benefit measures at the study site(s). Measurement error in primary valuation estimates may result from all the biases and inaccuracies associated with valuation methods in general
- Errors that arise from the transfer of study site values to the policy site. So-called generalisation error occurs when values for study sites are transferred to policy sites that are different without fully accounting for those differences. Such differences may be in terms of population characteristics (income, culture, demographics, education etc.) or environmental/physical characteristics (quantity and/or quality of the good or service, availability of substitutes, accessibility etc.). This source of error is inversely related to the correspondence of characteristics of the study and policy sites. There may also be a temporal source of generalisation error in that preferences and values for ecosystem services may not remain constant over time. Using value transfer to estimate values for ecosystem services under future policy scenarios may therefore entail a degree of uncertainty regarding whether future generations hold the same preferences as current or past generations
- The last source of error Brander et al. discuss is "publication selection bias" which may result in an unrepresentative stock of knowledge on ecosystem values. Journals publish mainly methodologically interesting papers while high-quality, empirical studies which contributes more to our knowledge of ecosystem values do not get published and exists only as working papers, Ph.D. theses, research reports to national EPAs etc

Transfer errors are generally expressed as the Mean Absolute Percentage Error (MAPE), which is defined as (observed value – predicted value)/ observed value (Brander et al. 2008). The measurement of transfer errors is not accurate itself because usually the study site values are treated as "true" values and compared to the transferred values as approximations. However, they are in fact both approximations.

Table 13 shows that errors in individual transfers vary a lot, both within and between different validity tests and for all transfer methods. Since some of the transfer validity tests are performed under ideal conditions (i.e. same SP survey instrument used on a similar good in a nearby location at the same point in time; e.g. Bergland et al. (1995)) they might underestimate transfer errors in practical transfer exercises. However, surprisingly many of these validity tests are performed under less than ideal conditions, and probably reflect quite well the transfer errors in practical value transfers. Several of the studies listed in Table 14 support the hypothesis that the greater the correspondence, or similarity, between the study site and the policy site, the smaller the expected error in benefit transfers. Lower transfer errors resulted from in-state transfers than from across-state transfers (Loomis 1992; Van den

Berg et al. 2001). This is potentially due to lower socioeconomic, socio-political, and socio-cultural differences for transfers within states, or political regions, than across states. In the Loomis et al. (1995) study, their Arkansas and Tennessee multi-site lake recreation models performed better in benefit transfers between the two regions (percent errors ranging from 1% to 25% with a nonlinear least squares models and 5% to 74% with the Heckman models) than either one when transferred to California (percent errors ranged from 106% to 475% for the nonlinear least squares models and from 1% to 113% for the Heckman models). This suggests that the similarity between the eastern models implicitly accounted for site characteristic effects.

Table 13. Summary of benefit transfer validity tests for environmental goods (reproduced from Navrud 2007)

Reference		Resource/Activity	Unit value Transfer Percent Error[28]	Function Transfer Error
Loomis (1992)		Recreation	4–39	1–18
Parson and Kealy (1994)		Water / Recreation	4–34	1–75
Loomis et al. (1995)	Nonlinear Least Squares Model	Recreation	---	1–475
	Heckman model		---	1–113
Bergland et al. (1995)		Water quality	25–45	18–41
Downing and Ozuna (1996)		Fishing	0–577	---
Kirchhoff et al. (1997)		Whitewater Rafting Birdwatching	36–56 35–69	87–210 2–35
Kirchhoff (1998)	Benefit Function Transfer	Recreation/Habitat	---	2–475
	Meta-analysis Transfer		---	3–7028
Brouwer and Spaninks (1999)		Biodiversity	27–36	22–40
Morrison and Bennett (2000)		Wetlands	4–191	---
Rosenberger and Loomis (2000a)		Recreation	---	0–319
Van den Berg et al. (2001)	Individual Sites Pooled Data	Water quality	1–239 0–105	0–298 1–56
Shrestha and Loomis (2001)		International Recreation	---	1–81

[28] All percent errors are reported as absolute values

Table 14. Examples of estimated transfer errors in water related studies reported in Bateman et al. (2010)

Study	Estimated benefits	Transfer errors (%
Loomis (1992)	Sport fishing benefits	5–40
Parsons and kealy (1994)	Water quality improvements	1–75
Loomis et al. (1995)	Water based recreation	1–475
Bergland et al. (1995)	Water quality improvements	18–45
Downing and Ozuna (1996)	Saltwater fishing benefits	1–34
Kirchhoff et al. (1997)	White water rafting benefits	6–228
Morrison and bennett (2000)	Wetlands	4–191
Rosenberger and loomis (2000)	Water recreation	0–319
VandenBerg, Pou and Powell (2001)	Water quality	0–298
Barton (2002)	Beach bathing water quality	11–26
Barton and Mourato (2003)	Beach bathing water quality	9–129
Brouwer and Bateman (2005)	Flood control benefit	4–51

Van den Berg et al. (2001) show accuracy gains when they transfer values and functions within communities that have shared experiences of groundwater contamination than transferring across states, within states, or to previously unaffected communities.

Brouwer (2000) suggests that if non-use values are motivated by overall commitment to environmental causes, they may tend to be relatively constant across populations and contexts. In a contingent valuation survey of the national populations in all Nordic countries Kristofersson and Navrud (2005) found that transfer errors are consistently smaller for the non-use value of a preservation plan for Nordic freshwater fish stocks. The results for a non-use value scenario by non-anglers in Norway and Sweden produced average transfer errors below 20%. Use values for anglers showed higher transfer errors. It may be that non-use value in these two countries is motivated by similar factors and is relatively context independent, while use value is more context-specific. Clearly, this could be different for other environmental goods, particularly if the good has higher cultural significance in one country (or part of a country).

To summarize, the transfer validity studies conducted to date show that the average transfer error for spatial value transfers both within and across countries vary substantially. Individual transfers could have errors as high as 100–300%. Function transfer does not seem to perform better than unit value transfer. Meta-analyses could also produce high transfer errors, and only those with a limited scope in terms of similar type of environmental goods and similar type, state-of-the-art methodology, should be used. The validity tests also support the hypothesis that it is preferable to find a study site located close to the policy site of interest. The closer the study site is to the policy site, the more likely that both the good being valued and the user population affected will be similar, and therefore the transfer errors would be lower. Transfer validity tests also suggest that transfer errors are smaller if people have had experience with the environmental good in question, but the transfer errors do not seem to be lower for use than for non-use values.

Acceptable transfer errors

An important question in discussing transfer errors is whether the transfer errors are acceptably low for policy uses. To answer this question, the policy makers could compare the costs of doing a new study with the expected costs of making the wrong decision when using the benefit transfer estimates.

Example of Practical Value Transfer Guidelines for Cost-Benefit Analysis

There are few detailed guidelines on value transfer. In the US there exist guides that cover the key aspects of conducting a value transfer, notably Desvousges et al. (1998) aimed at transfer for valuing environmental and health impacts of air pollution from electricity production. Adapted to the economic valuation of environmental goods in general Navrud (2007) propose the following eight steps guidelines:

- Identify the change in the environmental good to be valued at policy site
- Identify the affected population at the policy site
- Conduct a literature review to identify relevant primary studies (based on a database)
- Assessing the relevance and quality of study site values for transfer
- Select and summarize the data available from the study site(s)
- Transfer value estimate from study site(s) to policy site
- Calculating total benefits or costs
- Assessment of uncertainty and acceptable transfer errors

10.6.2 Value Transfer and scaling up

In this section we will discuss issues of particular relevance for scaling up in the meaning of spatial aggregation of transferred values.

Markets for ecosystem services and distance decay effect

This issue is relevant for value transfer regardless of scaling up, but may be of particular importance in scaling up, because the errors involved may be "scaled-up" as well if these issues are not handled properly.

In the following we will draw on Brander et al. (2008 and 2010) who include an interesting discussion on this topic. They note that the distance between a person and an environmental good like wetlands (in Brander's case), lakes and rivers etc. can be an important explanatory variable of this person's WTP for that good. Transferring average WTP values from a study site where the relevant population is located close to the site to a policy site where the population lives much further away is thus likely to lead to overestimation of total WTP. Since the distribution of the population is likely to differ between the policy and study sites, average distances between individuals and both sites are different, and value transfer studies should account for these differences.

Based on economic theory, the effect of distance on WTP is expected to be negative (at least for use values), indicating a distance-decay (DD) effect. DD implies that the WTP for a certain site decreases as the distance from the agent to the site increases. In other words, use values are expected to be decreasing with distance, because the costs of visiting a site increase with the distance you have to travel. The higher the distance, the higher are the costs, and the lower is the demand. One of the main reasons to include this DD effect is to determine the size of the geographical boundaries (market size) of the environmental good in question. This relevant market is the population over which the WTP values can be aggregated to calculate the good total economic value (TEV).

Besides direct use values, non-use values are an important component of TEV of any environmental good. The importance of distance for reliable value transfer or aggregation therefore depends on the type of value that a study site generates. There is no reason within standard economic theory why non-use values would also decrease with distance. The spatial discounting literature states that values that relate to what economics call non-use values should have much lower discount rates than use values (Brown et al. 2002). The extent to which distance is important for reliable value transfer therefore also depends on the type of values generated by the study and policy sites.

Other cases in which a distance decay effect is likely to occur are for goods that have importance on a large scale. In this case the DD effects are likely to be very small or negligible, meaning that even very far from the site, people are still willing to pay. The fact that something is either of national importance, of symbolic meaning or has the status of national parks implies that there are likely to be fewer substitutes. On the other hand, whenever goods have local importance due to some cultural association with the good, WTP is likely to fall beyond that political or social boundary. Examples are DD effects for "local "goods, suggested to be due to a "sense of ownership" (Bateman et al. 2004; 2010) or "spatial identity" (Hanley et al. 2003).

For non-unique sites, such as a lake in a lake district, the number of substitutes is high, lowering the WTP for a particular site. As the distance to a site increases, the number of available substitutes is likely to increase as well – especially for local goods. However, substitution effects alone cannot always explain DD effects.

Distance can be specified in many different ways and for reliable transfer or aggregation, the specification should be consistent. Approaches differ in objective versus perceptual or subjective distance and a straight line or based on the road net/ travel distance, using more sophisticated GIS applications. Travel cost studies typically use GIS based distance calculations, assuming that people minimize their costs by choosing the shortest route. However, for non-use values, which form a large share of the value of many environmental goods, the least cost travel route does not matter and other specifications might be reliable. Another issue is to which part of the asset the distance should be measured. Ideally, the distance from individual A to the nearest access point of a site should be used for use-values. However, the larger the site is, the more difficult it becomes to determine the distance.

Substitute and complementary sites

Brander et al. (2008, 2010) like Brouwer et al. (2009) note that one of the most important contextual factors in a value transfer exercise is the availability of substitutes. Ignoring substitutes means that if the transfer is performed between a landscape poor in ecosystem services to a landscape rich in ecosystem services WTP values are likely to be overestimated. The question is what happens to the WTP for good A if the quality in a comparable good B increases. A substitution effect in economics is usually defined as the increase of demand for good A when the price of good B increases. Again this issue is relevant for all VT cases, but treated as an issue of particular interest to the scaling up discussion.

The consequence of disregarding substitutes is generally an overestimation of WTP, as the sum of the value of goods measured individually is higher than the value measured for all goods at once. For instance, respondents in an area with several lakes whose water quality is polluted will value cleaning up the first lake more than cleaning up the second lake, because the first lake can be a substitute for the second lake and the respondents has a budget limitation which reduces the money available for cleaning up the second lake. Valuing goods separately and then adding up the values will overstate the true value, as every respondent will treat the ecosystem under study as if it were the first good.

Disregarding complementary sites causes underestimation of WTP. Complementary occurs when goods are consumed jointly, for instance when two sites are visited during the same trip, or when there are synergy effects in production, for instance when quality increases at one site automatically increase the quality of another site due to dependent ecosystems. The WTP of one site is therefore likely to be dependent on other available alternatives and their characteristics. As distance from the site or the geographical scale of the study increases, the number of complements is likely to increase.

In the economic geography literature, the spatial distribution of goods over the study area is addressed by including an indicator of accessibility. Fotheringham (1988) argues that if the WTP of both sites is dependent on distance, the substitution effect will be dependent on the relative distance between the sites. Just including distance from the agent to the substitutes therefore does not account for the proximity of substitutes, the spatial structure, and will lead to biased WTP estimates. However, no clear example of environmental valuation studies account for such spatial structure.

In order to determine the relevant substitutes for a certain environmental good, different criteria have been used:

- All available similar ecosystems in the study area or within a certain range, or
- All similar ecosystems known or visited by the respondent; or
- All nature sites in the study area; or even
- All possible recreation areas (not necessarily nature based)

Scaling up over what?

Regarding scaling up over the size of the affected population, one has to deal with several questions. One has to consider the uniqueness of the good in question, is it of local, regional or national importance, and what then, is the "relevant" population. One has to take account of the availability of substitutes and their quality and the distance decay effects in WTP as we have discussed above. One should also consider whether estimates should be aggregated over households rather than individuals (to avoid overestimating WTP), but this may depend on the unit of value estimate in the primary study.

Regarding scaling up over environmental good/service (like the relevant ecosystem) one often finds that the *unit of valuation* needed for policy making (e.g. hectare of an ecosystem) is not the same as those directly meaningful to ecologists; or how people think about environmental goods and services (which is what determines the unit used in SP surveys). Further one should be aware that often the discrete changes are valued (providing average values per unit of area), while there are marginal values that are needed. Further, marginal values are not constant; and the baseline quality/quantity matters for valuation results. Even if the choice is to aggregate at the ecosystem level, that leaves us with many options, as we have discussed to some degree already. Valuation studies and thus transfer of values has been conducted both at the ecosystem (hectare) level, and at the level of individual species for example.

Navrud (2008) notes that meta-analysis could be potentially very useful when scaling up due to the variability in size, quality, ecosystem functions, baseline quality, availability of substitutes etc. of primary studies included. The quality of meta-analysis will depend on the number of explanatory variables and explanatory power of the estimated Meta-analytic regression model (which could be improved if the scope of the analysis is narrowed in terms of domestic vs. international studies, valuation methods included, definition of ecosystem etc.).

One often used approach is to do scaling up over the size of the ecosystem, however some studies like Lindhjem (2007) in a meta-analysis of 30 studies in Norway, Sweden and Finland of mainly non-use values of coniferous forests found that WTP does not vary with size of forest area. This implies that transfers and scaling up-exercises using value per hectare will be biased. Hence we need to test validity of meta-analytic VT (and construct more primary studies with VT in mind).

Concerning meta-analysis and transfer errors, Lindhjem & Navrud (2007) tested the reliability of MA of non-timber benefits for international BT and found that the mean transfer error (MA-BT) was 47–126% (while simple unit transfer error was 62–86%. This would probably often not be sufficient accuracy for policy use. More MA of primary studies from other countries and other environmental goods needed before final conclusion can be drawn on MA for BT.

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11. Appendix 2 Stepwise conceptual approach to valuation of watershed services

This report follows a "TEEB approach" while applying specific recommendations from the AQUAMONEY Technical Guidelines for Practitioners.

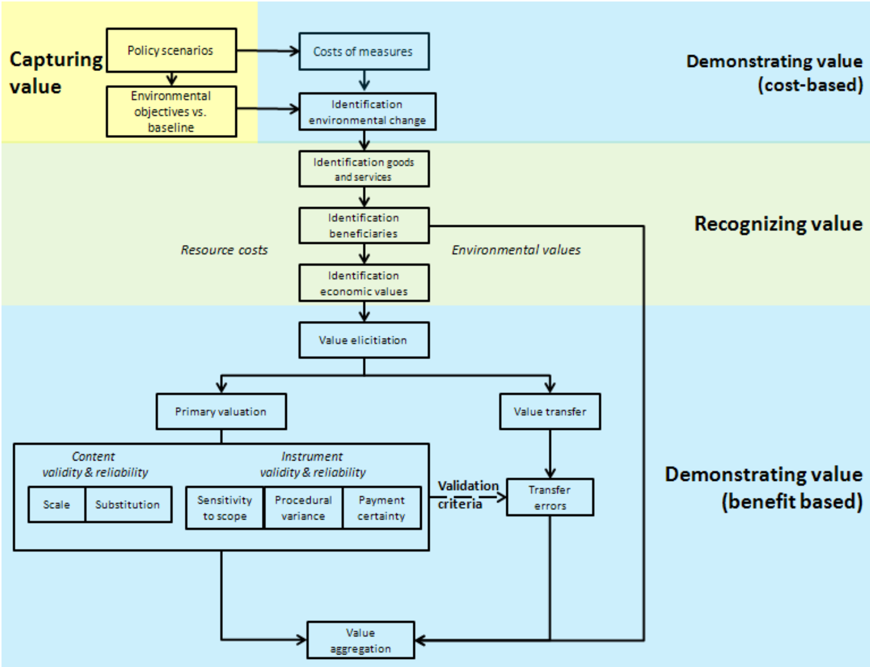
TEEB approach
The TEEB approach refers to a specification of the role that economic valuation of ecosystem services can play in awareness raising and policy decision-support. TEEB refers to:

- *Recognizing value* – involving identification of ecosystem services and their welfare effects reflecting a plurality of values, one of which may be monetary
- *Demonstrating value* – assessing the costs and benefits of biophysical changes resulting from alternative management options, highlighting trade-offs, and seen as the principal role of economic valuation
- *Capturing value* – involving the evaluation and introduction of mechanisms that incorporate values of ecosystem into decision-making, through incentives and price signals

We use the conceptual framework the AQUAMONEY Guidelines' steps originally designed for primary valuation of the benefits of achieving Water Framework Directive objectives (Brouwer et al. 2009). We found it useful to organize the case study discussion according to these steps in order to make them more comparable.

In this appendix we give some further details on the stepwise approach to valuation.

Figure 28.Iillustrates a methodological framework which also encompasses cost-based approaches within the TEEB approach. In this review we will use this AQ-UAMONEY-TEEB framework in reviewing selected case studies for two chosen "demonstration" or example watersheds in Norway and Denmark



Adapted from AQUAMONEY's General steps in economic valuation and key issues (Brouwer, Barton et al. 2010).

Both benefit-based and cost-based economic valuation methods are reviewed in this report. We set out with the aim to *demonstrate the scope of values of* ecosystem services in watersheds, *given reasonable and available future land and water management scenarios* for the chosen watersheds. While the aim of the study is not policy design capturing these values, policy-relevant valuation requires some definition of future policy scenarios, and so demonstration of values has some relevance to policy, provided the scenarios chosen are credible. It is also important to note that economic valuation is most usefully applied to differences (or changes) in ES between alternative (policy) scenarios, rather than to total values of ES.

Furthermore, cost-based approaches to economic valuation require specifying the change from a baseline to environmental objectives, for which costs are to be assessed. Again, valuation has bearing on policies setting those objectives.

In the following we briefly discuss the AQUAMONEY steps for primary (original) valuation studies. For our review and "demonstration case studies" AQUAMONEY's stepwise (Step 1–10) approach to valuation of watershed services will be adapted depending on the case study and data availability. In certain cases it will be reasonable to discuss some steps under the same heading depending on the detail of available data. However, we find it useful to present the steps individually and as they should ideally be addressed in a valuation study.

11.1.1 Step 1: Policy scenarios as basis for valuation

A starting point for valuation studies applied to policy analysis is the identification of policy scenarios and a baseline. As is seen in the case study examples in this report, existing valuation studies are often specific to a particular baseline and policy-context. In the context of watershed management what are possible approaches to the definition of baseline and policy scenario alternatives?

Definition of a baseline

Projected land and water use based on Water Framework Directive (WFD) River Basin Management Plans (2009–2015) for selected catchments or more long term forecasts of drivers have been proposed. For example, Braat and ten Brink (2008) used OECD Environmental Outlook to 2030 as the basis for information about future economic and demographic development, combined with the IMAGE-GLOBIO model to project changes in terrestrial biodiversity to 2050. Costanza et al. (1997), used a worst case scenario of catastrophic loss of ecosystem services as a baseline for valuing the benefits of conserving the worlds' ecosystem services, subject to a number of critiques. Studies focusing on specific watersheds must define the baseline also as a function of what kind of policy the valuation results will be used to evaluate. What is the expected timeframe within which effects of the policy are expected in the waterbodies concerned? This will also be defining for how far into the future the baseline needs to be projected and compared to the policy scenario alternative.

Policy (in) action leading to catastrophic loss. This type of policy scenario assumes that ecosystems are completely converted or lost within the watershed due to large scale landuse change or infrastructure. This is similar to a "total economic valuation" approach of Costanza et al. (1997). The advantage of this type of baseline assumption is that there is no need for dose-response modeling of incremental impact.. Disadvantages include quite simply that in most cases this is not a credible baseline, especially in the context of Nordic watershed management. Furthermore, there are problems of double counting because whole ecosystems and their services cannot be assumed to disappear without a cascade of effects in other ecosystems and markets off-site.

CBD policy inaction. Through COP10 CBD commitments Nordic countries in the EU have recently committed to avoid biodiversity loss and ecosystem service degradation by 2020. This scenario would entail no loss compared to current negative trends for example summarized by the TEEB Cost-of-policy-inaction (COPI) study (Braat and ten Brink

2008). An advantage of this baseline is that it is a credible policy scenario in the short to medium term. Disadvantages include that baselines must be downscaled from COPI study projections per biome, to ecosystem services in particular Nordic catchments. Alternatively, scenarios of negative biodiversity trends due to predicted landuse change must be constructed "from the bottom up" in the catchment in question. A large GIS modeling effort is required.

WFD policy inaction and status quo. Here a study will assume that no further effort is made to comply with WFD objectives of "good ecological status" in natural water bodies and "good ecological potential" in heavily modified water bodies. Water bodies not complying with WFD are projected to continue in the same state as they are currently in. Valuation then refers to the foregone benefits of WFD incompliance in selected Nordic watersheds by River Basin Management planning deadlines 2015 or 2027. Valuation addresses an improvement relative to current negative trends in most water bodies. This is the contingent scenario for a number of the more recent valuation studies, such as AQUAMONEY case studies. The advantage with this type of scenario is that it is a credible in Nordic countries, while not focusing on the specifics of individual measures. Disadvantages include that valuation requires the use of dose-response modeling or studies explicitly designed to address WFD objectives.

Ex ante conditions for watershed specific projects. In a number of watersheds large or long term infrastructure projects could cause significant ecosystem service changes. Examples include development of reservoirs and water transfers of hydropower and their transmission lines, flood control walls, urban or recreational home development in natural areas, and even planned protected areas. Where valuation studies can use baseline scenarios from completed Environmental Impact Assessments (EIAs), there are opportunities for valuation of high relevance. The advantage of these types of policy studies are high local policy relevance. Disadvantages include the fact that in most cases EIAs identifying environmental change will not have been carried out.

Our review of valuation studies showed that many valuation studies of watershed services uses variant of the latter two approaches.

11.1.2 Steps 2–3: Definition of measures and identification of environmental change

Explicit definition of the environmental change under evaluation is the keystone of economic valuation. In case of benefit-based or non-market valuation methods, measures are often not explicitly defined, but assumed as part of the valuation scenario identifying environmental change. This has the advantage of respondents not focusing on the cost of measures themselves, but on the environmental improvement, when stating willingness to pay or other welfare measures. A comparison of the environmental change evaluated in the chosen watersheds (policy site) with what was specified at a study site, is nevertheless important in checking the validity of benefit transfers (Bateman et al. 2010) (Step 9).

For cost-based valuation methods, a definition of the environmental objective, and the cost-effective measures necessary to achieve them starting from the baseline is required. Cost-based methods require significant dose-response modeling and are not necessarily easier or cheaper to carry out than primary benefit valuation studies (Browuer et al. 2010; Barton et al. 2005). The extent to which we can use cost-based valuation methods depends on the availability of cost-effectiveness studies and dose-response models in the chosen watersheds. In the candidate watersheds identified for this study there are a number of examples of such available studies, but we are far from covering all ecosystem services of relevance in any one watershed.

11.1.3 Steps 4–6: Identification of goods and services, beneficiaries and economic values

Identification and description of the relevant goods and services is a key in valuation of ecosystem services. This seems obvious, but is often not as easily done as it may seem. The reason is that many watersheds are not mapped in detail, and even if they are described according to natural scientific procedures, this information is not always easily transformed to ecosystem goods and services – and particularly the quantification of these goods and services may be difficult. Still, this is necessary in order to be able to put monetary value on them (by new studies or benefit or cost transfer). In some cases this is not possible, and we resort to more verbal, qualitative assessments.

A key message of the Millennium Ecosystem Assessment (2005) and TEEB is the identification of economic trade-offs between ecosystem service providers and beneficiaries. This was also pointed out in TemaNord (2009). Based on identification of environmental land use or water use changes defined in the chosen watersheds a number of watershed functions are affected. These determine changes in goods and services to wetlands beneficiaries, and as such identify economic values at stake. A qualitative identification of ecosystem service providers and beneficiaries, given the specific watershed chosen and policy scenario defined earlier can be made using maps, diagrams or matrices (see Table 1, main report).

A potential weakness of the MEA and TEEB ecosystem services classification is its lack of distinction between intermediate and final services to end-users or beneficiaries (Boyd and Banzhaf 2007). This is clearer in the AQUAMONEY framework where a distinction is made between function (intermediate) and outcomes (final services). This is discussed in more detail for flood water and water pollution retention functions of ecosystems in the demonstration case study examples. In fact some clarification is also called for in our interpretation of ecosystem function, ecosystem service (intermediate and final) benefit and value.

11.1.4 Step 7: Value elicitation / demonstration

"Value elicitation" encompasses benefit-based non-market valuation methods using both primary on-site valuation studies (primary studies and data) and benefit transfer (secondary sources and data) – to this definition in the AQUAMONEY guidelines we have added cost-based methods (Figure 1, Appendix 2). With examples from Nordic watersheds, we aim to illustrate all three valuation approaches. Even though our aim is to value the goods and services in order to give economic estimates, this has not been possible for all identified goods and services, either because there are no relevant valuation studies to transfer from or because the existing valuation studies or the scientific knowledge of the watershed is not in a form that makes transfer possible (as noted under steps 4–6).

Primary valuation

The call for proposal for this project did not require conducting primary valuation studies. However, one of the principle criteria for choosing the Nordic "demonstration" watersheds is the availability of primary valuation studies from the watershed in question. Where such studies are available, we discuss opportunities and limitations in extrapolating them using benefits transfer.

Benefit transfer

Any choice of "demonstration watershed" will require "cross-watershed benefit transfer " and even "international benefit transfer", in order to provide examples of valuation estimates for ecosystem services relevant to any particular watershed. In this study we limit ourselves to providing examples of value transfers using Nordic studies. Such benefit transfers are subject to large transfer errors that in some cases may make the transfer unusable for policy analysis or even "demonstration" and awareness raising (Bateman et al 2010). We discuss the pros and cons of benefits transfer in Appendix 1.

Cost-based valuation

In many cases, our review reveals that benefits transfer is a priori too uncertain for demonstration purposes, or there is no valid study sites in Nordic countries or for similar wetlands and ecosystem services, from which to transfer values. In such cases we use where possible cost-based estimates and available dose-response hydro-economic models. Costs of achieving politically approved environmental objectives such as "good ecological status" under the WFD may be taken as a lower bound estimate of economic benefit of the environmental improvement from the baseline. In the case of dose-response and hydro-economic models they are catchment specific cost-estimates derived from the original study sites and should in principle not be transferred to other catchments, unless differences in catchment characteristics can be accounted for and transfer errors described.

11.1.5 Step 8: Value aggregation – demonstrating value

Value aggregation consists of adding marginal valuation estimates (annual per household, per user, per user day etc.) across the economically relevant population, and discounting values for the relevant assessment period defined in the policy scenario above. In exceptional cases estimates of distance decay of willingness to pay are available which may be used to determine the economically relevant population (those with willingness to pay>0). In most cases strong assumptions have to be made regarding the population for which estimates are to be aggregated (typically neighboring municipalities to wetlands, population with catchments, or populations closest to water bodies relative to credible substitute sites). As aggregation assumptions are crucial to determining total benefits of ecosystem services of a given policy scenario, they should also be subject to validation criteria (Step 9 below).

Aggregation of valuation estimates for ecosystem goods and services may not be straight forward and has to be made with much care. On the one hand, there is a chance of double counting, as some goods and services (like the supporting and regulation services) are mainly experienced as "non-use values" in people's welfare (but still, they are important to identify). Secondly, most existing valuation studies value a composite or bundle of environmental goods and services like recreation services of different "size" combined with non-use-values or "biodiversity conservation". Therefore aggregation (and disaggregation) of former valuation estimates to "match" ecosystem goods and services may be a challenge.

11.1.6 Step 9 Validation of valuation assumptions, estimates and evaluation of policy relevance

We have interacted with a reference group of Nordic environmental economists with a track record in Nordic valuation of ecosystem services. Where we have provided valuation estimates they have been asked to evaluate these using benefit transfer validity criteria discussed in amongst others the AQUAMONEY guidelines.

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Valuation of Ecosystem services from Nordic Watersheds – from avareness raising to policy support? (VALUESHED)

The emergence of the ecosystem services concept suggests that economic valuation studies are already fulfilling a role in raising awareness by demonstrating the loss of nature's goods and services using monetary indicators. In order to have future relevance in capturing value and giving support to policy-makers, valuation methods must specifically address resource accounting, priority setting, and instrument design.

This report provides an overview of economic valuation methods of ecosystem services from watersheds in the Nordic countries. The study was commissioned by the Nordic Council of Ministers and conducted by The Norwegian Institute for Nature Research, The Norwegian Institute of Water Research and Sweco Norge during the period May – November 2011.

The study indicates that economic valuation methods can be applied to watershed management in multiple ways. However, policy makers should be wary of "one size fits all" valuation estimates that appear ready to use across different watershed types and stakeholder interests.

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