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Economic Valuation of Wetland Ecosystem Services

Case Studies from the Elbe River Basin

Technische Universität Berlin

Economic Valuation of Wetland Ecosystem Services. Accounting for wetland ecosystem service benefits in cost benefit analysis of river basin management options with case studies from the Elbe River Basin.

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Zusammenfassung

Feuchtgebiete stellen eine Vielzahl von Ökosystemdienstleistungen zur Verfügung, wie z.B. die Erzeugung von landwirtschaftlich nutzbarere Biomasse, Erholungsmöglichkeiten, Lebensräume für genutzte und ungenutzte Bestände von Wildtieren und Fischen oder die Regulierung von Nährstoff- und Treibhausgasflüssen. Die Bewirtschaftung von Feuchtgebieten erfordert zwangsläufig Abwägungen bezüglich der Nutzung und Allokation der beiden wichtigen Ressourcen Land und Wasser, die zusammengenommen den Status der Feuchtgebietsökosysteme und deren ökonomischen Nutzen im weitesten Sinne bestimmen. Die in dieser Arbeit vorgestellten Fallstudien beziehen sich auf zwei typische mitteleuropäische Feuchtgebietstypen: Flussauen und Niederungsmoore des Tieflands. Bei der Eindeichung und Drainierung von Auen- und Moorstandorten sind Abwägungen bezüglich des Nutzens dieser wasserbaulichen Maßnahmen hinsichtlich einer Erhöhung der agrarischen Produktion und deren sozialen Kosten hinsichtlich verminderter Produktion von solchen Ökosystemdienstleistungen, die den Charakter öffentlicher Güter haben, erforderlich. Dies gilt in der Umkehrung auch bei der Wiedervernässung von Mooren und Deichrückverlegung an Flussauen. Gleiches gilt für die Allokation von Wasserressourcen in einer Flussgebietseinheit, bei der zwischen dem erzielbaren Nutzen an Feuchtgebietstandorte und bei anderen Wassernutzungen im Einzugsgebiet abgewogen werden muss.

Aus der Perspektive der ökonomischen Bewertung von Handlungsoptionen für die Bewirtschaftung von Feuchtgebietstandorten sowie der Wasserressourcen eines Flussgebiets ist das zentrale Problem die Bestimmung des Wertes der Ökosystemdienstleistungen, die zugleich öffentliche Güter sind. In diesen Fällen besteht eine Kluft zwischen dem Marktwert und dem volkswirtschaftlichem Wert der Leistungen des Ökosystems. Die beiden zentralen Themen dieser Arbeit sind daher empirische Methoden zur Bewertung von öffentlichen Gütern sowie Methoden zur systematischen Berücksichtigung des Nutzens von Ökosystemdienstleistungen in der Kosten - Nutzen Analyse von Bewirtschaftungsmaßnahmen.

Die Arbeit basiert auf acht Artikel mit Fallstudien aus dem Einzugsgebiet der Elbe. Die ersten fünf Aufsätze haben einen methodischen Schwerpunkt und jedes wendet eine andere Methode zur Bewertung einer bestimmten Ökosystemdienstleistung an: (1) der Nutzen von Biodiversitäts- und Lebensraumschutz wird mit Hilfe einer Meta-Analyse von Zahlungsbereitschaften die mit der Zahlungsbereitschaftsmethode generiert wurden bewertet, (2) der Erholungsnutzen mit der Reiskostenmethode, (3) der Nutzen der Hochwasserschutzwirkung anhand des vermiedenen Schadens, (4) die Senkenfunktion für Treibhausgase sowie (5) die Nährstoffretention anhand des Schattenpreis bzw. der marginalen Vermeidungskosten.

Die letzten drei Artikel präsentieren Fallstudien zu einer integrierten ökonomischen Bewertung durch im Rahmen einer erweiterten Kosten-Nutzen-Analyse. Die Analysen stützen sich auf die Bewertungsansätze die in den vorangehenden Arbeiten entwickelt wurden. Im ersten Beispiel wird eine strategische Kosten-Nutzen-Analyse von Deichrückverlegungsoptionen für die Flussauen der Elbe vorgestellt. Das zweite Beispiel bezieht sich auf die Bewertung von Bewirtschaftungsoptionen für wasserstandsregulierte Niederungsfeuchtgebiete anhand einer Fallstudie aus dem Spreewald. Das letzte Beispiel untersucht die Auswirkungen des Klimawandels für die Ökosystemdienstleistungen von Niederungsfeuchtgebieten im Rahmen eines integrierten ökonomisch – hydrologischen Modellierungsansatzes zur Bewertung der Wasserverfügbarkeit auf der Skalenebene des gesamten Einzugsgebiet der Elbe.

Summary

Wetlands provide a multitude of ecosystem services such as livestock fodder production, recreational opportunities, habitat and biodiversity conservation or regulation of nutrient and greenhouse gas fluxes. Wetland management inevitably involves trade-offs regarding the management and allocation of the two key resources, land and water, that taken together determine the status of wetland ecosystems and the potential flow of benefits to human wellbeing. The case studies presented in this thesis addresses two types of typical central European wetlands: river floodplains and lowland peat wetlands. Floodplain wetland management requires trade-offs between the benefits of conducting activities on the floodplain against the risk and adverse consequences to these activities caused by flooding and trade-offs between the benefits and costs of reducing this flood risk, for example through the conversion of active floodplains to protected floodplains by construction of dikes. The management of lowland peat wetlands, on the other hand, requires trade offs between the benefits of drainage and conversion of wetland land for increased agricultural production and the loss of benefits from other ecosystem services. Wetland land uses are interdependent with water regulation and all wetlands require sufficient water at the appropriate time to maintain their wetland status. Particularly lowland peat wetlands are dependent on the inflow of river water and constitute important water users within river basin that compete with other water uses. Trade offs are therefore also required regarding the allocation of scarce water resources both within the basin and within wetlands.

From an economic perspective, the key difficulty in determining whether restoration of wetlands or water allocation to wetlands are an appropriate policy or management goal lies in the difficulty of determining the value of the public benefits provided by wetlands. There is a gap between the market valuation and the economic value of many ecosystem services. The challenge of valuation of ecosystem services that have a public goods character and the integration of wetland ecosystem service benefits into the economic appraisal of river basin management options in a cost - benefit analytical framework are the two central issues of this thesis.

The thesis is based on eight papers with case studies from the Elbe River Basin (Germany). The first five papers have a methodological focus and each applies a different valuation method to the valuation of a specific wetland ecosystem services. These are the (1) the provision of habitats and biodiversity using stated preference methods and benefits transfer, (2) recreation using the travel cost method, (3) flood risk regulation using avoided damage method, (4) greenhouse gas regulation using a shadow price or alternative cost approach and (5) nutrient regulation using an alternative cost approach.

The final three papers present case studies of an integrated economic assessment using the framework of an extended cost benefit analysis. These analyses build on the valuation approaches developed in the previous papers. The first paper presents a strategic cost benefit analysis of floodplain management options for the Elbe River. The second paper presents an assessment of water management options for regulated lowland peat wetlands with a case study from the Spreewald wetland. The final paper presents wetlands as major water users amongst all other water uses within an integrated economic-hydrologic approach to assess the effects of climate change induced risks of low flows at the scale of the complete Elbe River Basin.

Content

Part A: Overview and synthesis

Economic valuation of wetland ecosystem services: an overview of concepts and applications

Part B: Manuscripts

ECONOMIC VALUATION OF WETLAND ECOSYSTEM SERVICES: AN OVERVIEW OF CONCEPTS AND APPLICATIONS

1 Introduction

This thesis, which is a collection of essays on the valuation of wetland ecosystem services, is located at the interface between the literature on "valuing water" and "valuing ecosystem services". Water and land management are closely interrelated in wetland management. Wetland management can be considered a prototypical example for what is meant by integrated water resources management, because it necessarily requires an integrative perspective on water and land management, on interconnections between upstream and downstream localities in river basins and on the various competing uses for water in a basin, of which the water demand for wetlands ("water for nature") is one.

The need for economic analysis for the design and implementation of efficient water resources management policies is well documented in the economics literature. This need is also emphasised in the European Union's recent Water Framework Directive (cf. WFD-CIS 2003a). The need to include the multifunctional nature of wetland water use and the ecosystem service benefits that are generated by wetlands in the assessment of water management options have been recognised in principle and are promoted, for example in guidance documents of the Convention on Wetlands of International Importance (cf. Ramsar Convention Secretariat 2007), the EU Water Framework Directive (cf. WFD-CIS 2003b) or the International Union for Conservation of Nature (Emerton and Bos 2004).

The conservation of wetland ecosystems is also the focus of a various national and regional conservation policies in Germany (cf. Schopp-Guth 1999, BMU 2007). Several state governments have developed strategies for the conservation and restoration of peat wetland habitats (cf. MLUV-MV 2009, LUA-BB 1997, Kowatsch 2007). These wetland conservation strategies are based on the three principles of strict protection of remaining natural wetland habitats; appropriate land management for modified wetlands under agricultural use that still have wetland characteristics and an increased effort to restore wetlands where feasible. Likewise, the Federal Ministry of Environment (BMU) and the Federal Agency for Nature Protection (BfN) actively promote the concept of an integrated approach to the management and development of floodplains (BMU and BfN 2009, Korn et al. 2006). Such an approach seeks to harness multiple benefits for flood protection, water resource management, nature and biodiversity conservation and climate change mitigation.

Historically, conservation in Germany has been justified primarily on an ecological science related discourse and ethical reasons. Economic justifications have moved more into focus with recent efforts to generate information on the economic implications of loosing nature and biodiversity. A major undertaking was the UN’s Millennium Ecosystem Assessment (MEA 2005), that was structured on the concept of ecosystem services and human welfare. In response to the economic arguments for an active climate change policy provided by the Stern Review (Stern 2007), similar efforts to analyse costs of biodiversity loss and benefits of preventive action, notably the Economics of Ecosystems and Biodiversity initiative (TEEB 2010) or the Cost of Policy Inaction Initative (COPI; Braat et al. 2008) have been initiated. The studies point to the absence of sufficient quantitative data of ecosystem service values that allow generalisations and transfer beyond the case study context. There is both a need for additional primary valuation studies and methods to transfer value estimates across different spatial scales that are required to appraise local, regional and national policies.

The available policy guidance literature for economic valuation of wetland ecosystems has closely reflected the development of valuation methods and in many cases wetlands have served as prototypical applications. The Ramsar Convention has long recognized the importance of wetland economic valuation in contributing to well-informed planning and decision-making, and in 1997 the Secretariat published "Economic valuation of wetlands: A guide for policy makers and planners" by Barbier, Acreman, and Knowler. State of art reviews on the thinking on valuation of wetland ecosystems services as time progresses can be found for example in the edited volume by Turner, van den Bergh and Brouwer (2003) on “Managing Wetlands. An Ecological Economics Approach” and the recent volume by Turner, Georgiou and Fischer (2008) entitled “Valuing Ecosystem Services. The Case of Multifunctional Wetlands”.

However, despite conceptual advances, in the practice of economic assessment of water management at a basin or sub-basin scale, the economic value of ecosystem service benefits provided by wetlands are still generally omitted. One of the reasons for this neglect lies in the difficulty and lack of experience in determining the value of public goods benefits. Recreational uses of wetlands, the conservation of water dependent habitats in wetlands or regulation of nutrient and greenhouse gas fluxes are typical examples of such public goods. Although there is some overlap, the valuation methods appropriate for public environmental goods differ from those for private goods. Over the years a substantial literature has developed that presents various applications of valuation methods for diverse benefits provided by wetlands. The meta-analysis on wetland valuation studies by Woodward and Wui (2001), Brander at al. (2006) or

Gherimandi et al (2008) provide summaries of the available studies to date. However, only few studies explicitly address the valuation of benefits as a function of water availability or water allocation towards wetlands. Such an approach is a prerequisite for the assessment of management options that affect the water availability for wetland sites in any water resources modelling framework (cf. Young 2005). The challenge of valuation of ecosystem services that have a public goods character and the subsequent integration of wetland ecosystem service benefits into economic appraisal of river basin management options are the central two foci of this thesis.

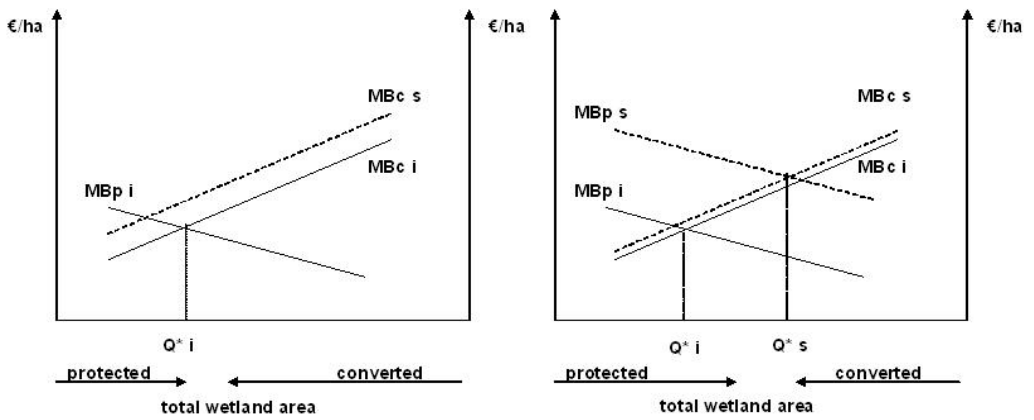
This introductory paper provides both an overview of key concepts that constitute the necessary background for understanding the issues addressed in the separate papers as well as a synthesis of the major findings and conclusions from these papers. This paper begins with an introduction of some general economic concepts that form the framework for the following detailed analysis. This includes an economic concept of wetland conversion and restoration, an overview of the ecosystem services approach and an introduction to some critical aspects of cost-benefit analysis. The second section focuses on methodological issues of ecosystem service valuation. This includes approaches to describe the production of ecosystem services and the valuation of the ecosystem service benefits. This introduction is followed by an overview of the key objectives and structure of the thesis and a short summary of the specific issues addressed in the individual papers. Finally, this overview closes with conclusions that can be drawn from the research program. Based on the research questions of the thesis, this section discusses key results and implications from a methodological and a policy perspective.

2 Ecosystem services and policy appraisal: basic economic concepts

2.1 Why value ecosystem services? Economic approaches to integrated wetland water and land management

Wetland management inevitably involves trade-offs regarding the management and allocation of the two key resources, land and water, that taken together determine the status of wetland ecosystems and the potential flow of benefits to human wellbeing. The case studies presented in this thesis addresses two types of typical central European wetlands: river floodplains and lowland peat wetlands. Floodplain wetland management requires trade-offs between the benefits of conducting activities on the floodplain against the risk and adverse consequences to these activities caused by flooding. It also requires trade-offs between the benefits and costs of reducing this flood risk, for example through the conversion of active floodplains to protected floodplains by construction of dikes. The management of lowland peat wetlands, on the other hand, requires tradeoffs between the benefits of drainage and conversion of wetland land for increased agricultural production against the associated loss of benefits from ecosystem services. Wetland land uses are interdependent with water regulation and all wetlands require sufficient water at the appropriate time to maintain their wetland status. Particularly lowland peat wetlands are dependent on the inflow of river water and constitute important water users within river basin that compete with other water uses. Trade offs are therefore also required regarding the allocation of scarce water resources both within the basin and within wetlands.

Wetland economics can therefore generally be understood in terms of balancing the marginal benefits of converting or protecting and restoring natural wetland land or allocating water in favour or disfavour of wetlands. To illustrate this, a stylized conceptual framework developed by Heimlich et al. (1998) can be used. This is presented in Figure 1. The horizontal axis represents the total stock of wetland sites. A large share of this initial stock has already been converted to agricultural land use by drainage or construction of dikes. The vertical axis represents an index of value.



Source: based on Heimlich et al. 1998

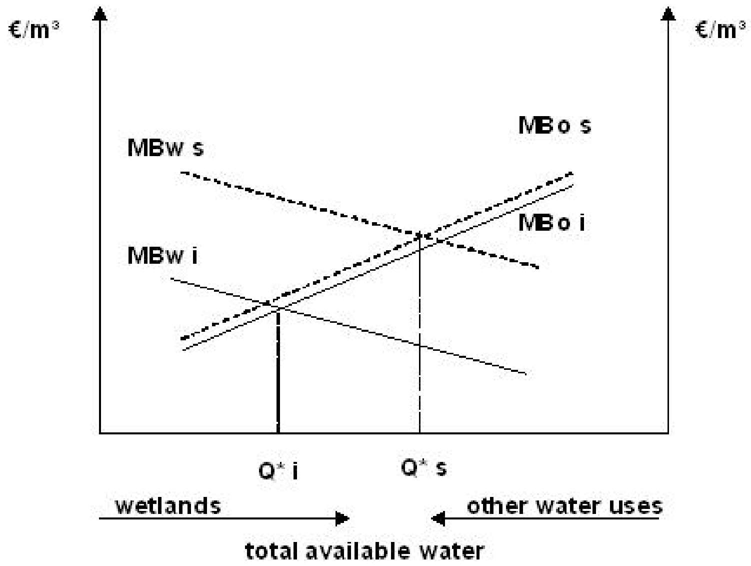
Figure 1: Optimal wetland conversion and protection

The net marginal private benefits (MBp i) that can be realized from protecting (or restoring) an incremental unit of wetland may be relatively low, since there are few benefits of wetland protection that landowners can capture in terms of private benefits. These may include benefits from extensive land use such as haying, grazing or forestry. The benefits considered here are net in that they include direct costs of conversion such as drainage but not the economic opportunity costs of not converting wetland. These are embodied in the net marginal private benefits (MBc i) to conversion. This benefit from drainage and embankment of an incremental unit of wetland may be relatively high as conversion makes possible intensive agricultural production and settlement development. MBc i would be expected to decline as the area of converted wetland increases –because sites that can be converted at low costs are converted first. The privately optimal allocation of the stock of wetland is represented by the point Qi*. At this point protecting an additional unit of wetland area would cost more in terms of forgone benefits from conversion than would be gained in benefits from protection.

This general framework can be used to illustrate a key issue in wetland economics: the difference between the public and private incentives to protect and convert wetlands. Both the protection and conversion generate public as well as private benefits. The public or private characteristics of goods and services can be defined along a continuum from rivalry to non rivalry of use and from excludability to non excludability of users.

The public benefits of conversion of natural wetlands may include increased agricultural production and lower consumer prices. Adding the incremental public benefits to the private benefits results in the social marginal benefit curve for conversion (MBc s). In contrast, most benefits from protection and restoration of natural floodplains are public in nature. Examples include flood control, water quality improvement, fish and wildlife habitat and recreational opportunities. Adding these public benefits to the individual benefits results in the significantly higher social marginal benefits curve for protection (MBp s). The socially optimal allocation of the stock of wetlands Q*s thus entails a higher share of protected wetlands than under the privately optimum allocation.

This framework can also be used to trace the historical trajectory of wetland land use policies. Historically in central European countries such as Germany, the private and public benefits of wetland conversion where perceived to be large and public incentives and investments were made to encourage floodplain conversion in order to promote economic growth. The public benefits of wetland protection were not generally recognized. Public policies where therefore designed to move land allocation towards an optimum at the intersection of MBpi and MBcs to the left of Q*i. As a result the larger share of German wetlands has been converted. For example about 80-90 % of the floodplain of the large German rivers Rhein, Elbe, Donau and Odra has been protected from flooding by the construction of dikes (Brunotte et al. 2009). Likewise ca. 95 % of German peat wetlands have been drained for agricultural purposes (Schopp-Guth 1999). Over the course of the 20th century the public benefits of wetland protection came to be more fully appreciated. This can partly be attributed to the increased scarcity of the remaining natural floodplain and wetland landscapes and habitats. In addition, it is only relatively recently that the significance of the regulating services provided by floodplain ecosystems has been clearly recognized. As a result society increasingly values conserving and restoring wetlands over converting them for private economic use. Public policies now are increasingly designed to induce a shift towards the socially optimal allocation at Q*s. This involves restoration of drained and embanked wetlands and an increasing stock of protected wetlands that provide typical ecosystem functions of natural wetlands.

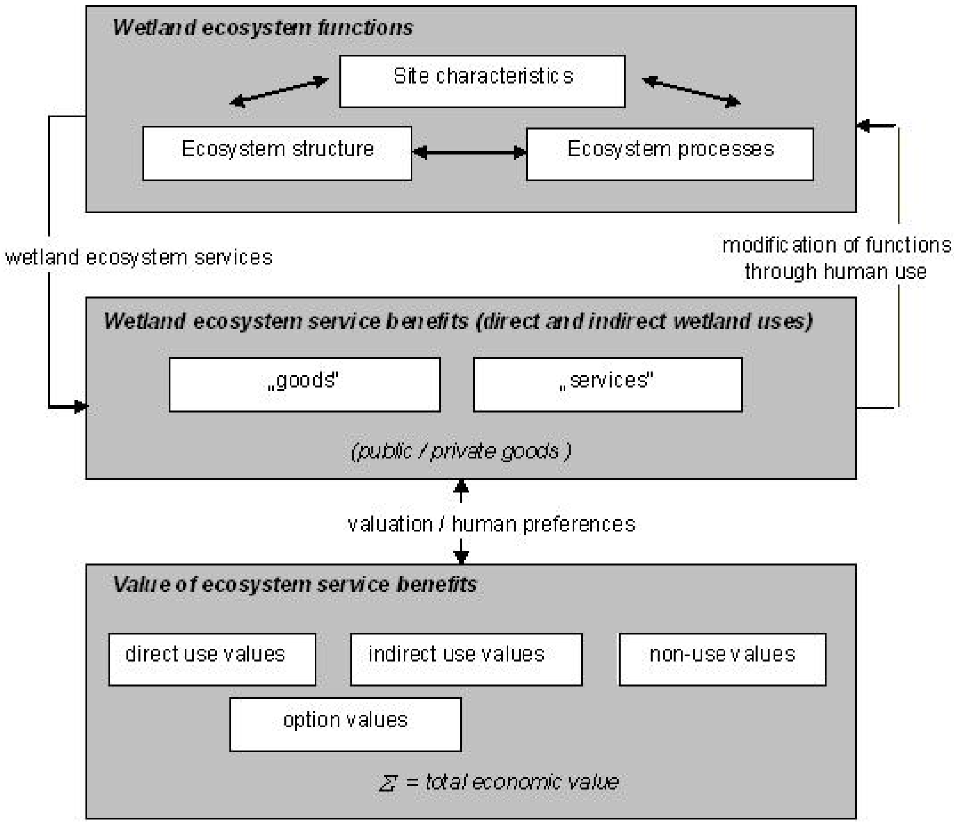


Source: own illustration

Figure 2: Optimal water allocation between wetlands and other water uses in a river basin

Beyond the allocation of land resources within wetlands, this analytical framework can also be used to explore the allocation of water towards wetlands in closed river basins, where water is a scarce resource. This is illustrated in Figure 2, which shows the marginal private and social benefits of an allocation of the available water resource to wetlands (MBw) and all other alternative basin water uses (MBo). Again, the marginal net benefits of allocating an incremental unit of river water to wetlands have a private and public benefits component that is determined, amongst others, by the wetland land management (share of drained and natural wetland area). The benefits of water allocation to most other basin water uses such as power generation, industry, irrigation or municipal water are largely private in nature. The socially optimal allocation of water Q*s will therefore be larger than an allocation based only on the private benefits Q*p. Again, as the benefits of wetland protection come to be more fully appreciated, the arguments for shifting water allocations in favour of wetlands to a social optimum Q*s are reinforced.

From an economic perspective, the key difficulty in determining whether the targets of a wetland restoration policy are appropriate lies in the difficulty of empirically determining the value of the public benefits (Heimlich et al, 1998). It is this problem that is addressed in this thesis. The provision of such information is essential if an efficient level of ecosystem resource conservation and restoration is to be determined. Maintaining or restoring wetlands is rarely costless, in most cases there are substantial opportunity costs associated with forgone other land uses. This underscores the importance of making explicit the value of the multiple services that wetland ecosystems perform and of assessing this value within a framework that allows comparison with gains to be made from conversion. Such an approach should serve to contribute to improved environmental decision making to the benefit of society at large. Economic valuation is therefore a logical extension to other assessment methods of the services provided by wetland ecosystems for the purpose of public decision making.



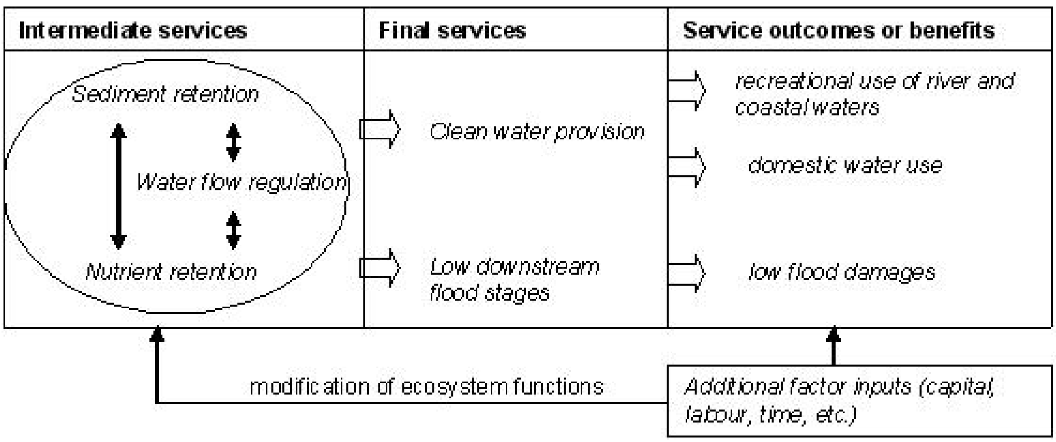
Source: modified from Turner et al. 2008

Figure 3: Conceptual framework: wetland functioning, uses and values.

2.2 The ecosystem services approach

It is well acknowledged in the environmental economics literature that the public goods characteristic has traditionally caused many ecosystem services of wetlands to be undervalued in the assessment of wetland management options. The concept of ecosystem services has become an important model to systematically link functions of ecosystems to human welfare (cf. Turner et al. 2008, NRC 2005). This concept builds on the conceptual differentiation of ecosystem functions (processes and structures), the uses and benefits that these functions support (goods and services) and the economic values of these goods and services. Figure 3 summarizes the conceptual framework for ecosystem service valuation based on ecosystem functions, uses and values. The concept of ecosystem functions and ecosystem services is the essential link between natural science approaches and economic approaches to describe interactions between ecosystems and the economic sphere.

Turner et al. (2008) propose that "ecosystems services are the aspects of ecosystems consumed or utilized to produce human well being either directly or indirectly". In this definition, ecosystem services are ecological phenomena and describe ecosystem structures and processes that are consumed by humans either directly or indirectly. Economic values are based on the functions of wetlands for which there is a perceived value to human beings. The ecosystem functions (structures and processes) have no economic value in themselves: the economic value is derived from the existence of a demand for the benefits they give rise to. In the ecosystem services approach as developed and outlined by Turner et al. (2008), services are designated to be either intermediate or final services, with human welfare only emanating from the final services. This is illustrated in Figure 4. This classification is advantageous for an operationalisation of the ecosystem services concept for economic appraisal, as it helps to avoid double counting the benefits from interdependent services, several of which may contribute to the production of a single benefit. By focusing on the final outcomes or benefits from ecosystem services and not the multitude of underlying services, the problem of double counting the benefits that each service contributes to, is avoided.



Source: based on Fischer et al. 2009

Figure 4: Conceptual relationship between intermediate and final services and service benefits

In order to describe ecosystem functions, ecosystems need to be characterised regarding their boundary conditions, their structure and processes. Ecosystem structures describe the biotic and abiotic characteristics of the elements of the ecosystem such as soils, water, vegetation and fauna. By contrast, ecosystem processes describe the dynamics of transformation of matter and energy. Ecosystem services are then the results of interactions among characteristics, structures and processes.

According to Turner et al. (2008), the services provided by wetlands can be categorized according to whether they are hydrological, geochemical or ecological services (Table 1). Hydrological services refer to the wetlands ability to regulate water and sediment flows. Examples are flood water detention, groundwater re- and discharge and sediment retention. Biogeochemical services refer to the transformation and storage of substances that can have significant effects on the quality of the environment. Examples are nutrient retention or greenhouse gas regulation. Ecological services relate primarily to the maintenance of habitats within which organisms live. Examples are habitat provision for plants and animals (for example feeding and resting habitat for migratory species, nursery habitats for fish) and the support of food webs inside and outside of the wetland through the production of biomass.

The ecosystem services provide human benefits in terms of direct and indirect benefits that derive from the utilization (or use) of the services (Fischer et al. 2009). However, one of the key points is that to realize the benefits from ecosystem service provision, typically other forms of input (capital, labour, travel time, skills, etc.). To illustrate this with an example: nutrient retention is an intermediate ecosystem service that human utilize indirectly – for example through the consumption of clean water for drinking or recreational experiences. Provision of clean water is therefore the final ecosystem service. Potable water and bathing opportunities are the benefits – that require additional inputs, such as abstraction and piping technology or travel to a bathing site to be utilized.

Finally, for cost benefit analysis, the benefits need to be translated into a monetary value. Economic values are dependent on individual human preferences. The economic value of a change in benefits from ecosystem services is defined as the amount of other resources that individuals are willing to forgo to obtain or prevent a change in benefits. Economic values are thus relative in the sense that they are expressed in terms of other benefits that are given up and they are related to incremental changes of the status quo (Young 2005).

Table 1: Examples of wetland ecosystem services with examples of underlying ecosystem functions and socio-economic service outcomes

Services	**Ecosystems function (structure or process) maintaining the service**	**Socio-economic service outcome or benefits**
Hydrological services		
Flood water detention	Storage of overbank water, reduction of flow velocity	Reduced flood damages
Groundwater recharge / discharge	Infiltration / seepage of water to / from groundwater	Enhanced water supply for different uses
Sediment retention	Sediment deposition	Enhanced soil fertility, reduction of channel sedimentation, improved water quality
Biogeochemical services		
Nutrient retention	Uptake of nutrients by plants, storage in soil, transformation and gaseous export	Improved water quality, sink for nutrient emissions from human activities
Carbon sequestration	Organic matter accumulation	Mitigation of climate change, peat for fuel / horticulture
Ecological services		
Food web support	Biomass production	Farm animal fodder, energy biomass, timber, reeds, fish & wildlife harvest
Habitat provision / landscape structural diversity	Habitat (permanent, nursery, migratory resting, etc) for plants and animals	biodiversity conservation, recreation, fishing, hunting, tourism

Source: based on Turner et al. 2008

2.3 Value concept and cost benefit analysis

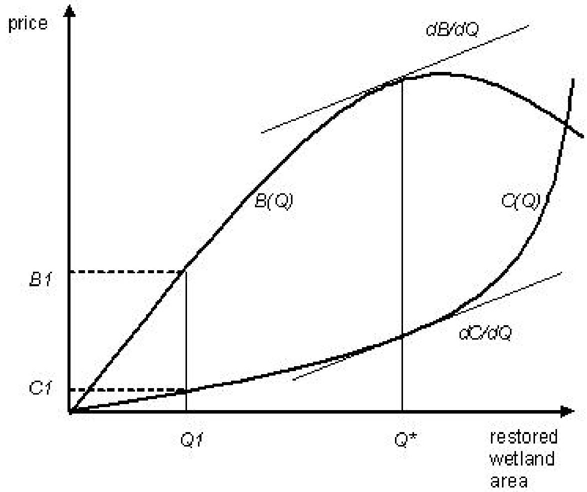
The valuation of individual benefits and the aggregation of benefits on the basis of monetary units in cost benefit analysis is based on economic welfare theory (cf. Hanley and Barbier 2009 or Young 2005 for application of welfare theory in cost benefit analysis of environmental policy). This derives the monetary value of utility from decision problems of individual households. According to this approach, each household seeks to maximise its individual benefit function subject to a limited household budget.

Application of welfare theory assumes that individual benefits correspond to the sum of available income. This assumption allows an aggregation of benefit levels from the individual to social welfare for society as a whole.

Within a cost-benefit analysis framework, the effects of a measure or project on the change in the aggregate utility of consumer goods (Δ CG) and the change in the costs (Δ C) are assessed. The level of consumer good availability has already been identified as a determinant of individual benefit. It is evaluated in terms of willingness to pay. Willingness to pay is a monetary measure of the intensity of individual preferences. The second component considered is the change in costs (Δ C), which corresponds to the change of resource use related to the implementation of the measure or project. This use of resources ultimately corresponds to a relinquishment of consumer goods which could otherwise have been produced using these resources. The change in real income or change in welfare (Δ W) is given by summation of the components: Δ W = Δ CG - Δ C.

Cost-benefit analysis of wetland management pertains to the evaluation of policies and projects regarding the objective of economic efficiency in the development, allocation and management of wetland land and water resources (cf. Hanley and Barbier 2009). While there are many other criteria along which to evaluate policies, under conditions of scarcity of land and water resources, economic efficiency becomes an increasingly important social objective. Economic efficiency is "an allocation of resources such that no further reallocation is possible which would provide gain in production or consumer satisfaction to producers or individuals without simultaneously imposing losses on others" (Young 2005). This definition of economic efficiency is termed Pareto-optimality. Parteo efficiency is achieved when the marginal benefits of using a good or service are equal to the marginal cost of supplying it. Because few policies would meet the strict Pareto standard of making no one worse off, in practice policies are evaluated against the compensation criterion that tests for potential Pareto improvements. According to this criterion (Kaldor-Hicks criterion), a policy can be considered as economically advantageous if the benefits are larger than would be required to in principle compensate losers.

Cost-benefit analysis applications typically examine rather large discrete increments of change to assess whether the move is in direction of Pareto efficiency (Young 2005). A policy which generates incremental benefits in excess of incremental costs is then considered to be Pareto superior. Figure 5 illustrates the comparison of Pareto efficiency and cost benefit criteria. The curve denoted B is a representation of aggregate benefits of increasing levels of wetland ecosystem service provision, while C represents the associated aggregate costs. Their general functional form reflects the conventional assumptions that benefits increase at a decreasing rate and that costs increase at an increasing rate. The Pareto efficient solution is Q* - the maximum vertical distance between B and C. At Q* the marginal benefits equal the marginal costs. However, rather than seeking the global optimum, cost benefit analysis in practice considers whether a project or policy would induce a desirable shift (for example from status quo to Q1). The conventional test therefore compares the aggregate incremental benefits (B1) with aggregate incremental costs (C1). If incremental benefits exceed incremental costs, then the change is a Pareto improvement.



Source: based on Young 2005

Figure 5: Parteo-efficiency and cost-benefit criteria compared.

The aggregation of the values of the benefits of the main ecosystem services provided by an ecosystem has been labelled total economic value (TEV) (Turner et al. 2008). The use of total economic value concept in the analysis of alternative management options has been developed to ensure that the full social benefits provided by wetlands are taken into account. This is necessary to indicate whether a wetland policy is associated with a true economic efficiency gain. The total economic value concept identifies value components that add up to the total economic value. The main distinction is made between use values and non use values. Use values can either arise from direct or indirect use of the wetland ecosystem services. Direct use values may be consumptive, as in harvesting of biota or non consumptive, as in recreational uses. In contrast, non use values reflect values that are independent or additional to the use of an ecosystem service by an individual, for example the satisfaction derived from the conservation of wetland habitats and biodiversity independent of any direct recreational use. In practical applications the assessment of total economic value is limited to those components that are both feasible to quantify and that are expected to be particular important elements of the total economic value in decision making context. The case studies presented in this thesis expand on the majority of existing cost-benefit assessments in that they account for a wider range of value components.

3 Ecosystem service valuation in practice: approaches and issues.

3.1 The assessment process

For the practical application of cost-benefit appraisal of wetland land and water management, the following basic analytical steps generally need to be taken: scoping the valuation issue, assessment of the impact on ecosystem service provision, valuation of benefits and generation of decision relevant information and criteria. These steps are summarised in Table 2. The following sections address methodological issues that are relevant for understanding the general approach of this thesis and that are related to (a) the methods for the quantification of wetland ecosystem service provision, (b) the methods for valuation of benefits and (c) the methods for generating information for decision making.

Table 2: Summary of the main steps of the cost benefit assessment process

1. Scoping the decision problem
- definition of policy or management options
- delineation of the affected ecosystem
- identification of the potential service provided by the wetlands
- identification of the groups of beneficiaries and the service benefit areas
- define costs and benefits to be considered for cost benefit analysis
2. Assessment of ecosystem service provision levels
- modelling of the service provision in baseline and management options
3. Valuation
- valuation of ecosystem service benefits in baseline and management options
- estimates of costs of management options
4. Generating information for decision making with cost benefit analysis
- set up tableau of costs and benefit in time
- calculate decision relevant criteria

3.2 Wetlands and water: mapping and modelling ecosystem service provision

Adequately specifying the production function of ecosystem services is at the basis of any assessment method. Determining a production function with a reasonable degree of accuracy is a challenge that can only be solved in an interdisciplinary approach. The key challenge for the valuation of wetlands is to determine the production functions as a function of water availability. The following section highlights some of the relevant key concepts that are important for this thesis.

Joint production and multi-functionality

One of the properties of wetlands is that each unit of wetland land and water in general produces more than one ecosystem service. Not only do wetland ecosystems deliver multiple ecosystem services, but ecosystem services can themselves provide multiple benefits – for example the water flow regulation service can generate a multitude of benefits for downstream water uses such as navigation, water supply or recreation. Ecosystem services therefore have characteristics of joint products and wetland land- and water use is multifunctional (cf. Turner et al. 2008). This is a difference to most other water using activities in a river basin context, where water is an input to the production of a single benefit, for example energy or irrigated crops. Two main reasons for the jointness of production of ecosystem services can be identified: interdependencies in the ecosystem service production process and non-allocable inputs (OECD 2001). Interdependencies in the production process are at the origin of many negative externalities of agricultural production on wetland soils, because water level regulation regulates various ecosystem functions or processes at the same time. The second type of jointness arises where multiple outputs are produced from the same, non allocable input – in this case land. While the ecosystem service generated from unit of wetland may be joint they are rarely produced in fixed proportions and those proportions can be modified by land and water management.

From a standpoint of economic valuation it is important to recognize that some wetland services are complementary or have complementary requirements regarding water and land management. However, others have competing requirements or are even mutually exclusive. For example, the optimal water levels for agricultural production and the greenhouse gas sink function is different, so that a trade off has to be made between these services when water regulation targets are defined. In summary, accounting for wetland value must recognize the multifunctional nature of wetland land and water use. At the same it is necessary to avoid double counting competing or mutually exclusive ecosystem services in too simplistic assessments, that add all possible benefits and ignoring that they may well be mutually exclusive.

Spatial concepts: spatial variability and locational interdependencies

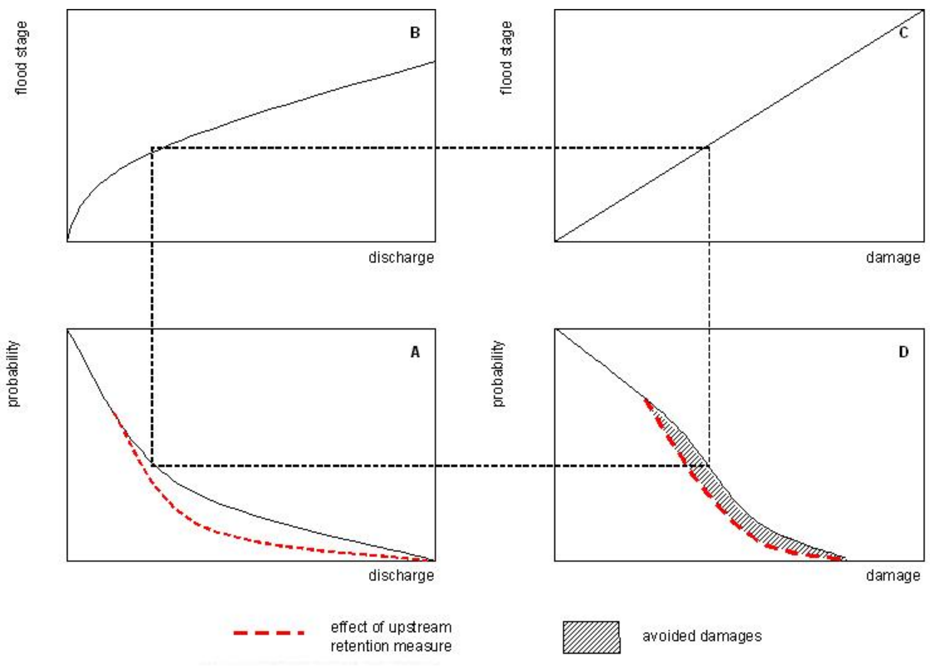
A further important aspect of ecosystem services is that their production is not homogeneously distributed across landscapes. The approach to systematically assess the effects of water as a site specific production factor taken in this thesis is to combine water dependent production functions for ecosystem services with spatially explicit hydrological models. The integration of wetlands into a river basin modelling framework is a prerequisite to adequately describe upstream - downstream interdependencies, both regarding the impact of water availability and basin management on the level of wetland ecosystem service benefits, as well as the downstream effects of changes in wetland management.

The central concept for the spatially explicit modelling of ecosystem service production that is used in this thesis is the concept of hydrological response units. Hydrological response units (HRU's) are distributed landscape entities (not necessarily contiguous) having a common climate, land-use and underlying pedo-topo-geological association controlling their hydrological dynamics (Kronert et al. 2001). With this concept the heterogeneity of the three dimensional properties of the drainage basin can be preserved and it is therefore suited for spatial scale transfer of processes coupled to ecosystem structures. Generally GIS analysis of available data on topography, pedo-geological association and land cover is used to generate hydrological response units.

The second central modelling concept that is of importance to this thesis is that of hydrological and hydraulic node – link network models that are an abstract representation of the locational relationship between the physical entities in the river basin (Loucks and van Beek 2005). Nodes represent water users and wetland sites and links represent the linkage between these entities. Flows (water, flood waves, nutrients, etc.) are balanced for each node in each time step and the flow transport in the basin is calculated based on the spatial linkages and transformation of water availability, nutrients or flood waves along the trajectory. Wetlands, represented by one or many hydrological response units, are implemented as a node. The sum of hydrological responses from all the HRU's at gives the hydrological, eco-hydrological or hydraulic reaction of the wetland being investigated. At the same time it is possible to couple ecosystem processes, such as biomass production or greenhouse gas emissions, to the hydrological processes on the basis of the HRU concept.

Risk: environmental variability

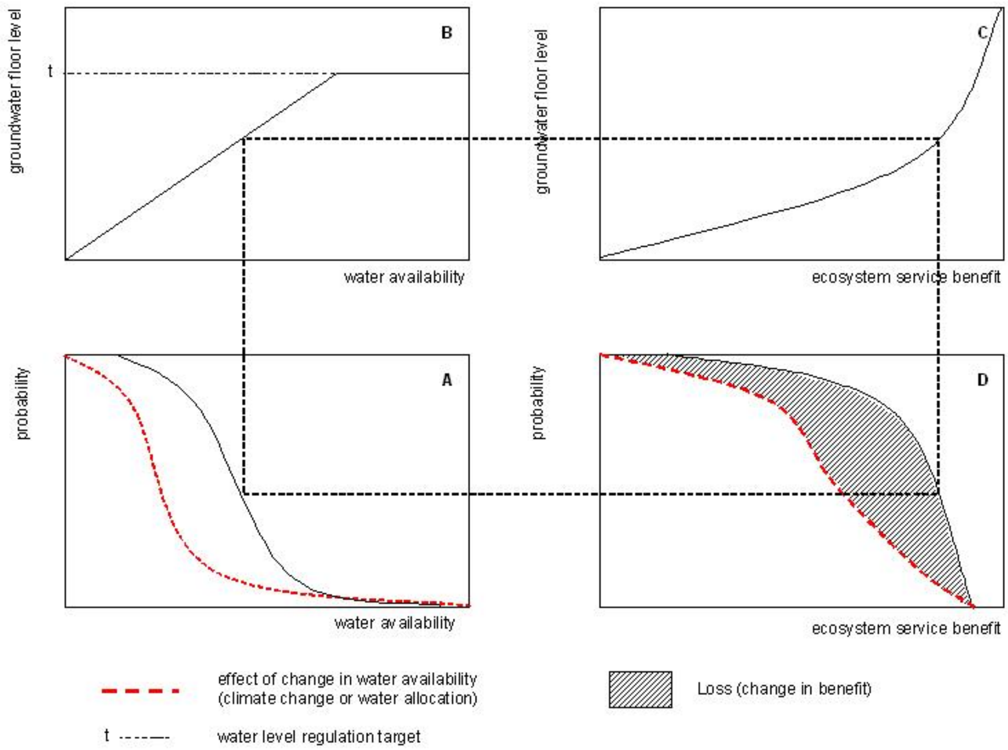
Ecosystems and the ecosystem services they provide are not only inhomogeneous across landscapes, they also have a large inter- and intra annual variability. One of the key drivers of this variability that is addressed in this thesis is the variability of climate and water flows. Variability that can be assigned meaningful probabilities such as return periods can be described in terms of risk (NRC 2000). Risk can be incorporated into an economic appraisal framework, by attributing probabilities to possible outcomes and estimating the expectation value of ecosystem service provision or benefits.



Source: based on Young 2005

Figure 6: Conceptual approach to calculating reductions in flood risk with an example of an upstream flood retention measure

Such a risk based approach is the standard procedure for the economic appraisal of flood risks (cf. Young 2005, NRC 2000, Penning-Rowsell et al. 1986). The basic framework for risk based evaluation is presented in Figure 6. Three basic relationships are the foundation of the appraisal process and these three are used to estimate a fourth. The first, in the lower left (quadrant A) is the flood discharge frequency (or discharge - probability function), which describes the probabilities of occurrences of various discharges. Generally a lower discharge can be expected to recur with a higher frequency then a higher discharge. The second relationship relates discharge to water level in the floodplain: the stage-discharge curve. This curve is shown in the upper left (quadrant B). The third relationship, the stage damage function (quadrant C), describes the potential damage at any flood stage. Generally, as flood stages rise, damages increase. Finally, the cumulative damage frequency or probability function can be derived by mapping the flood frequencies onto damage frequencies (quadrant D). The area under this last curve (the integral of the probability function) is the expectation value or average annual damage. The effect of different measures can be assessed by modelling the effects on the specified relationships. For example an increase in the upstream retention capacity of wetland sites changes the discharge - probability function for a downstream location (cf. the example in Figure 6). The benefit of such a measure then is the difference in the expectation value of damage or the avoided average annual damage.



Source: own illustration

Figure 7: Conceptual approach to calculating the expectation value of wetland ecosystem service benefits for variable conditions of water availability.

In this thesis, the risk based approach is not only used to asses flooding frequencies of river floodplains but is also applied to determine the benefits from fen wetland ecosystem services as a function of water availability. This is conceptually illustrated in Figure 7. The lower left quadrant (A) describes the cumulative distribution of the wetland water balance that is a function of climate and basin water availability. This information is generated from climate and hydrological models for the river basin. The water balance is translated to ground water floor levels (quadrant B) by the wetlands hydrology sub-model integrated into the basin model. The water levels in turn are the key determinant for the ecosystem service production functions (quadrant C) that are integrated into the wetlands model. Following this procedure, the cumulative frequency distribution of water availability can be transformed to a cumulative probability function for the provision of ecosystem service benefits (quadrant D). Again, the change in benefit can be determined as the difference between the baseline and the resultant probability distribution curves for a change in water availability, for example induced by climate change or by change in water management.

More formally, the resultant benefit function can be considered to be a function of a random distributed variable describing the wetland water balance. As a result a cumulative distribution function and an expectation value can be established. The expected annual benefit is then defined as:

$$E(B) = \int_{-\infty}^{\infty} \lambda(x) f(x) \delta x \tag{1}$$

where $\lambda(x)$ is the benefit function, x is a continuous random variable describing the water balance or water levels, f(x) is the probability density function.

From a discrete number of randomly distributed realisations r generated with a simulation model, the expectation (or average annual) level of wetland ecosystem service provision E(ES) in a year t under varying conditions of water availability can then be calculated as follows:

$$E(ES_t) = \sum_r \left[P_r \cdot \left(\sum_{wl} ES(WL)_{wl} \cdot A_{wl,r,t} \right) \right] \tag{2}$$

where P is the occurrence probability of a realisation r and where $\sum_{r=1}^{n} P_r = 1$ to ensure normalisation, wl is the water level and Awl is the area of the spatial aggregation unit with an average annual water level of wl in realisation r, and ES(WL) is the ecosystem service provision level as a function of water level wl.

Temporal concepts: environmental change

Finally, ecosystems change across time. Two key drivers for wetland ecosystems are changes in land management and water availability. The most important climate stimuli that influence the hydrological cycle of a river basin and hence the variability of water resources are temperature and precipitation (IPCC 2007, Aerts and Droogers 2004). Amongst the various relevant drivers, this thesis focuses on the effects of global climatic change as a driver of future water availability. The studies in this thesis use the above concept of risk to explore the gradual shift in the expectation value of benefits with changes in climatic conditions. Such an analysis requires quantitative projections of expected future climatic conditions (and their variation or probability distribution) from which impacts can be determined. This procedure is also illustrated in Figure 7. In analogy to the above example from flood risk, a shift in climatic variability induces a shift of the cumulative probability distribution of water availability (Figure 7 A). The resultant effect of such a shift in climatic variability is reflected in the cumulative probability distribution of ecosystem service benefits that can be described by the change in the expectation value of benefits. The change of benefit (in this example a loss) corresponds to the shaded area in Figure 7 D.

However, what is defined as risk (or vulnerability to climatic variability) is not only reflective of the exposure and sensitivity of the wetland ecosystems to climatic variability but also the capacity to cope, adapt or recover from the effects of these conditions (Smit and Wandel 2006). Adaptations are manifestations of adaptive capacity and they represent ways of reducing exposure or susceptibility to climatic variability. Adaptive capacity can be analysed using the concept of coping ranges (Smit and Pilifosova 2003). The term coping range is understood to denote the shorter term capacity to deal with climatic variability and the term adaptive capacity to denote longer term, autonomous or managed adjustments. Coping with climate variability has always been part of water management (Veraart and Bakker 2009). One of the implications of climate change for water resources management is that long-term planning can no longer be based on static assumptions regarding climatic conditions and water availability. In this thesis, the vulnerability of ecosystem service provision of wetlands to climate change is investigated. Various water management options designed to mitigate existing water deficits are therefore investigated and assessed with regard to their performance under projections of future climatic conditions.

3.3 Wetlands and people: valuation of wetland ecosystem services

Next to describing the production of ecosystem services, the second major challenge in any appraisal is the valuation of the ecosystem service benefits. The following section addresses methods suited for valuing different goods and highlights some of the relevant key concepts that are important for the studies presented in this thesis.

Valuation methods

There are numerous difficulties in estimating the values of wetland ecosystem services. Young (2005) differentiates four basic scenarios for the valuation of impacts of measures, based on the availability of market prices:

1. Impacts for which markets exist and market prices reflect scarcity values. In the analysis of wetland policies this is more often the case for the cost side than the benefit side.

2. Impacts for which market prices may be observed, but that fail to reflect social values although they can be adjusted. This is especially relevant in the valuation of benefits from agricultural land use, that in many cases need to be adjusted for government transfer payments to the agricultural sector.

3. Impacts for which market prices do not exist although it is possible to identify surrogate market prices. This is relevant for most ecosystem service benefits that are not traded in markets

4. Impacts for which market or surrogate prices are not meaningful.

For the valuation of wetland ecosystem services, categories 2 and 3 are most typical. In these instances, the value of ecosystem service benefits needs to be determined on the basis of so called accounting or shadow prices that are determined using non market based valuation techniques. A distinction between market based and non market based methods is therefore a useful way to classify the available valuation methods (Young 2005, Hanley and Barbier 2009, Turner et al. 2008, Haab and McConell 2002). Market based valuation means that existing market behaviour and market transactions are used as the basis for the evaluation. Economic values are derived from existing market prices for production inputs or consumer goods. Many wetland ecosystem services are not traded in markets and therefore remain unpriced. Both direct and indirect methods can be used for valuation. Direct methods use direct elicitation of willingness to pay in hypothetical market situations. These are also called stated preference methods and comprise contingent valuation and choice based valuation methods. Sometimes the prices public institutions are willing to pay for enhanced provision of ecosystem services, for example under agri-environmental schemes, are used as a surrogate for aggregated willingness to pay based on collective choice decisions. This method is termed public pricing. Indirect methods extract value estimates from market based prices for complementary or alternative goods and services. These methods can be based on revealed preferences, as in the travel cost method and hedonic price method or on the indirect value of wetland resources in production processes. Other indirect methods use various costs as a proxy for benefits – such as the avoided damage costs, alternative or replacement costs, defensive and restoration costs. The underlying assumption is that benefits are at least as high as the costs involved in repairing, avoiding or compensating for damage. However, while widely used for ease of application, these methods only produce valid estimates if it can be shown that the repair or alternatives will provide a perfect substitute for the ecosystem service and that there is actually a demand for the service at the assumed prices. Although there is some overlap, the valuation methods appropriate for the valuation of private goods differ from those for public goods. Table 3 provides an overview of the available methods for the valuation of ecosystem service benefits.

Table 3: Valuation methods to value ecosystem service benefits

Method	Description	D*	I*	N*
A. Revealed preference approaches				
Market prices				
Observation of market transactions / prices	Observed prices from transactions of rights for ecosystem services	X	X	
Production function approaches				
Econometric estimation of production and cost function	Econometric analysis used to relate output or costs to of production of a marketed good by treating ecosystem service as one input	X	X	
Change in net rents	Constructed residual models for deriving estimate of net producers income or rents attributable to an increment in ecosystem service provision	X	X	
Mathematical programming	Constructed residual model for deriving estimate of marginal net producers rents or marginal costs attributable to change in ecosystem service provision	X	X	
Surrogate market approaches				
Travel cost method	Econometric analysis to infer the value of recreational site attributes from the variation of expenditures incurred by consumers to travel to the site	X	X	
Hedonic Price Method	Econometric analysis of data on real property transactions for different sites with varying availability / proximity of ecosystem services	X	X	
B. Cost based approaches				
Replacement or alternative cost method	Value attributable to cost savings / additional costs from implementing the next best alternative source (shadow projects) of ecosystem service	X	X	
Damage cost methods	Maximum willingness to pay given as the monetary value of avoided damages from a change in ecosystem service provision		X	
Mitigative or avertive behaviour method	Change in costs of actions undertaken to mitigate or avoid incurring an external cost as a partial measure of the benefits		X	

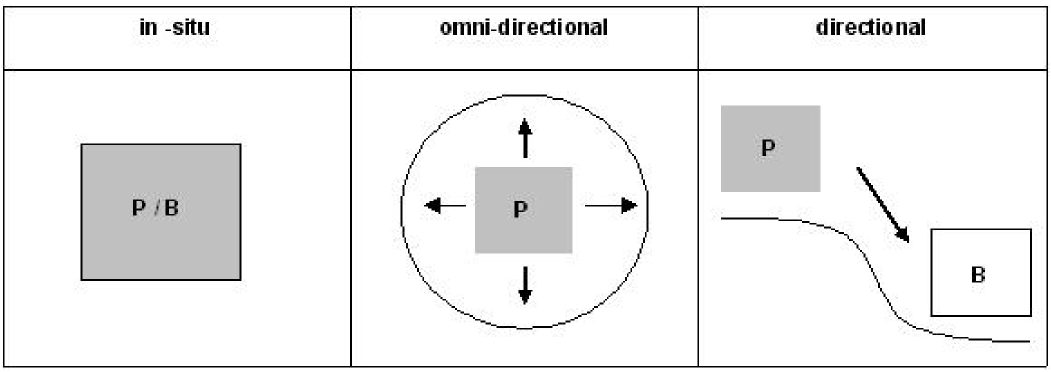
	of a change in ecosystem service provision			
C. Stated preference approaches				
Contingent valuation	Construction of a hypothetical market by direct surveying of a sample of individuals to state willingness to pay for a change in ecosystem service provision	X	X	X
Choice modelling	Construction of a hypothetical market by direct surveying of a sample of individuals to make trade offs between different goods, varying levels of provision of ecosystem service provision and willingness to pay	X	X	X
D. Public expenditure approaches				
Public Pricing	Public investment for instance for land purchase or monetary incentives as a surrogate for market transactions, with public expenditure assumed to be a proxy for aggregated individual demand	X	X	X

* D = direct use values, I = indirect use values, N = non use values

Source: based on Young 2005 and Turner et al. 2008

Spatial dimensions: scale of service benefit areas and scope of wetland ecosystem services.

Besides the spatial heterogeneity of ecosystem service production that was already introduced above, there are two further important spatial dimensions that need to be taken into consideration in an analysis of wetland ecosystem service benefits: the scale of the service benefit area and the scope of the wetland service. The spatial distribution of the demand for and the relative availability of ecosystem services are key contextual factor that determine the value of ecosystem services (cf. Hein et al. 2006).



Source: Fischer et al. 2009

Figure 8: Ecosystem service benefit areas: possible spatial relationships between service production (P) and benefit (B) areas.

First, there are differences in the geographical scale of the service benefit area. This concerns the spatial scale or extension of the benefits beyond the point where they are generated and where they are perceived and valued by society. There are two important properties related to dimensions of scale: direction and spatial extent. Figure 8 provides a classifications scheme to describe the relationships between the locality of service production and the area where the benefits are realized (cf. Fischer et al. 2009). This is based on the differentiation between in-situ, omni-directional and directional benefit areas. For example, the value of directional services such as flood protection is dependent on the demand for flood risk reduction below the wetland. In contrast, the benefits from greenhouse gas emission reductions to the atmosphere are omni-directional. The second attribute of scale describes the spatial extent over which individuals hold a value for the ecosystem service benefit. This may range from a global service benefit area as can be assumed for emission reductions of greenhouse gases to local or on-site benefits from biomass production to agricultural enterprises. However, for many ecosystem services it is likely that the mean value placed on a change in service provision falls, the further an individual lives from a site because an individual's preferences are related to the intensity of use made of the valued resource. This phenomenon is referred to as distance decay (cf. Bateman et al. 2006). Individuals that are active recreational users of wetland sites can be assumed to have stronger preferences for improving the wetland habitat quality than non users. While non use values may in principle be held by anyone irrespective of the distance from an individual's home to the relevant site, it seems reasonable to propose that the share of users declines with distance to a site. Distance decay effects need to be considered for the correct aggregation of individual demand over the population of the service benefit area.

The second spatial dimension is related to the scope or the relative extent of a change in the availability of wetland sites and wetland ecosystem service benefits in relation to the availability of substitute sites or substitute services in the service benefit area. Scope is most readily measured in terms of wetland area. Provided that wetland conservation is a normal good, economic theory would suggest that the marginal benefits would be decreasing with an increasing availability of the different ecosystem services maintained by wetlands. This implies that the marginal benefit of wetland restoration programmes would be decreasing with increasing scope. Likewise, an increased availability of substitutes would lower the marginal benefit. For example, it could be expected that the recreational value of additional wetlands would be lower in areas with high availability of substitute wetland sites that can be accessed at the same costs by a recreation seeking population. However, the substitute does not necessarily have to be a wetland ecosystem service. Taking another example, the value of the nutrient retention function of additional restored wetlands is dependent on the general availability of (substitute) nutrient reduction options and their costs within a river basin.

Taken together scope and scale effects imply that as ecosystem services become scarcer, the marginal values of incremental changes in wetland service provision increase. However, whether an incremental change is meaningful in terms of marginal analysis is conditioned by the scale of the policy decision. For example, the loss of a part of a wetland may already be a non-marginal change from a local perspective, while the loss of the greenhouse gas sequestration capacity of a whole wetland landscape is likely to constitute a marginal change from the perspective of a national greenhouse gas budget. Within cost benefit analysis, the issue of marginality is related to the definition of standing, or whose benefits and costs are to be counted. One basic principle for deciding who has standing is to base the decision on the widest definition of service benefit area. For practical considerations, the most frequently used standing is the population of a country, because impacts of local or regional wetland management projects on costs and benefits generally extend beyond the boundaries of the project area. One of the specific contributions of this thesis is to investigate the relevance of scale and scope effects for the value of various ecosystem services and to develop approaches that facilitate the appropriate scaling of the value estimates for wetland restoration measures of varying scope.

3.4 Wetlands and policy: cost benefit appraisal of wetland management options

A cost benefit framework for appraisal of wetland land and water management

This thesis uses a cost benefit analytical framework to evaluate wetland land and water management options. People and property assets are not part of the trade-off in the set of case studies considered here. These basically only involve the conversion of agricultural and forestry land to a more natural wetland status. The case studies build on the well developed basic framework for evaluating the benefits from land drainage, water and flood regulation for the enhancement of agricultural production (Penning-Rowsell 1986). According to this appraisal approach, protecting agricultural land from flooding or improving its drainage and water level regulation requires investment in water regulation infrastructure such as embankments, underdrainage, drainage and irrigation ditches, weirs and pumping stations. To offset these costs, two types of benefit are expected. First, a greater return from agricultural production activities is expected from the agricultural use of the drained wetland area. Its value is generally calculated from the difference between the total gross margin (returns less variable costs) before and after the scheme implementation, less any change in fixed costs. Secondly, the crop damage or loss from regular flooding or water level variations is expected to be reduced by embankment or water level regulation measures. The value of this improvement can be calculated from the expectation value of annual losses based on the loss of gross margin. This view of the benefits of drainage, water regulation and flood protection can be summarised as follows.

$$B = \sum_{t=0}^{n} (1+r)^{-t+1} \cdot (gm_t^m - gm_t^b) - \Delta f_t - \Delta E(L_t) \tag{3}$$

with B total benefit, t is a year during the schemes life, n expected life of the scheme, r is the discount rate, gm is the gross margin from the agricultural production activity for the water regulation targets in year t in the baseline b and the management option m, Δf is the net change in fixed costs associated with the change in the production system and ΔE(L) is the change in the expected average annual loss from crop damages by flooding or drought compared. This formula yields the capital sum to be weighed against the investment costs. Subsidies should not be included in neither costs nor benefits. Subsides or transfer payments are redistribution of welfare and therefore do not constitute changes in welfare.

A key argument of this thesis is that external environmental effects need to be taken into account. In most standard applications of cost benefit analysis in project appraisal the environmental effects of diking and drainage are considered to be intangible effects (cf. Meyer and Messner 2005, Holm-Müller and Muthke 2001). By definition, intangible effects cannot be given monetary values that allow an inclusion in cost benefit analysis. However these intangible environmental benefits are particularly relevant for the appraisal of wetland conservation and restoration policies and projects. In these types of projects, the improvements in the provision of public goods constitute the main benefits whereas the loss in agricultural productivity constitutes an opportunity cost. The use of cost benefit analysis in a decision making context where these non- market impacts are expected to be significant has stimulated an extensive debate and literature (cf. Hanley and Barbier 2009, Brouwer and Pearce 2005, NRC 2005). When including also external effects or public goods that do not have a market price in monetary units, this is often referred to as an extended cost benefit analysis.

The standard appraisal framework therefore has to be extended to include the public ecosystem service benefits. These are essentially the non-agricultural benefits. In addition it has to be able to accommodate for the long-run and short-run effects of management options. For this thesis, management options are considered that reduce the short term variability of water availability (for example by increasing the water supply by inter-basin water transfer or changes in the water allocation) and that require longer term changes in the water level regulation targets and land use of wetlands. Following the standard with and without procedure which sets the net discounted costs and benefits of each management option against the baseline management option, this expanded view of the change of benefits from changes to wetland land use and water level regulation in the long run and short run changes of inter annual availability of water can be summarised as follows:

$$\Delta B_t = \Delta tB_t - \Delta E(L_t) \tag{4}$$

with

$$\Delta tB_t = (tB_t^m - tB_t^b) \tag{5}$$

$$\Delta E(L_t) = E(L_t^m) - E(L_t^b) = (tB_t^m - (E(aB_t^m)) - (tB_t^b - E(aB_t^b)) \tag{6}$$

where B are the agricultural and other, non agricultural ecosystem services benefits, tB is the target benefit at water level regulation target, E(aB) is the expectation value of the actual benefit under actual conditions of water availability for the measure m and baseline b. E(L) is then the expectation value of the average annual loss compared to the target water level.

This general formulation of the benefits of changes in wetland management has further useful properties for the assessment methodology developed in this thesis, because it facilitates the combination of a comparative static approach to the assessment of long-term restoration or adaptation measures (for example by dike relocation or wetland rewetting) with the risk based approach to analyse short term variability of climatic conditions and water availability (for example risk of flood events or drought) within a single framework. While integrated economic-hydrological modelling approaches are used to directly estimate ΔE(L) in a dynamic modelling framework, ΔtB is estimated on the basis of static comparisons of land use change.

The economic feasibility criterion for evaluating changes in wetland and basin water management then can be written as:

$$PVNB = \sum_{t=0}^{n}\left[\frac{\Delta B_t^{AGR}}{(1+r)^t}\right] + \sum_{t=0}^{n}\left[\frac{\Delta B_t^{oESB}}{(1+r)^t}\right] - \sum_{t=0}^{n}\left[\frac{\Delta C_t}{(1+r)^t}\right] - \sum_{t=0}^{n}\left[\frac{\Delta D_t}{(1+r)^t}\right] \tag{7}$$

Where PVNB is the present value of net benefits, t is a year during the schemes life, n is the expected life of the scheme, r is the discount rate, B^{agr} and B^{oESB} are the incremental benefit from agricultural and other ecosystem service benefits induced by changes in wetland or basin water and land management, C is the change in capital and operating costs for wetland and basin water management and D is the incremental disbenefit (forgone benefits or external costs) to other water using sectors in the basin. According to the general approach to cost benefit appraisal outlined in previous sections, the economic feasibility hypothesis to be tested then is:

$$\text{PVNB} > 0? \tag{8}$$

Of course the test can also be expressed in the alternative but largely equivalent form of the benefit cost ratio or internal rate of return. Non market economic valuation (shadow pricing) will generally be required to estimate the terms B and D and possibly for elements of C. Implementing this test therefore requires the application of appropriate methods to estimate the marginal or incremental benefits and benefits forgone from changes in wetland land and water management.

The most commonly applied economic appraisal method to assess the potential of various measures to contribute to an efficient realisation of sectoral targets for river basin management (e.g. flood risk reduction, nutrient load reduction) is cost effectiveness analysis (cf. Engelen et al. 2008). The cost effectiveness (CE) of a wetland restoration measure is defined as:

$$CE = \sum_{t=0}^{T} \frac{1}{(1+r)^t} (PC_t + OC_t - oESB_t) \Big/ \sum_{t=0}^{T} ESB_t \quad (9)$$

where PC and OC are the project and opportunity costs of the measure, ESB is the primarily targeted ecosystem services measured in physical units and oESB are the economic values of secondary environmental benefits from other ecosystem services that are jointly produced by the measure. Both full cost benefit analysis and cost-effectiveness analysis of wetland restoration measures therefore require information on the value of ecosystem service benefits.

Discounting and treatment of time

Deriving an aggregate measure of costs and benefits over time requires an adequate inter-temporal aggregation method, such as discounting. The use of discounting is integral to cost benefit analysis (Hanley and Barbier 2009). The rationale for discounting is that costs and benefits that occur in the future are not valued as highly as those that occur in the present. High discount rates are often justified based on the opportunity cost of capital, though to be correct this is relevant only for financial analysis. It is important to realize that two different types of discounting may be practiced in economic analysis: utility and consumption discounting (Turner et al. 2008).

What is normally referred to as the discount rate is in fact the utility discount rate, also known as the pure rate of time preference or the social discount rate (Young 2005). There is no reason for this discount rate to be positive, the value of the utility discount rate reflects the relative valuations that are placed on the utility in present and future time periods. The consumption discount rate is conceptually different. It represents the weight placed on increments of consumption at different dates. Even if future utilities are valued the same as present utilities, a future increment of consumption may still be valued different from the same increment today. One reason for this is the expected change in standard of living in the future that taken together with a diminishing utility of increasing consumption leads to a lower increment of utility for a unit of consumption in future. If this approach is accepted, this implies a positive discount rate if living standards are expected to rise over time.

In more practical terms, the utility discount rate is applicable in general equilibrium analysis and the consumption discount rate in partial equilibrium analysis (NRC 2005). Most of the environmental valuation problems presented in this thesis are of a partial equilibrium nature, so the consumption discount rate applies. Discounting consumption is unavoidable in the utilitarian value framework; however determining the appropriate rate to be applied in practice is difficult. In order to maintain coherence over policy appraisals in different sectors it is common practice to resort to government guidelines on the appropriate range of values to use and to test for sensitivity across this range.

Treatment of risk and uncertainty

In an economic appraisal, uncertainty is associated both with physical outcomes and their economic consequences. If there are reliable probabilities available, describing the magnitude of variation of possible outcomes, these can meaningfully described by risk (NRC 2000, 2005). An approach to quantify the risk from variable climatic conditions and water availability has been outlined above. In contrast, one speaks of uncertainty when data based probabilities are entirely unknown. While some aspects may be amenable to a risk based approach, there are other sources of uncertainty, for example the model uncertainty that arise from uncertainty about the relationships between key variables or parameter uncertainty, that arises from uncertainty about the correct specification of parameters in the model.

Sensitivity analysis, Monte - Carlo analysis and scenario analysis are possible responses to model and parameter uncertainties. In sensitivity analysis, various plausible values are used for key variable in the evaluation. This provides a range of estimates within which the true value can be expected to fall. Sensitivity analysis is best based on statistical distributions of possible magnitudes. However more frequently they are based on expert judgment regarding plausible ranges of parameter value or subjective probabilities, which are based on the strength of the belief in the likelihood of an outcome. A more sophisticated way to incorporate uncertainty in a valuation study is to use Monte-Carlo analysis. This method can provide an estimate of the probability distribution of possible values that is derived from the uncertainty about the underlying parameters and relationships. A prerequisite for such an analysis is however some probabilistic information about the elements of a valuation. Scenario analysis can also be used to incorporate uncertainty through the comparison of results using parameter values that represent different possible futures. All of the outline approaches are applied in this thesis.

Analysis of uncertainty can create ambiguity regarding the decision criteria, but it is a necessary component of economic valuation. Making good decisions is more difficult under conditions of uncertainty than risk. Adaptive management, risk averse, precautionary and safe minimum standard approaches are possible reactions to deal with the ambiguity and uncertainty of appraisal results in translation to practical management decisions (NRC 2000, 2005).

4 Overarching research objectives of the thesis

From an economic perspective, the key challenge in determining whether wetland restoration or reallocation of water to wetlands are appropriate policy or management goals lies in the difficulty of determining the value of the public benefits provided by wetlands. The dual focus of this thesis is first the valuation of ecosystem services that have a public goods character and second the integration of wetland ecosystem service benefits into the economic appraisal of river basin management options.

Building on the outlined framework for the valuation of ecosystem services from wetlands, the three overarching objectives of this thesis can be defined as:

1. To develop approaches to model the production of wetland ecosystem service provision and ecosystem service benefits as a function of water, flood wave and nutrient flows in river basin models that are compatible with approaches to model and appraise water resources management options. The particular focus is to develop methods that are (a) suited for large scale assessment models covering whole river basins or river trajectories using hydrologic, hydraulic and eco-hydrologic modelling approaches and (b) take a risk based approach to the evaluation of benefits from wetlands.

2. To generate new empirical evidence on the economic value of a large range of major ecosystem services provided by wetland ecosystems in the context of a major German river basin, by (a) applying a range of suitable valuation methods of different complexities, (b) contributing incremental methodological innovations in the application of the valuation methods and (c) with a particular focus on the effects of scale and scope on value estimates.

3. To provide exemplary case studies to demonstrate the applicability of the ecosystem services approach to improve the information for decision making in integrated water resources management, in particular regarding (a) the inclusion of wetlands in strategic approaches to economic appraisal of integrated water resources management on the river basin scale and (b) the possible impacts of climatic change on ecosystem service provision and benefits.

5 Structure of thesis

The thesis is based on eight papers written as stand-alone manuscripts that are published or submitted to peer-reviewed journals. The first five papers have a methodological focus and each applies a different valuation method to the valuation of specific wetland ecosystem services. These are flood risk regulation using an avoided damage approach (Paper 1), recreation using the travel cost method (Paper 2), the provision of habitats and biodiversity using a meta analytical function approach to the benefit transfer of stated preferences (Paper 3), greenhouse gas regulation (Paper 4) and nutrient regulation (Paper 5) based on the marginal abatement cost method. The final three papers present case studies of an integrated economic assessment using the framework of an extended cost benefit analysis. These analyses build on the valuation approaches developed in the previous papers. The first of these papers presents an integrated assessment of riverine floodplain management options (Paper 6). The second paper presents an assessment of water management options for regulated lowland peat wetlands (Paper 7). The final paper presents wetlands as a water user amongst other water uses within the framework of an economic approach to evaluate the changes in water availability in large river basins (Paper 8). All papers refer to case studies from the Elbe River Basin. The structure of the thesis is summarised in Table 5.

Table 5: Overview of the structure of the thesis and the included manuscripts

		Flood risk reduction	Recreation	Habitat and biodiversity	Greenhouse gas regulation	Nutrient retention	Agricultural biomass
	Valuation methods						
1	Large scale assessment of the flood risk and the effects of mitigation measures along the Elbe River	X					
2	Impacts of boating trip limitations on the recreational value of the Spreewald wetland: a pooled revealed / contingent behaviour application of the travel cost method		X				
3	Accounting for scope and distance decay in meta-functional benefit transfer: an application to the willingness to pay for wetland conservation programmes in Europe			X			
4	Social benefits and abatement costs of greenhouse gas emission reductions from restoring drained fen wetlands. A case study from the Elbe River Basin.				X		
5	Economic value of the nutrient retention function of restored floodplain wetlands in the Elbe River Basin.					X	
	Integrated assessment						
6	Strategic cost benefit analysis of an integrated floodplain management policy for the Elbe River	X		X		X	X
7	Integrated economic-hydrologic assessment of water management options for regulated wetlands under conditions of climate change: a case study from the Spreewald (Germany).		X	X	X		X
8	Economic risks associated with low flows in the Elbe River Basin (Germany): an integrated economic-hydrologic approach to assess vulnerability to climate change			X	X		X

6 Summary of the manuscripts: specific issues and key results.

This section provides a summary of the specific methodological issues addressed and key findings presented in each of the methodological and integrated assessment case study papers.

The first paper ("Large scale assessment of the flood risk") presents an application of the avoided damage cost method to value reductions in flood risk by various floodplain wetland restoration options. To this end, the first large scale flood risk model that encompasses the complete trajectory of the Elbe River was set up. The River Elbe served as an example to demonstrate the usefulness of a rapid, GIS-based flood risk assessment methodology. A hydraulic routing model was extended to include the effect of planned (regulated and unregulated) and unintended retention (dike breaches) on the peak water levels. Further an inundation model for dike breaches due to dike overtopping and a macro-scale economic approach to assess the flood damage where added. The flexible approach to model the effects of measures by means of volume storage functions allows for rapid assessment of combinations of retention measures of various proposed dimensions and at multiple locations. The model is applied to a series of exemplary flood risk mitigation measures to show the downstream effects and the additive effects of combinations of measures on the flood risk along the river.

The second paper ("Impacts of boating trip limitations") addresses the valuation of recreational use of wetlands using a revealed preference approach. This paper presents an innovative application of the zonal travel cost method to the valuation of variable water flows in a wetland setting. It is an innovative contribution, in that it combines data on actual trips taken to a site (revealed behaviour) with data on anticipated trips that are stated as a response to hypothetical scenarios constructed for survey respondents (contingent behaviour). These two sources of data are combined in order to assess whether and to what extent the maintenance of minimum in-stream flows for boating matter in demand for trips to a recreation site. The data from the on-site survey is used to estimate an aggregate count data travel cost model. The findings indicate that variations in navigability significantly affect demand and associated welfare measures.

The third paper ("Accounting for scope and distance decay in meta-analysis") addresses the valuation of non-use and non consumptive use values associated with the conservation of wetland habitat based on stated preference methods. The paper presents a meta-analytical function approach to benefit transfer. The paper argues that key factors that need to be considered for benefit transfer of stated preferences estimates are the size of the wetland for which changes to habitat quality are proposed (scope effects) and the market size or spatial extent of the sample population (distance decay effects). These effects have not been demonstrated in previous meta-analysis of wetland studies.

Because the number of empirical studies on wetland valuation has risen continuously, a smaller, but more homogeneous dataset was extracted from the literature compared to previous studies. In this way it was possible to single out the effects of scope of measures, market size and income on the value estimate. The paper is able to demonstrate a theoretically consistent meta-transfer function, that shows willingness-to-pay to increase with program size but at a decreasing rate (scope effects) and to decrease with increasing size of the sample area (distance decay). This enhances the potential to use the results of the meta-regression for benefit transfer. The results further indicate that choice of method have significant influence on the mean value estimate.

The fourth paper ("Social benefits and abatement costs of greenhouse gas emission reductions") addresses the valuation of greenhouse gas emission from peat wetlands. The paper presents estimates of the costs and benefits of reducing greenhouse gas (GHG) emissions through fen peat wetland restoration. This study takes previous research on GHG emissions from peat wetlands further by coupling water level dependent emission functions with a large scale assessment of the hydrology and water management of wetlands. For this purpose a water management model for the Elbe River Basin is used, that includes the major lowland fen wetland sites as water users. Based on the resultant estimates of the GHG emissions and reduction potential of wetlands under more realistic description of water availability, the paper provides improved estimates of the benefits of restoration in terms of the shadow price of carbon and the abatement costs of wetland restoration measures. An econometric approach is used to develop abatement cost estimates. The paper shows that wetland restoration can be a low cost option for greenhouse gas mitigation. An approach focused on restoration is a more efficient strategy compared to an approach centred on agri-environmental schemes, even though both components are required in a zoning approach. However, it is also shown that the initial reduction of greenhouse gas emissions that can be realised by restoration is to a large extent compensated by increases in emissions due to reduced water availability over the next twenty five to fifty years. The effects of anticipated climatic change may reduce the effectiveness of wetland restoration measures by roughly 50 %.

The fifth paper ("Economic value of the nutrient retention function") presents an application of an indirect method, the alternative or replacement cost method, to value the nutrient regulation function of floodplain wetlands. The paper presents a novel cost minimisation model for nutrient abatement measures for the River Elbe that is based on an existing simulation model for nutrient emissions and flows in large river basins. The model is applied to estimate the marginal shadow price of phosphate and nitrogen nutrient retention by restored floodplains for a range of basin wide abatement requirements. The marginal benefit of restored floodplain area in first line varies with the nutrient load reduction target and to a lesser degree with the scope of the floodplain restoration project. In addition, this paper presents an empirical cost function for the costs of floodplain restoration measures in the Elbe Basin. In conjunction with the shadow prices, this allows for a rapid strategic assessment of the costs and benefits of 45 potential restoration sites along the Elbe trajectory. In spite of the large investment costs for dike realignments, a result of this study is that the nutrient retention effects alone may in many cases generate sufficient benefits to generate an economic efficiency gain. Floodplain restoration may therefore, under advantageous circumstances, constitute a cost effective nutrient abatement measure.

The final three papers address integrated assessment of wetland and river basin management option in an extended cost benefit analytical framework. The papers draw on the results of the valuation methods presented above.

Paper number six (“Strategic cost benefit analysis of an integrated floodplain management policy”) then addresses strategic land use choices for floodplains from an integrated floodplain management perspective. It applies the ecosystem services approach to explore the economic effects of floodplain management programs of various dimension and composition in an extended cost-benefit analytical framework. The paper builds on the results of the papers that address the valuation of flood risk, nutrient retention and habitat conservation. The analysis presented in this paper is novel, in that it explicitly accounts for issues of scale and upstream-downstream interdependencies in the valuation approach. Particular attention is given to scope effects in the assessment of benefits from flood risk reduction, nutrient retention and wetland habitat conservation for programs of increasing scale. The choice of the appropriate strategy for floodplain management is contested between stakeholders of nature conservation and flood risk management. Whereas flood risk management interventions have focused on dike strengthening (“hold-the-line” strategy), nature conservationists are arguing for an integrated approach that includes large scale floodplain restoration and realignment of dike lines (“space for the river” strategy). The key empirical result is that large scale restoration of floodplains of the River Elbe provides an economic efficiency gain. The results therefore support the general policy shift in floodplain management from a “hold-the-line” to a “space for the river” strategy. It is argued that an extended cost-benefit analysis should be one component of a wider strategic policy appraisal process that integrates targets of river basin -, flood risk - and floodplain land use policies.

The seventh paper (“Water management options for regulated wetlands”) presents an economic assessment of wetland water management options for a major water level regulated lowland wetland in the Spree River Basin - the Spreewald. Wetlands are the major environmental water users in the Spree River Basin that may withdraw more than half of river flows in summer month. These wetlands provide many ecosystem services that are directly regulated by basin water availability. From both a hydrological and an economic point of view, wetlands such as the Spreewald must be understood as multi-functional water users competing with other water users upstream and downstream for sufficient water supplies. This paper takes previous research further by providing a methodology for the systematic integration of multifunctional wetland water uses into a water resources modelling and assessment framework for large river basins. The paper uses a water resources modelling system that is also used by various state water management authorities in Germany for long term water resource planning. This paper presents the integrated economic assessment methodology. It combines different valuation approaches for different ecosystem services provided by wetlands in an integrated, cost-benefit analytical framework. The economic assessment is based on the valuation of following ecosystem services: grassland fodder production, recreational boating, habitat and biodiversity conservation and regulation of greenhouse gas regulation. It is found that under future climatic conditions wetlands such as the Spreewald will require an increasing amount of water to maintain the current levels of benefits derived from the wetlands ecosystem services. Additional inter basin water transfer could compensate some of the negative effects of increased water demand. However, the assessed transfer option is not economically efficient. However water management approaches that increase the inter-temporal water storage in the wetland soils by higher groundwater level regulation targets are found to generate net gains in benefits compared to the current management water management without requiring an increasing of the water supply.

The eighth and final paper ("Economic risks associated with low flows in the Elbe River Basin") presents the scaling up of the method developed for the Spreewald to the scale of a water resources management model for the complete Elbe River Basin. The paper presents the approach and methods used to incorporate economic valuation of changes in water availability for all important water uses – including the water use of all major lowland wetland sites - into the model. It is the first integrated large scale economic-hydrologic river basin model to be presented for Germany that addresses issues of water quantity. A key methodological advance is the development of economic valuation functions for a majority of important water uses in the Elbe Basin: regulated wetlands, irrigation, hydropower, thermal power plants, industry, municipal water supply, transport shipping, pond fisheries and selected recreational water uses. The inclusion of recreational and environmental water uses in the assessment is an innovative element. These uses typically have high economic value in developed economies but are often neglected due to methodological problems associated with their valuation. The paper presents the application of this model to assess in economic terms the potential effects of climate induced changes in water availability on the main water uses within the Elbe Basin. One of the key results is that wetlands are amongst the most vulnerable water uses both under current and future conditions in the German part of the basin. A further result is that the effects of reduced water availability will tend to exacerbate existing water shortages. This is interesting, as it implies that the adaptation efforts should indeed begin with mitigation of existing water management problems.

7 Synthesis and conclusions

7.1 Implications for environmental decision making

Multiple wetland benefits – evidence and policy implications

The key empirical result of this thesis is that the restoration of major fen and floodplain wetlands in the River Elbe generates an economic efficiency gain – largely independent of the specific type of measures and the scope of proposed projects that were considered. The results of this thesis therefore provide supporting evidence for wetland management policies that promote increased restoration of wetland sites and the stabilisation of the wetland water balance. All of the studies presented in this thesis have highlighted the substantial economic benefits associated with an increase of the provision of ecosystem services provided by wetland ecosystems. The case studies show, that in many instances the benefits from a single ecosystem services may be large enough to justify wetland restoration. For example, given the large investment costs for dike realignments, it is a more surprising result that the nutrient retention effects alone may in many cases generate sufficient benefits to generate an economic efficiency gain (Paper 5 "nutrient retention"). Floodplain restoration may therefore, under advantageous circumstances, constitute a cost effective nutrient abatement measure. Likewise it is found that wetland restoration is a relatively low cost greenhouse gas mitigation option, if compared to the estimated marginal abatement costs in other sectors (Paper 4 ”greenhouse gases").

However the key thrust of the argument is that wetland restoration and water management have to be assessed as multifunctional projects or water users that provide a multitude of benefits. This requires an integrated approach to developing, appraising and implementing water management or major public works in a coherent manner. An integrated approach to wetland management also makes multi-functional projects more advantageous, as has been shown in the case studies of fen and floodplain restoration. The aim of promoting multifunctional projects is to provide a range of ecosystem services (and address a range of policy targets) at a lower cost than if each where provided separately. For example, the EU Floods Directive, the Water Framework Directive and the Fauna-Flora-Habitat Directive are important pillars of European environmental policy whose policy fields overlap in water dependant habitats such as wetlands. An integrated, ecosystem services based approach can help to identify the potential synergies in realizing the benefits targeted by different policy goals. The ecosystem services approach is also compatible with the Convention on Biological Diversity (CBD) and its ecosystem approach, which has been adopted as a key delivery mechanism for conservation.

Extended cost benefit analysis can contribute to the development of such an integrated approach by providing an economic efficiency oriented perspective. In particular in cases where the opportunity cost of restoration mainly involves the loss of lower value agricultural land, as in the case studies presented in this thesis, an efficiency oriented analysis based on a cost benefit analysis can provide decisive information.

Changing environments: climatic risks and adaptation to climatic change

Several studies presented in this thesis specifically address the effect of climate change on wetland ecosystem service benefits and the long term effectiveness of conservation measures. It is shown that the effects of climate change reduce inflows of additional water to wetlands and increase the water required to offset their increased evapo-transpiration. Second to land use changes driven by agricultural development and policy, water availability is the key factor that has to be accounted for in developing options for wetland restoration.

One of the case studies (Paper 4 "greenhouse gases") demonstrates that without any further action, decreasing water availability as a result of climatic change will lead to a 2 - 5 % increase of greenhouse gas emissions from fen peat wetlands in the Elbe lowlands over the next 50 years. A comparative assessment of climate risks across all major water using sectors in the Elbe Basin (Paper 8 "CBA low flows") comes to the conclusion that wetlands and the ecosystem services they support are among the most vulnerable water users in the basin. All of the investigated major lowland wetlands of the Elbe River Basin are affected by reductions in water availability. The associated losses of ecosystem service benefits are substantial compared to losses of other sectors. Not only does this imply that additional efforts are required to maintain the current status of lowland wetlands, but that water availability also has to be considered as a major limiting factor determining the cost-efficiency of the restoration of wetlands in many parts of the Elbe Basin. For example, it is shown that an initial reduction of greenhouse gas emissions realised by restoration is to a large extent compensated by increases in emissions due to reduced water availability over the next twenty five to fifty years (Paper 4 "greenhouse gases").

These finding provide additional arguments to proceed with adaptations in wetland water management that are required to restore a more natural water regime. Adaptation options that increase the efficiency of water use within the wetland, for example by changes in land and water management and options that enhance or secure the water allocation within the basin management need to be considered. In the case study of water management options for the Spreewald (Paper 7 "CBA water level regulation") it is shown, that additional water transfers could compensate some of the negative effects of increased water demand. However, water management approaches that prioritise the restoration of wetlands and the reactivation of the water storage capacities of wetland soils are found to substantially improve the benefits from wetland ecosystem services compared to the current management regime, without requiring an increase of the water supply. In situations of increasing water scarcity, the question of a reallocation of water resources based on the benefits from water use becomes more relevant. This thesis has provided evidence on the values associated with wetland water use and has provided methods to estimate the effects of marginal changes in water allocation on ecosystem service benefits. These methods can be used in future assessments of basin water allocation to systematically identify allocations that generate the largest possible benefit from the use of the available water resources.

7.2 Methodological advances: moving from case studies to standard practice

This thesis has presented applications and case studies of approaches for valuing wetland ecosystem services in the context of major river basins in Germany. Although there is an increasing body of literature on wetland valuation, the application in decision making context, at least in Germany, is still rare. This thesis has demonstrated that the application and integration of such valuation approaches into economic assessment frameworks that actually play a certain, though limited role in the practice of generating management and policy relevant information in Germany, is feasible and generates decision relevant information. These applications are to be found in the realm of flood risk management (cost benefit analysis of flood risk mitigation plans), nutrient management (cost effectiveness analysis of programme of measures) or water resources management (cost benefit analysis of changes in water management and augmentation of water management infrastructure).

Despite the steps forward that have been made in the valuation of wetland ecosystem services in recent years, a major challenge is to ensure that the methods and results of these studies are actually fed into the decision making process and that these are more actively used by economist who conduct appraisals of water and wetland related policies and projects. Whilst there is a whole bundle of factors that determine the application in regular decision making processes, this thesis has addressed possible methodological barriers. These are discussed under the three headlines of (a) production functions (b) value transfer and (c) uncertainty.

Integration of ecosystem service production functions into river basin models.

Economic assessments are dependent on a reliable quantification of the ecosystem services. Specifying production functions for ecosystem services that are sensitive to variations of key factors that are subject to management, for example wetland water levels or flooding frequencies, and that are able to generate information in a form amenable to economic analysis, remains a major challenge. Whilst the work presented in this thesis largely builds on available approaches, this study has taken previous research further by constructing and coupling water management sensitive production functions with large scale assessment of the hydrology and water management of wetlands.

Such approaches need to be sufficiently detailed to capture variations in water management but not overtly complex to allow integration into water management modelling frameworks at the spatial and temporal scale of the assessment problem. For example, this thesis presented a rapid flood risk assessment methodology for the River Elbe based on relatively sparse data, that can be applied at the scale of the complete river trajectory and that allows for the assessment of combinations of retention measures of various proposed dimensions and at multiple locations (Paper 1 "flood risk"). Likewise, a cost-minimisation model for nutrient abatement measures for the complete Elbe Basin was developed, that includes wetlands as management options (Paper 5 "nutrient retention"). In another case study, greenhouse gas emission factors where combined with a dynamic hydrologic modelling framework for lowland wetlands (Paper 4 "greenhouse gases"). Such approaches were not previously available.

The productions functions for the regulatory ecosystem services describe above are largely dependent on the understanding of physical, geochemical or biological processes. Whilst approaches for water dependent production of agricultural biomass have long been developed in the context of irrigation planning, comparable approaches for other ecosystem services are only slowly forthcoming. For example, in recent years increased attention has being devoted to quantifying the greenhouse gas emissions from peat wetlands in Germany, mainly to provide a better scientific basis for emissions accounting under the United Nations Framework Convention on Climate Change. Regarding the nutrient retention capacity of restored floodplains, large uncertainty remains and there is continued need for the development of a robust functional approach. Comparable to the case of greenhouse gases emissions, it can be expected that the continued need to develop plans for water quality improvements under the EU Water Framework Directive will provide an impetus for the development of more standardised prediction approaches in this field.

It is much more difficult to establish the causal links between specific changes in wetland management and the intensity of recreational use or the preferences for wetland habitat conservation. Generally, determining the linkage between water flows or ecosystem services and opportunities for recreation is feasible, but difficult. This becomes easier to the degree that the recreational usage is directly determined by the availability of a specific water flow or ecosystem service. This thesis has presented an example for an aspect of water based recreation (Paper 2 "travel cost method"), that is rather directly dependent on flows: boating. In contrast, it is much more difficult to develop descriptions or indicators of wetland quality at the level of landscape and habitat diversity that can be used in the context of studies to elicit effects on general recreational use of the landscape or non use values. This thesis has attempted to describe the commodity "wetland habitat conservation" according to two dimensions: quality and quantity of changes in wetland habitat (Paper 3 "meta analysis"). For the purpose of benefit transfer it is assumed that the valuation scenarios offered to respondents in valuation studies to imply that the measures are suited to maintain or restore something like a "good wetland habitat quality status". It had to be assumed that the quality dimension of the wetland habitat and biodiversity conservation commodity can be perceived as a relatively homogeneous good. This makes the application of valuation estimates to value gradual differences in the level of ecosystem service provision in a cost benefit analysis difficult. There is need for more research here, for example in generating a common metric to describe quality changes (something like a "wetland quality ladder"). However, when calling for more primary studies to augment the sparse evidence on the recreational (and non use) values, there is also a need for more precise definition and documentation of the valuation scenarios and more complete reporting of information on the samples to enhance the usability of such studies for benefit transfer.

Transferring values: service benefit areas and scope of changes in service provision.

Besides the spatial and temporal variation in the production of ecosystem services, this thesis has analysed in greater detail several contextual factors that determine the value of ecosystem services. Such factors are important to the further use of the generated value estimates for the transfer to other valuation contexts (cf. Navrud and Ready 2007 on "value transfer"). One of the roles of strategic economic assessments, as presented in this thesis, is to provide information that can subsequently be used in more detailed project based cost benefit analysis or as a secondary environmental benefit (co-benefits) in sectoral cost-effectiveness analysis. In an ideal case, this would facilitate a coherent evaluation approach across different projects and policies. A second reason that may make recursion to benefit transfer necessary is the high costs associated with generating new evidence on the value of ecosystem services. This is particularly relevant for stated or revealed preference methods that require interview surveys to elicit primary value estimates for a proposed policy or measure. But also other methods, for example a cost minimisation model approach, require time and experience. Where such research is not possible or not justified because of budget and time constraints, benefit transfer is a second best strategy. There are two broad approaches to benefit transfer: unit transfer and function transfer (Navrud and Ready 2007). Unit transfer encompass the transfer of a single point estimate from a study site or a measure of central tendency from several benefit estimates from several sites (average value) derived by meta-analysis. Function transfer encompasses the transfer of a valuation function from a single study or a meta-regression function derived from several studies. Function transfer then adapts the parameters of the function to fit the specifics of the policy site such as socioeconomic characteristics of the population, extent of market and scope of the resource.

Even though benefit transfer is mostly discussed in relation to stated and revealed preference studies (cf. Bergström et al. 2006), it can also be used for all other types of value estimates. For example, several of the existing meta-analyses of wetland valuation studies (Woodward and Wui 2001, Brander et al. 2006 and Gerhimandi et al. 2008) pool value estimates based on different value concepts, such as cost based, producer surplus and consumer surplus based values. However, this thesis argues, that pooling value estimates for different ecosystem services makes the identification of the appropriate moderator variables that are required to make adjustments for benefit transfer difficult (Paper 3 "meta-analysis"). These variables essentially depend on the determinants for demand that are relevant for the different goods (such as fodder, reeds, timber, fuel wood) and regulating services (such as nutrient retention, flood risk reduction, greenhouse gas sink) that a wetland provides.

The valuation studies presented in this thesis all address spatial aspects of contextual modifiers regarding both the aggregated demand for final ecosystem services within the service benefit area and the change in availability of ecosystem services (scope) that need to be taken into account. The thesis addresses several important spatial determinants of aggregated demand for ecosystem services.

With regards to recreational and non-use values, demand is modelled as a function of the size (and share) of the population that hold a value for wetland habitat conservation or are active recreational users of a resource (Paper 2 "travel cost method" and Paper 3 "meta analysis"). Key modifying factors that where addressed (explicitly or implicitly) in the travel cost method and the meta functional approach to stated preference value estimates are the quality of the site, the availability of substitutes and the distance of the population to the site. Whereas the meta functional approach is explicitly constructed to be able to address the modification of these variables in a benefit transfer exercises, the value estimates provided by the travel cost application can only be transferred on a per unit value to sites that have similar characteristics regarding the user population and the quality of the site. The aggregate demand is the product of individual demand and the size of the market population. Correctly specifying the population over which to aggregate non use values is as important as the precision of the willingness to pay estimate for estimating the resultant aggregate demand. However determining the correct population is fraught with difficulties related to lacking evidence in many of the valuation studies regarding distance decay of willingness to pay. In contrast, the aggregation of recreational benefits based on per trip values generated by the travel cost method is comparatively straightforward, if data on aggregate recreational use of a site is available.

The regulatory or intermediate ecosystem services addressed in this thesis were all valued using indirect, cost based methods. These methods only produce valid estimates if it can be reasonably assumed that there is actually a demand for the provided service at the assumed prices for the repair of damages or the considered abatement alternatives. All of the cost based valuation approaches presented here assume that there is such a demand for these services. This aggregate demand is assumed to be equivalent for example to the net value of elements at risk from flooding (Paper 1 "floodrisk" and 6 "CBA floodplain") or the lowest costs incurred to achieve a water quality target set out under a water management policy (Paper 5 "nutrient retention" and 6 "CBA floodplain"). The aggregate demand has to be specified for the service benefit area of a wetland, where the services provided by the ecosystem can be a potential perfect substitute for other measures to provide the finally demanded benefit. For the two examples of flood water and nutrient retention the service benefit areas are directional towards downstream section of the river basin. Given a certain distribution of emissions and abatement options in a river basin, the water quality targets for river sections below the wetland are shown to be key determinants of the value of the nutrient regulating ecosystem services provided by wetlands (Paper 5 "nutrient retention"). Likewise, it is shown that the value of reduced flood damages is dependent on the aggregate value and distribution of elements at risk from flooding below the wetland (Paper 1 "flood risk"). In contrast to these examples, the service benefit area for greenhouse gas regulation is omni-directional and quality targets are homogenous in space, i.e. not spatially confined to a river basin (Paper 4 "greenhouse gases"). The value estimates for greenhouse gas emissions therefore do not require any adjustment to a local or river basin context. It can be concluded that requirement for contextual adjustments decreases with spatial specificity of the demand for a service – for the considered regulating ecosystem services this would then be in decreasing order from greenhouse gas regulation, nutrient retention and flood risk reduction.

Provided that wetland conservation is a normal good, economic theory would also require the value estimates to be sensitive to the scope of changes in the availability of ecosystem services. The expectation is that the marginal value decreases with increasing availability. This thesis provides evidence on the decreasing marginal benefits to a range of wetland ecosystem services. This is shown, with qualifications, with respect to the effects of increasing volumes of flood retention capacity for the value of avoided damages (Paper 1 "floodrisk"), for the increasing capacity of nutrient retention by floodplain wetlands on the replacement value (Paper 5 "nutrient retention") and the increasing area of restored wetlands sites on the willingness to pay by the general populace for wetland habitat conservation (Paper 3 "meta-analyis").

Taken together, the results imply that simply multiplying with a constant unit value for ecosystem service benefits will lead to an undervaluation of a negative change or an over estimate of the value of an improvement. Appropriate adjustments to marginal values to account for demand and scope effects in ecosystem service provision are thus required. This is not always an easy task, but this thesis has presented various approaches that facilitate the appropriate scaling of ecosystem service value estimates.

Uncertainty of decision relevant information

All value estimates presented in this thesis are subject to uncertainty, stemming from uncertainty about the models of ecosystem service production and the valuation approaches. Important sources of this uncertainty relate to the modelling and valuation approaches themselves and to the uncertainty regarding the available data used for the specification of model parameters.

The approaches to valuation presented in this thesis where all geared in first line to reduce uncertainty about the value of ecosystem services by reducing model uncertainty. Each of the approaches was designed to improve upon previous valuation studies by more explicitly taking spatial and temporal variations of key determinants of ecosystem service provision and value into account – such as scope of restoration efforts, spatial distribution of nutrient emissions in a basin, variability of climatic conditions or water availability. A central contribution of this thesis is also a consequent application of a risk based approach to reduce uncertainty about ecosystem service benefits by systematically accounting for the variation in water flows on the value of ecosystem services (Paper 1 "flood risk"; Paper 4 "greenhouse gas emissions", Paper 6 "CBA floodplain", Paper 7 "CBA water level regulation", Paper 8 "CBA low flows").

The studies presented also address aspects of uncertainty related to variations in underlying data. Specifically, for the travel cost application (Paper 2 " travel cost") and meta- analysis (Paper 5 "meta-analysis") that were conducted, implications of measures of statistical uncertainty were discussed. However, for value estimates that were derived from complex simulation models, no statistical uncertainties were generated. In these cases only sensitivity analysis on key parameters was conducted, for example regarding the specification of the green house gas emission function (Paper 4 "greenhouse gases") or the assumed width of dike breaches (Paper 1 "flood risk"). Greater use of Monte-Carlo simulation, for example in flood risk appraisal, that takes into account known error ranges for the underlying data and parameters, would help to generate a better understanding of this aspect of uncertainty in results from simulation models.

Taken together, all measures of uncertainty generated in this thesis show rather high margins of errors and ranges of plausible variation. Whether this margin of error is considered large or too large depends on the use of the results. For some projects and policy applications it is must be considered acceptable and uncertainty of the final results can be dealt with through sensitivity analysis in the subsequent cost benefit analysis. This could be enhanced if better use of uncertainty estimates based on statistical uncertainties generated for each of ecosystem services could be made in more sophisticated approaches to dealing with uncertainty in cost benefit analysis. However, the sensitivity analysis conducted in this thesis, all indicate that the basic empirical findings of potential efficiency gains from wetland restoration are stable over a large range of plausible uncertainty ranges for the value of wetland ecosystem services.

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Valuation approaches

1. De Kok, J.L. & Grossmann, M. (2010): Large scale assessment of the flood risk and the effects of mitigation measures along the Elbe River. *Natural Hazards 52 (1): 143-166.*

2. Grossmann, M. (2011): Impacts of boating trip limitations on the recreational value of the Spreewald wetland: a pooled revealed / contingent behaviour application of the travel cost method. *Journal of Environmental Planning and Management 54 (2): 211 - 226.*

3. Grossmann, M.: Accounting for scope and distance decay in meta-functional benefit transfer: an application to the willingness to pay for wetland conservation programmes in Europe. *Manuscript*

4. Grossmann, M. & Dietrich, O. (accepted): Social benefits and abatement costs of greenhouse gas emission reductions from restoring drained fen wetlands. A case study from the Elbe River Basin. *Irrigation and Drainage*

5. Grossmann, M. (accepted): Economic value of the nutrient retention function of restored floodplain wetlands in the Elbe River Basin. *Ecological Economics.*

Integrated assessment

6. Grossmann, M. & Hartje, V.: Strategic cost-benefit analysis of an integrated floodplain management policy for the Elbe River. *Manuscript*

7. Grossmann, M., Dietrich, O. (accepted): Integrated economic-hydrologic assessment of water management options for regulated wetlands under conditions of climate change: a case study from the Spreewald (Germany). *Water Resources Management*

8. Grossmann, M., Koch, H., Lienhoop, N., Vögele, S., Mutafoglu, M., Möhring, J., Dietrich, O., Kaltofen, M.: Economic risks associated with low flows in the Elbe River Basin (Germany): an integrated economic-hydrologic approach to assess vulnerability to climate change. *Manuscript*

Statement of author's contribution

De Kok, J.L. & Grossmann, M.: Large scale assessment of the flood risk and mitigation measures along the Elbe River.

MG developed the management options, developed the flood damage assessment method, compiled the database of dike heights, flood damage potential and retention sites using GIS analysis, and analyzed the results. JLK developed the hydraulic model and implemented the integrated model code. MG and JLK wrote the corresponding sections of the manuscript.

Grossmann, M. & Dietrich, O.: Social benefits and abatement costs of greenhouse gas emission reductions from restoring drained fen wetlands..

OD developed the water resources management model for lowland wetlands and conducted the hydrological analysis. OD and MG developed the management scenarios. MG developed the greenhouse gas emission functions and the cost estimates, conducted the analysis of results and wrote the manuscript.

Grossmann, M. & Hartje, V.: Strategic cost-benefit analysis of an integrated floodplain management policy for the River Elbe.

MG developed the conceptual design of the study, the cost benefit model, the management scenarios, conducted the analysis of results and wrote the manuscript. VH contributed to the conceptual design and discussion of the study results.

Grossmann, M. & Dietrich, O.: Integrated economic-hydrologic assessment of water management options for regulated wetlands under conditions of climate change: a case study from the Spreewald (Germany).

OD developed the hydrological management model for lowland wetlands and conducted the hydrological analysis and contributed to writing the water resources related sections of the manuscript. OD and MG developed the management scenarios. MG developed the ecosystem service production functions, the valuation approach and the cost estimates, conducted the corresponding analysis of results and wrote the manuscript.

Grossmann, M., Koch, H., Lienhoop, N., Vögele, S., Mutafoglu, K., Möhring, J., Dietrich, O., Kaltofen, M.: Economic risks associated with low flows in the Elbe River Basin (Germany): an integrated economic-hydrologic approach to assess vulnerability to climate change

MG led the development of the risk based assessment framework, conducted the analysis of results and wrote the manuscript. The supplement was compiledby MG based on descriptions of the valuation approaches provided by the coauthors. The economic valuation approaches for irrigation, wetlands, recreational boating and hydropower were developed by MG and for shipping by MG and JM. The economic valuation approaches for industry was developed by KM, for power plants by SV and HK, for pond fisheries and municipal water supply utilities by NL. The various valuation approaches where implemented into the hydrological modeling framework by HK, OD and MK.

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PAPER I

LARGE-SCALE ASSESSMENT OF FLOOD RISK AND THE EFFECTS OF MITIGATION MEASURES ALONG THE RIVER ELBE

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The downstream effects of flood risk mitigation measures and the necessity to develop flood risk management strategies that are effective on a basin scale call for a flood risk assessment methodology that can be applied at the scale of a large river. We present an example of a rapid flood risk assessment methodology for the River Elbe. A 1D hydraulic routing model is extended by including the effect of planned (regulated and unregulated) and unintended retention (dike breaches) on the peak water levels. We further add an inundation model for dike breaches due to dike overtopping and a macro-scale economic approach to assess the flood damage. The flexible approach to model the effects of measures by means of volume storage functions allows for rapid assessment of combinations of retention measures of various proposed dimensions and at multiple locations. The method allows for the comparison of the flood risk at the scale of the main river trajectory, which has not been possible for the River Elbe to date. The model is applied to a series of exemplary flood risk mitigation measures to show the downstream effects and the additive effects of combinations of measures on the flood risk along the river. We further demonstrate the increase in the downstream flood risk resulting from unilateral decisions to increase the dike height at upstream locations. As expected, the results underline the potential effectiveness of increased retention along the river. The effects of controlled retention at the most upstream possible location and largest possible extent generate the most pronounced reduction of average annual damage. As expected, the effect of uncontrolled retention with dike relocations is significantly lower.

Keywords: flood risk assessment, 1D hydraulic routing model, macro-scale damage assessment, floor risk mitigation measures

1 Introduction

During the flood catastrophe of August 2002 the river Elbe and its tributaries were heavily affected in terms of damage (IKSE, 2004a). Efforts to improve flood risk management have increased as a result (Petrow et al., 2006). For example, the International Commission for the Protection of the Elbe (IKSE) formulated a flood action plain (IKSE, 2004b), in which potential measures, such as the reactivation of retention capacity in the floodplains, increased storage capacity in upstream reservoirs, improvement of the existing river dikes, and flood preparedness are proposed. Within the concept of integrated floodplain and river basin management, it is not only the reduction of flood risk that guides the future development of the floodplains. Other goals to be taken into consideration are, for example, the restoration of the ecological function of floodplain habitats, the improvement of the nutrient retention capacity of the floodplains or the capacity of the river as an important waterway for transport.

The Elbe pilot Decision Support System or Elbe DSS is an integrated tool aimed to promote the discussion on integrated river basin management by enabling the analysis and comparison of different long-term strategies that take multiple river functions into account. The DSS is described in detail by De Kok et al. (2008) and Berlekamp et al. (2005). To take into consideration the inherent tradeoffs between goals and possibly conflicting interests of different stakeholders, the effects of different interventions on different goal indicators such as flood risk have to be considered simultaneously. This paper reports on the approach chosen to integrate a rapid flood risk assessment approach into this model system. The application in an interactive decision-support system (DSS) calls for flexible models that are easy to set up and adapt to changing user demands. Comprehensive 2D hydrodynamic models are well able to capture the dynamics aspects of a flood, but the data requirements and computational load make these models less practical for application in a large-scale risk assessment, particularly when multiple scenarios have to be analyzed and compared interactively, for example during sessions with stakeholders (Apel et al., 2006). An ongoing development is the application of 1D models combined with volume storage functions derived from GIS analysis for large-scale risk assessment (De Roo et al., 2000; ICPR, 2001; Zerger, 2002; Förster et al. 2005; Knebl et al., 2005; Apel et al., 2006; Lindenschmidt et al., 2006), but the majority of these studies pertain to only a small section of the whole river trajectory. As the intention of the Elbe DSS was to apply existing models as much as possible, the choice was made for an existing 1D hydraulic model as the basis for large-scale flood risk assessment.

The large extent of the Elbe floodplains and the absence of any previous economic evaluation at the scale of the river required the development of an innovative rapid assessment approach. We therefore combine the 1D model with a macro-scale approach for damage assessment that is based on the method originally developed for assessing flood risk for the Rhine Flood Action Plan and Rhine Atlas (ICPR, 2001). Macro-scale approaches have been used for risk assessment on a large scale with scarce data in several studies (Penning-Rowsell et al., 2003; Meyer and Messner, 2005; Messner et al., 2007). See Meyer and Messner (2005) and Messner et al. (2007) for a review of applications. Of the federal states along the Elbe, the State of Sachsen has recommended a damage assessment method which is also based on the Rhine Atlas Method (LTV, 2003). The state of Mecklenburg followed a different approach, which is an adaptation from the so called German meso-scale approach (Messner et al., 2007). This approach was also recommended for flood risk assessment along the Elbe in the IKSE Action Plan (IKSE, 2004b), but it has not been elaborated since. An abridged version of this approach has also been used by Förster et al. (2005) for their assessment of the mitigating effects of a number of large retention polders at the mouth of the Havel River. Other federal states do not give a recommendation for a river flood damage assessment method. Currently, efforts are under way to further improve the basin-scale analysis of the flood risk in the Elbe River (e.g. VERIS, 2008).

The aim of this paper is to examine the usefulness of the combined flood risk assessment approach which has been used in the Elbe DSS. This approach comprises four steps: the generation of artificial flood events based on statistical analysis of hydrological data, the routing of the flood event along the river including the effect of controlled retention and dike breaches, inundation modeling, and modeling of the expected damage. We demonstrate possible applications of the method by analyzing various risk mitigation measures discussed in the IKSE action plan as case examples. The measures consist of various combinations of dike heightening and operation of (un)controlled retention polders along the river.

This paper is organized as follows. Section two introduces the study area and the risk mitigation measures that we consider. The methodology is outlined in section three. We begin with the routing of flood events and inundation model, followed by a presentation of the approach for flood damage assessment. The outcomes of the flood risk assessment are presented in section four. The paper ends with a conclusions section, elaborating on the effectiveness of measures, potential bottlenecks of the method, and room for future research.

2 Case study description

The German part of the Elbe catchment (Figure 1) covers an area of 97175 km^2 and has 18.5 million inhabitants. The Elbe River has characteristics of a lowland river with large floodplains downstream of Dresden. Approximately 80% of the floodplains along this river stretch are protected by dikes. The generally desired design standard for dikes protecting settlements is a recurrence interval of 100 years plus a 1 m freeboard. Protection standards in rural areas are generally lower, for example a recurrence interval of 25 years is recommended for single buildings and for agricultural areas a recurrence interval of 5 years (LTV, 2003). In 2000 over 470 km of dikes required maintenance (IKSE, 2004b). The actual freeboard varied between -1.70 m and +1.30 m. The flood of August 2002 was due to extreme rainfall in the Elbe catchment, and was estimated to have resulted in 6.2 billion € damage in the state of Sachsen, 1 billion € in Sachsen-Anhalt, 0.2 billion € in Brandenburg, 0.2 billion € in Niedersachsen, 0.04 billion € in Mecklenburg Vorpommern and 0.004 billion € in Schleswig-Holstein (IKSE, 2004a). Twenty-one dike breaches occurred along the River Elbe. The peak discharges in the Elbe river during the flood are estimated to have had a recurrence interval of 200 years (IKSE, 2004b). The technical condition of about 45 % of the 1200 km long Elbe dikes was considered to be insufficient (IKSE, 2004b). This partially explains the large number of dike breaches. To what extent upstream dike breaches lowered the downstream water levels is not known exactly, but it can be assumed that some areas escaped inundation in this way (Apel et al., 2006). The purpose of the IKSE action plan (IKSE, 2004b) is to develop a comprehensive flood risk management strategy for the river. The proposed measures include amongst others, reactivation of the retention capacity along the river floodplains and reconstruction of dikes to the desired safety standard. Dike shifting has been proposed and discussed mainly as a nature conservation measure (cf. Meyerhoff and Denhardt, 2007), but realignment may also prove beneficial by shortening the dike line. Since the flood of 2002, both the IKSE and the German federal states have commissioned a series of studies to evaluate potential sites for dike relocations and retention polders (Figure 1).

Table 1: List of the risk mitigation strategies that were compared.

Strategy	Type	River stretch (km)	Capacity or magnitude
D S + 1	Dike heightening	60 - 180	+ 1 m along 60 km
DR I	Dike shifting	117 - 536	738 million m³
DR II	Dike shifting	120.5 - 536	251 million m³
POL A	Controlled retention	117 - 427	494 million m³
POL P	Controlled retention	180	138 million m³
POL H	Controlled retention	427	112 million m³

The number, exact location, area and retention volume of potential sites is the subject of public debate and constant review. Within the Pilot Elbe DSS, the proposed sites and dimensions from four data sources were included (Merkel, 2002; Ihringer et al., 2003; IKSE, 2004b; Förster et al. 2005). In case of divergent information on dimensions for a site, the larger alternative was chosen for this study.

For analytical purposes this paper concentrates on six combinations of measures (see Table 1) which were chosen to illustrate the magnitude of effects which could be achieved.

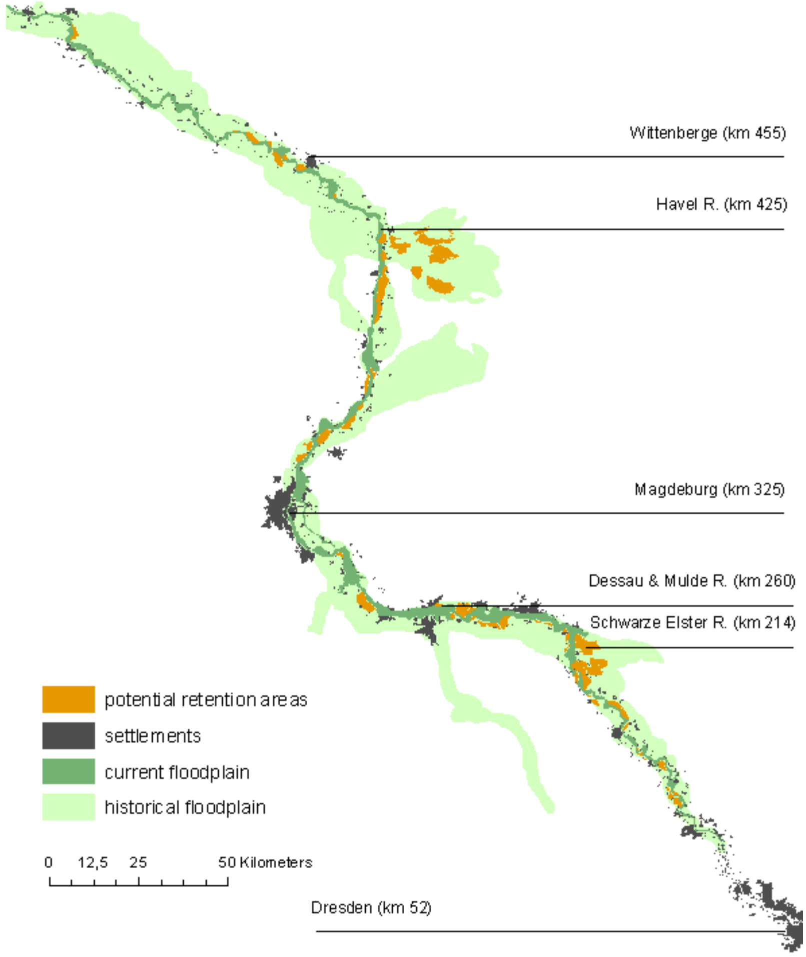

Figure 1: Map of the study area showing the location of the potential retention areas.

The strategies are compared to the baseline scenario that describes the situation as outlined in the flood action plan for the year 2000. Recent improvements of the dikes since the flood are not included in the database. The first strategy looks at effects of dike heightening, the second two strategies compare retention effects of dike relocation measures of different magnitude and the last three strategies compare the retention effects of controlled polders of different magnitude and location:

strategy D S +1: implementation of the design standard of a 100-year recurrence interval with an additional freeboard of 1 m for all dikes in Sachsen for which this protection standard is stated. The total length of the modified dikes is 60 km in the river stretch between Elbe km 60 and Elbe km 180. The purpose is to compare the upstream damage reduction with possible increases of the flood damage downstream.

strategy DR I: dike relocation (uncontrolled operation) of all 60 potential sites included in the database irrespective of their designation for dike relocation along the river stretch Elbe km 117-536. The total floodplain area is 34658 ha with a storage capacity of 738 million m^3. The purpose is to examine the potential effects of a dike relocation program which is much larger than the 15 000 ha analyzed in Merkel et al. (2002) or otherwise currently under discussion.

strategy DR II: dike relocation (uncontrolled operation) of the 33 potential sites identified in the IKSE action plan (IKSE, 2004b) in the river stretch Elbe km 120.5-536. The total area is 9432 ha with a storage capacity of 251 million m^3. The purpose is an assessment of a flood risk mitigation program of a realistic dimension as is currently being discussed.

strategy POL A: controlled operation of 31 potential sites for retention polders identified in IKSE (2004b) along the river stretch Elbe km 117-427 with a total area of 25 576 ha and a total storage capacity of 494 million m^3. The polders in Sachsen-Anhalt are dimensioned according to Ihringer et al. (2003) and the polders on the Havel are included and dimensioned according to Förster et al. (2005). The purpose is an assessment of the hypothetical maximum attainable damage reduction through the retention effect.

strategy POL P: controlled operation of only the largest 5 potential sites for retention polders identified in Ihringer et al. (2003) near Elbe km 180 with a total area of 4557 ha and a storage capacity of 138 million m^3. The purpose is to assess the contribution of the largest upstream sites to the maximum attainable damage reduction of alternative POL A.

strategy POL H: controlled operation of the 8 existing retention polders at the mouth of the River Havel near Elbe km 427 with a total area of 9909 ha and a capacity of 112 million m3. The purpose is to illustrate the effect of a set of major retention polders in the middle reaches.

3 Methodology

The approach is based on four consecutive steps: generating flood events, modeling dike overtopping and inundation, flood damage assessment, and assessment of the flood risk (Figure 2).

3.1 Generating flood events

Due to limitations in the availability of quality discharge data the large-scale risk analysis in the Elbe DSS takes an artificial flood event at the gauge station of the city of Dresden, 56 km downstream of the Czech-German border, as the starting point. Daily average discharge data for the gauge stations of the Elbe have been collected since 1853 and were subject to a detailed statistical analysis (Helms et al., 2002a, Helms et al., 2002b) aimed at the regionalization of the hydrological parameters along the German section of the river. For longer time periods (e.g. 1936-1995, 1903-1995) the data quality was considered insufficient due to human modifications to the river and changes in the basin hydrology (Helms et al., 2002a). Instead the more reliable and hydrologically consistent discharge data for the period 1964-1995 have been used. A regionalized flood frequency analysis of these data resulted in longitudinal sections for the yearly peak discharges along the river stretch Elbe km 0-536 (Helms et al., 2002a). The 1964-1995 flood frequency analysis has also been used to generate an artificial flood event with a 100-year recurrence interval for the gauge station at Dresden (Merkel et al., 2002, Helms et al., 2002b).

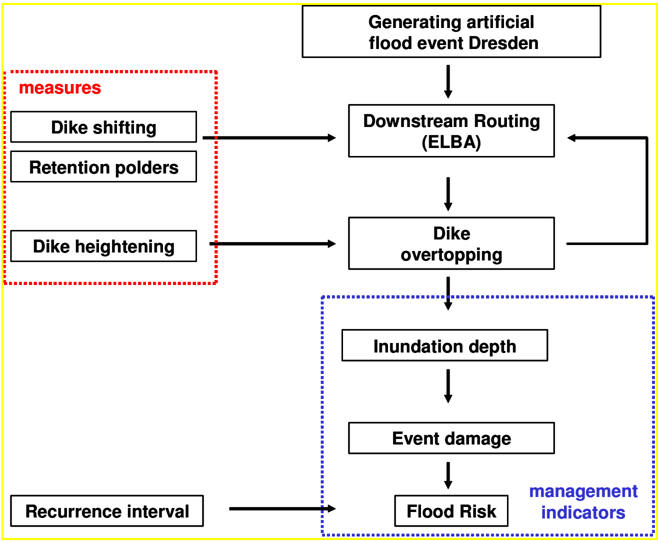

Figure 2: Methodology for flood risk assessment.

To obtain the flood events with a different recurrence interval the discharge values for the 100-year event are rescaled on the basis of the ratio of the peak discharges. This is justified by the fact that the peak discharge is the most relevant parameter for dike overtopping and inundation. There exist also a number of major tributaries along the Elbe River, the outflows of which play an important role. The generation of consistent discharge data for the tributaries is a hydrological challenge beyond the scope of the pilot DSS. Therefore, the contributions of the three main tributaries, the Schwarze Elster, the Mulde and the Saale, have been generated artificially corresponding to the statistical analysis of the Elbe data (Helms et al., 2002a) and are rescaled proportionally to the discharge for the main channel.

To route the flood hydrographs downstream along the main channel the German Federal Institute of Hydrology developed the 1D translation-diffusion model ELBA (Fröhlich 1998, Busch et al., 1999). This empirical model was developed for quick routing of flood events along rivers such as the River Elbe. The model has been calibrated for seven sections along the Elbe river. In the model three discharge regimes are distinguished, which can be superimposed. The hydrograph is separated into one hour pulses, which are multiplied with a dimensionless system function for routing along the river sections:

$$h(t) = \frac{L}{2t\sqrt{\pi Dt}} \exp\left[-\frac{(ut - x)^2}{4Dt}\right] \tag{1}$$

where *h(t)* is the system function, *L* is the length of the modeled river section in km, *u* is the translation coefficient in km per hour, *D* is the diffusion coefficient in km^2 per hour, and *t* is the time step in hours. The model parameters were determined for seven river sections in the trajectory downstream of Dresden for three discharge regimes. For parameter values we refer to (Helms et al., 2002b).

3.2 Inundation modeling

The peak water levels in the main channel were determined by means of stage-discharge relationships which are available every 500 m. These have been determined with the 1D steady-flow hydraulic model HEC-6 (Otte-Witte et al., 2002) for discharge values up to the peak discharge with a recurrence interval of 100 years. The functions have been extrapolated to cover discharges with a recurrence interval in the range 100 – 1000 years. The information on the dikes is based on the 2001 status report on the Elbe dikes (IKSE, 2001), which comprises the design flood recurrence interval and an additional freeboard value for each dike section. GIS analysis was used to generate a geo-referenced map of the positions of each dike section (Jankiewicz et al., 2005). The dike segments and floodplain area were allocated to the river kilometrisation for every 100 m stretch of the main channel using the closest distance function. The dike height above sea-level was derived from the water level corresponding to the design recurrence interval plus the given freeboard. This approach was chosen to calibrate the dike heights to the river kilometrisation and corresponding discharge-stage functions. The wide floodplains of the lowland Elbe are compartmentalized into separate areas by dikes and natural areas of high ground. Seventy-one compartments of the floodplain with corresponding dike segments on both sides of the river were delineated by extrapolating the water level corresponding to a 200-year peak discharge from the main channel into the floodplain. Whilst the compartmentalization is obvious for many areas, in the very wide floodplains in the vicinity of confluences with tributaries such as the Havel the compartmentalization is more difficult to implement, also because the barrier effects of structures such as roads embankments are unclear.

In case of overtopping of the dike the most upstream overtopped dike cell with the lowest recurrence interval is assigned as overtopping location for the inundation of the protected area. In view of the computational efficiency the inundation process is not modeled within the Elbe DSS but determined on a 100x100 m grid by means of precomputed volume storage functions. The flooding volume depends on the water level in the main channel, the inflow rate at the location of the dike breach, and the capacity of the protected area, and is directly translated into inundation depths, using the volume storage functions and the available elevation data (BKG, 2003). The inundation depths in areas of the floodplain not protected by dikes are determined directly from the water level and elevation data.

The probability of a dike breach due to overtopping depends on the duration of the overtopping and the overtopping height (Apel et al., 2006; Kamrath, 2006). Analysis of the dike failure probability for the Rhine River (Apel et al., 2006) showed that this probability approaches 100 % in the case that the overtopping time is more than a few hours and/or the overtopping height exceeds 10 cm. Here the flood events are described with a one-day time step and it is assumed that these conditions are met for all dike overtopping locations, with a dike breach as certain consequence.

The effect of dike overtopping on the downstream peak water levels is included in the analysis. This makes it possible to analyze flood events at the scale of the complete modeled trajectory. The retention effect of dike overtopping instances on the downstream peak discharges is relevant. For example, in case of a mean inundation depth of 2 m, the stored flood volume in the floodplain protected by dikes can exceed 100 million m^3, which is an order of magnitude larger than the capacity of most proposed retention polders (see section 2). For a large-scale risk analysis it is therefore essential to include this effect if one wishes to compare the potential damage between different locations and consider the analysis as an event at the river scale.

To include the consequences of the potential dike failures on the peak discharges in the main channel the shape of the flood event is corrected by assuming an inflow based on the simple weir overflow equation (Chen, 1995):

$$q_{inflow} = \frac{2}{3} B(2g)^{1/2} h^{3/2} \tag{2}$$

where q_{inflow} is the inflow rate in m^{3}s^{-1}, B is the width of the dike break, g is the gravitational acceleration in ms^{-2}, and h is the head difference in m between the water level in the main channel and the lowest point of the dike breach, which was assumed to follow the overtopping. During the 2002 flood the dike breaches that occurred along the Elbe River varied in size between 20 and 200 meters, with the most frequent value being ca. 20 m. A log-normal distribution with a mean of 64 m was fitted to dike breach width data for the Elbe 2002 flood from Gocht (2002) and Horlacher et al. (2005) and this mean was used for all dike overtopping locations. Apel et al. (2004), on the basis of case reports, assume that the range of breach width on the lower Rhine is 100–400 m, whereas Kamrath et al. (2006) assume a breach width ranging from 50-150 m. A standard value of 3 m was used for the head difference, approximating the lowest dike height according to the IKSE tables (IKSE, 2001).

Within the Elbe DSS the effects of three types of flood risk mitigation measures can be modeled. The first is the heightening of dikes by section; the other measures are reactivation of the floodplain retention capacity by dike relocation or the construction of flood retention polders. Both retention options are conceptualized as retention polders with the difference that the flooding process of dike relocations is uncontrolled, whereas the retention polders are flooded in a controlled way. In the first case the retention polders are considered to be always open and flood freely with the rising flood wave up to the maximum capacity. In the case of controlled operation the polders are assumed to be opened at the optimum time to lower the peak discharge to the maximum extent possible. The effect of retention polders on the peak discharges in the main channel is modeled in a way similar to the modeling of the dike breaches with the help of volume storage functions.

3.3 Damage assessment

The damage assessment is based on a modification of the method that was originally developed for assessing the flood risk for the Rhine Flood Action Plan and Rhine Atlas (ICPR, 2001). Key characteristics of this method are (a) the application of relative damage functions and (b) a macro-scale approach for describing the value of elements at risk. Relative damage functions describe the flood damage as a percentage of the value of the element at risk as a function of the inundation depth. In contrast to object-oriented approaches, macro-scale damage assessment methods are characterized by a very high level of aggregation of the data describing both the spatial distribution and the value of the elements at risk. We use the CORINE land cover data (EEA, 2002) that were derived from remote sensing data, to characterize the spatial distribution and data from national accounting to estimate the value of the elements at risk. Whilst details will be presented below, the principle of the macro-scale method is that the total values of the elements at risk for an administrative region are divided by the area of the corresponding land use class in that administrative area to derive the specific value densities. The method thus makes the implicit assumption that the values of elements at risk are completely homogenous regarding their characteristics and distribution within the corresponding land use class.

For the implementation of this approach a selection of elements at risk to be considered has to be made and three harmonizing sets of data have to be generated: (a) the value density of each element at risk (b) a map of the spatial distribution of the elements at risk, and (c) a specification of the damage functions describing the damage as a function of the inundation depth. Whereas we apply the damage functions developed for the Rhine Atlas Method directly, we develop new estimates of the specific value densities on the basis of statistical data for the former East German federal states of Thüringen, Sachsen, Sachsen-Anhalt, Brandenburg and Mecklenburg. The reason is that no effort has been made to date to develop the value densities required for the application of this method for the East German states. The specific value densities can be expected to diverge from the Western German values because of the post-socialist transformation of the regional economy in Eastern Germany. We further compare our method with four alternative adaptations of the Rhine Atlas methodology to the Elbe River Basin in order to give some indication of variability of results associated with different possible implementations of the method. The considered elements at risk and the corresponding CORINE land use classes, value densities and damage functions of all the damage assessment approaches are summarized in Table 2 and discussed in more detail below.

The classification of the elements at risk follows the classification of stocks and flows typically accounted for in the expenditure approach of national accounting. The expenditure approach measures the total expenditure on final goods and services produced in the domestic economy within a year. A stock variable is measured at one specific time, and represents a quantity existing at that point in time, which may have been accumulated in the past. The capital stock is the total value of equipment, buildings, inventories, and other assets in the economy. The stock of capital is increased by the flow of new investment and depleted by the flow of depreciation. Of all stocks, we only consider the stock of consumer durables of households, tangible fixed assets (constructed assets and machinery and equipment), and inventories of producers (industrial, commercial and agricultural sector). Currently only information on the stock of tangible assets is available from the official statistics (Statistisches Bundesamt, 2003), so that separate estimates for consumer durables and inventories had to be determined. The data on tangible fixed assets are subdivided into constructed assets and machinery and equipment. Constructed assets are further subdivided into buildings and traffic infrastructure, and machinery and equipment are further subdivided into machinery, equipment, and vehicles. In addition, we developed estimates of the stock of consumer durables, the inventories (or standing crop) of the agricultural sector (livestock, grassland, arable land, forest) and the inventories of the commercial and producing sector. For all value estimates the net concept is applied, which means that the consumption (depreciation) of fixed capital accumulated since the time of investment is deducted. The net concept is the correct concept for flood damage assessment because the damage to the economy would be overestimated if full replacement (or gross) values are used (Penning-Rowsell et al. 2003).

We used data on the net value of fixed assets at the level of the federal states (Statistisches Bundesamt, 2003). The data on constructed assets are subdivided into buildings and traffic by assuming a specific net value of 50 € m^{-2} for traffic (road and railway) infrastructure (cf. Meyer, 2005) and attributing the remaining assets to built stocks. The values for machinery and equipment and vehicles of the producing and commercial sector are split according to the shares taken from the statistical data for Germany (Statistisches Bundesamt, 2005a). According to these data, the share of vehicles is 22 % of the total machinery and equipment. Of the residual, 85 % is allocated to the commercial and 15 % to the producing sector. Separate estimates were developed for inventories and consumer durables based on literature values. The inventories are estimated to be 25 % and 15 % of the value of machinery and equipment for the commercial and producing sector respectively (cf. Meyer, 2005). Livestock, arable crop, grassland and forest field inventories are valued using standard values of 1000 € per head of livestock unit, 600 € ha^{-1} for cropland, 300 € ha^{-1} for grassland and 1000 € ha^{-1} for forest land. The total value of household consumer durables is calculated using literature values for the net value per residential floor area[i] combined with statistical data on total residential building floor area and residential land use area for each federal state (Statistisches Bundesamt, 2005b). This gives an average of 200 € ha^{-1} for residential building floor area and 21 € ha^{-1} for residential land use.

The value densities are calculated from the area of the corresponding cadastral land use classes and later adjusted to the CORINE land cover classification, which is used to describe the spatial distribution of elements at risk. The CORINE land cover data (EEA, 2002) provide readily available land use information on a 100 m grid derived from satellite remote sensing and comprises 44 classes of land use. Areas smaller than 25 ha and line objects wider than 100 m are generalized. For the Rhine Atlas Method, the land use classes were aggregated into six classes: urban fabric (u), industrial areas (i), traffic areas (t) (airports, harbors and rail yards), forests (f), arable land (a), grassland (g) and others. For the Elbe DSS, the land use is further aggregated into only four flood-risk relevant classes with the following percentage cover in the area at risk considered in the model: land with buildings (urban fabric, industry and traffic) (9.1 %), grassland (59.2 %), arable cropland (10.2 %) and forest (19.5 %). The allocation of the elements at risk to these land use classes is summarized in Table 2.

The value densities are calculated by first dividing the total value of elements at risk by the area of the corresponding cadastral land use classes from the official statistics (Statistisches Bundesamt, 2004). The elements at risk associated with residential housing, commercial, industrial and traffic sectors are divided by the respective cadastral land use area. We consider the aggregate of residential and commercial land use classes as urban fabric. These values are corrected to account for the relative share of these land use classes in the aggregated CORINE land cover class they correspond to. The scale factor is 0.75 for built up areas (urban fabric, industrial and traffic) of which the share of residential land use is roughly 60 % and of commercial and producing sectors roughly 40 %. The scale factor for linear traffic elements as a share of the total area share is 0.04 and 1.00 for all other land uses.

Table 2: Summary of the considered elements at risk, the corresponding aggregated land use classes, value densities and stage damage functions for the five variants of the damage assessment method.

Element at risk	CORINE land cover classes *		Value density [€ mP^{-P2} of CORINE land cover class]					Damage Function [in % as a function of inundation depth h in m]
	(I) DSS	(II–IV)	(I) DSS	(II) IKSR adj.	(III) LTV	(IV) IKSR unadj.	(V) IKSE	
Constructed Assets								
All buildings	U,I,T		73					Y= MIN(90;2h²+2h)
Urban areas		U		186	145	233	104	Y= MIN(90;2h²+2h)
Industrial areas		I		189	207	246	25	Y= MIN(90;2h²+2h)
Traffic areas		T		250	200	250	25	Y= MIN(10h;10)
Traffic infrastructure.	U,I,T ,A,G, F		1.77				0.35	Y= MIN(10h;10)
Traffic infrastructure		A,G		7		7		Y=IF(h>0;1;0)
Machinery and equipment, inventories and consumer durables								
Urban aggregate		U		44	40	55	9.23	Y=MIN(100, 11.4h+12.625)
Household	U,I,T		15					Y=MIN(100,12h+16.25)
Producing	U,I,T	I	3	63	72	82	0.58	Y=MIN(7h+5;100)
Commercial	U,I,T		26					Y= MIN(11h+7.5;100)
Traffic		T		2	2	2	0.58	Y= MIN(10h;10)
Livestock	U,I,T	G	0.4				0.06	Y=MIN(h*50;100)
Vehicles	U,I,T		4.4					Y=IF(X>0;MIN(22.667*(LN(h))+36.345;60);0)
Agriculture		A, G			0.4			Y=IF(h>0;1;0)
Cropland	A		0.06	0.1		0.1	0.1	Y=IF(h>0;50;0)
Grassland	G		0.03	0.05		0.05	0.1	Y=IF(h>0;50;0)
Forest	F		1	1	1	1	0.025	Y=IF(h>0;1;0)

* aggregated CORINE land cover classification: U= urban fabric, I = industrial, T = traffic, A = arable, G = grassland, F = forest

We compared our method with four slightly different methods of adapting the Rhine Atlas Methodology to the conditions of the Elbe Basin. These are: 1) application of a correction factor as described by the Rhine Atlas Method (ICPR, 2001) to adjust value densities from the conditions of Western Germany to the conditions of Eastern Germany[ii] (2) the ad-hoc adaptation of the Rhine Atlas Method as proposed by LTV (2003) for use in the federal state of Sachsen (3) a direct transfer of the values for Western Germany as used in the Rhine Atlas Method (ICPR, 2001) without any adjustment to density values as compared to Western Germany and (4) utilization of the specific value densities proposed in the IKSE Action Plan (2004b) with an adaptation to the CORINE land cover data[iii]. The value densities and their allocation to the different aggregations of CORINE land cover data are summarized in Table 2.

The stage-damage functions to determine the percentage damage as a function of the inundation depth for each value component are also summarized in Table 2. Other factors describing the flood hazard besides inundation depth, such as velocity, inundation duration or contamination with oil or factors influencing the susceptibility of assets, flood proofing and disaster preparedness are not considered here (cf. Merz et al. (2004) and Büchele et al. (2006)). The damage functions are taken from ICPR (2001) and are based on a statistical evaluation of the empirical data on flood damage from the HOWAS database (cf. Merz et al. (2004) for a critical appraisal) of around 2000 damage incidences from flood events in Germany. The damage functions were estimated separately for buildings and their content (machinery and equipment plus inventory and consumer durables) and consist of an evaluation of the damage for inundation depths exceeding the ground floor level of the property, a damage maximum and a choice for the a functional form describing the flood damage for inundation depths in between. Following IPCR (2001) an exponential functional form was chosen for buildings and a linear function for equipment and inventories.

Finally, the total damage (D_{total}) in € per flood event scenario for every 100 x 100 m grid cell is obtained from:

$$D_{total} = \sum_{clc=1}^{n} \sum_{e=1}^{n} \sum_{h=0}^{n} (r(h)/100)_e * A_{h,clc} * V_e * f_{clc} * s_e \tag{3}$$

where $A_{h,clc}$ is the area in m² of land use class clc inundated with a depth of h in m, V_e is the specific value density of the element at risk e in € m^{-2}, r(h) is the relative damage at inundation depth h in %, f_{clc} is the factor to correct the value density of the cadastral database to CORINE land cover in m² per m² and s_e is the share of the value component in the total area of the aggregated land use class in m² per m².

3.4 Flood risk assessment

The damage model was applied to assess the flood damage for individual flood event scenarios, but it was also used to calculate the expected average annual damage as an integrated indicator of flood risk. The change in expected average annual damage is the correct way to estimate the monetary effect of a mitigation measure in a cost-benefit analysis (Penning-Rowsell et al. 2003, NRC 2000).

In the context of the risk based approach flood risk is understood to be the product of the flood hazard (i.e. extreme events and associated probability) and the resulting damage. Ideally, a flood risk analysis should take into account all relevant flooding scenarios, their associated probabilities and possible damage. From these both a risk curve, i.e. the full distribution function of the flood damage, and the annual expectation value of the flood damage can be derived.

In the Elbe DSS the flood risk is calculated from a limited number of flood event scenarios by repeating the damage assessment for a series of flood events with recurrence intervals of 2, 10, 20, 50, 100, 200, 500 and 1000 years at the gauge station of Dresden (Elbe km 56):

$$\langle EAD_{tot} \rangle = \{(1 - P_1)D_1 + \cdots + (P_{n-1} - P_n)D_n + \cdots + P_N D_{\max}\} \tag{4}$$

where $\langle EAD_{tot} \rangle$ is the expected average annual value of the flood damage in €, P_1 is the exceedance probability of the lowest peak discharge causing flood damage with a recurrence interval of 2 years, P_n is the exceedance probability of flood event with a recurrence interval of n years, D_n is the corresponding total flood damage in €, and D_{max} is the maximum flood damage for event N (a 1000-year event). Because this approach requires repeating the calculations for a series of flood events it is necessary that the damage assessment is carried out with a rapid hydraulic model, if it is to be included in a DSS framework.

4 Results

4.5 Comparison of the damage assessment methods

First we compared the estimates for the potential total damage for the five variations of the damage assessment methodology (see section 3.3). For this purpose we used the water levels corresponding to a HQ 200 peak discharge along the trajectory and extrapolate these into the floodplains under the assumption of absence of the dikes. The results of this comparison are shown in Table 3.

Table 3: Comparison of the total damage estimate of five variants of the macro-scale damage assessment method for a water level corresponding to HQ 200.

	(I) DSS	(II) IKSR-adj	(III) LTV	(IV) IKSR unadj.	(V) IKSE
Mio. €	3706	5346	4488	6642	2487
Mio. € per ha*	0.013	0.019	0.016	0.023	0.009

* inundated model area is 2866 km².

The total damage estimate varies by a factor of 2.2 and ranges between 2,480 and 5,340 million €. The distribution of the flood damage over the inundation depth classes follows a similar pattern for all variants. The total flooded area is 286 648 ha. This yields an average damage of 0.01 – 0.02 million € per ha. This estimate is low compared to the estimate of roughly 7,607 million € (IKSE, 2004a) total damage in the German part of the Elbe Basin in the year 2002 which, for an inundated area of approximately 300 km², corresponds to an average damage density of 0.25 million € per ha. A comparison of the contribution of the damage categories to the total damage is presented in Table 4. These are compared to the relative shares reported for Sachsen and Sachsen-Anhalt for 2002 (IKSE, 2004a).

Table 4: Comparison of the contribution of the damage by category as a percentage of the total flood damage for three variants of the macro-scale damage assessment method for a water level corresponding to HQ 200.

	SN 2002*	ST 2002*	(I) DSS	(II) IKSR-adj	(III) LTV
Urban	59	30	82	94	96
Infrastructure (traffic)	14	50	13	3	3
Agriculture and Forestry	1	9	5	3	1
Emergency management	2	1	-	-	-
Infrastructure (flood protection)	24	10	-	-	-

* For comparison: share of total damage in Sachsen (SN) and Sachsen-Anhalt (ST) for the Elbe Flood 2002 (IKSE, 2004a)

The predominance of damage to buildings and their content with shares of total damage between 70 – 95 % is reflected in all model approaches and the observed damage. In comparison to the observed damage, the model results underestimate the damage to traffic infrastructure and do not consider the damage to the flood protection system.

We conclude that the estimate we developed from the available statistical data yields results that are of a similar order of magnitude compared to the ad-hoc approaches of LTV (2003) and the method of adjustment proposed by the ICPR (2004). The uncorrected transfer of data from Western Germany yields higher damage because higher values for all fixed asset categories are assumed and the adjustment proposed by the IKSE yields lower values because lower values are assumed.

4.6 Flood risk in the baseline scenario

Next we present results for the analysis of flood damage along the river trajectory for the baseline scenario without measures for flood events with various recurrence intervals (Figure 3).

The baseline scenario is based on the existing dike heights, the distribution of elements at risk in the inundated areas and the effects on peak water levels of dike overtopping upstream of the specific site. Comparison of the damage for flood events of increasing recurrence intervals along the river trajectory leads to two observations. In river sections where the damage occurs mainly for flood events with a recurrence interval of 20 years or more the flood damage is primarily caused to objects that are not protected by dikes. We find this damage to be high especially in the vicinity of Magdeburg (km 300-350), Dessau (km 200-250) and Bleckede (km 500-550). Furthermore, the model results point to sections, where the risk of damage by a dike breach is high. These are the sections that show pronounced higher damage resulting from flood events with a recurrence interval of more than 100 years. We find that the river sections km 100 -150 and 150-200 above Dessau and 450 – 500 in the vicinity of Wittenberge have the highest flood risk emanating from a dike breach. This is in line with the expectations and the river stretches that were at risk during the 2002 flood event along the River Elbe.

Table 5: Analysis of the effects of variations of the width of dike breaches on model results (for a flood event with recurrence interval of 200 years)

	dike breach width (m)			
	20	50	100	200
number of overtopped dike segments	13	13	13	13
total flooded inner dike area (kmP^{2P})	192	292	359	404
mean inundation depth inner dike areas (m)	1.6	2.0	2.9	4.9
damage in inner dike area (million €)	168	259	438	765
total damage (million €)	352	451	629	956

The volume of water diverted by a dike breach is an important determinant of both the damage at a site, which is determined by the water level in the protected area and the downstream damage which reduces the downstream peak water level. A sensitivity analysis for various widths of dike breaches demonstrates this (Table 5). Whilst the number of dike breaches remains constant, the total flooded inner dike area, the mean inundation depth and the resulting damage increase significantly with larger width.

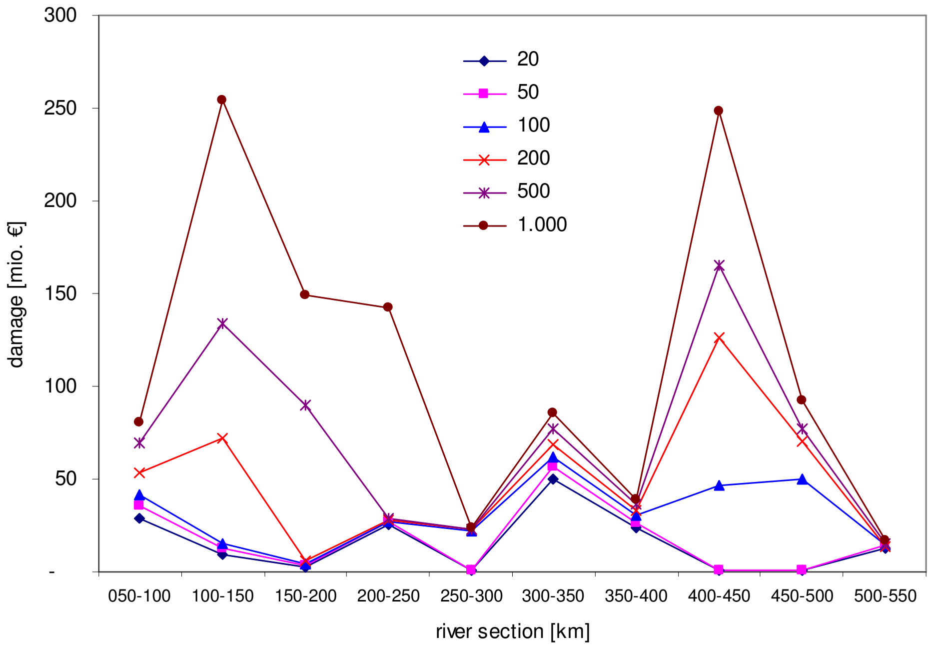

Figure 3: Flood damage by river section for flood events of an increasing recurrence interval.

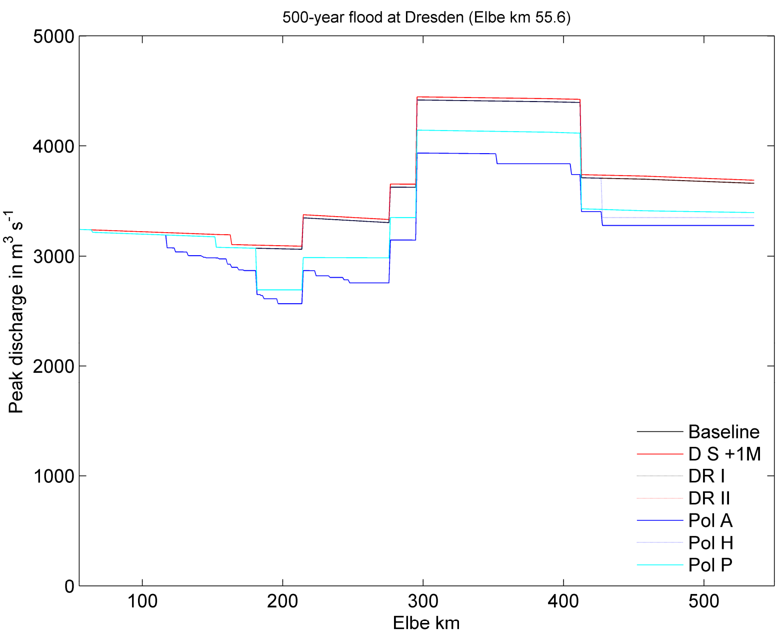

Figure 4: Effects of the assessed mitigation strategies on the peak discharges along the river trajectory for a flood event with a recurrence interval of 500 years. The discharge curve for the DR I and DR II measure are not distinguishable from the baseline scenario.

4.7 Effects of mitigation strategies

Next, we compare the effectiveness of various risk mitigation strategies on the peak discharge, the number of overtopped dike segments, and the average annual damage.

Figure 4 shows the peak discharge along the river trajectory for the baseline scenario and the different strategies (see section 2) for a flood event with a recurrence interval of 500 years. Previous analyses (Helms et al., 2002a) already demonstrated the retention effect of dike relocations on the water levels along the Elbe River to be significantly lower in comparison to the retention effect of polders with controlled operation. Implementing the design standard of a 100-year recurrence interval with an additional freeboard of 1 m for all dikes in the upstream state of Sachsen causes an increase in discharge, because the retention effect of dike breaches in Sachsen is lost.

Table 6: Avoided annual average damage (in million €) of the assessed mitigation strategies.

	Measure					
	D S+1	POL A	POL P	POL H	DR I	DR II
protected by dikes	0.39	5.82	3.94	0.00	1.89	0.00
not protected by dikes	0.00	20.15	9.44	1.36	3.82	0.64
Total	0.39	25.96	13.38	1.36	5.71	0.64

Table 6 gives a summary of effects of the mitigation strategies in terms of the avoided average annual total damage in comparison to the baseline scenario for areas protected by dikes and those not protected by dikes.

The number of overtopped dike segments and the damage for flood events with increasing recurrence intervals for the different management options are shown in Figures 5 and 6 respectively. In Figure 7 the distribution of the avoided annual average damage along the river is shown. In terms of overall performance, the maximum reduction of the Expected Annual flood Damage (EAD) is achieved by the controlled operation of the maximum potential of retention polders (POL A). This option significantly reduces the number of dike overtoppings and associated damage for flood events with higher recurrence intervals (Figures 5 and 6). The reduction of the average annual damage is highest for the sections that have less areas protected by dikes (km 200-250, 300-350, 500-550), but is also observable at sections km 150-200 and 450 -500 that are mainly at risk from overtopping (Figure 7). Singling out the effect of two major polder groups included in POL A, one located more upstream (POL P) and one more downstream (POL H) shows that approximately 50 % of the avoided damage of the POL A measures can be traced back to the effect of the upstream polder group POL P alone (Table 6). The effect of the downstream polder group POL H on the flood risk in the downstream sections km 400-450, 450-500 and 500-550 is similar to that of the upstream polder group POL P.

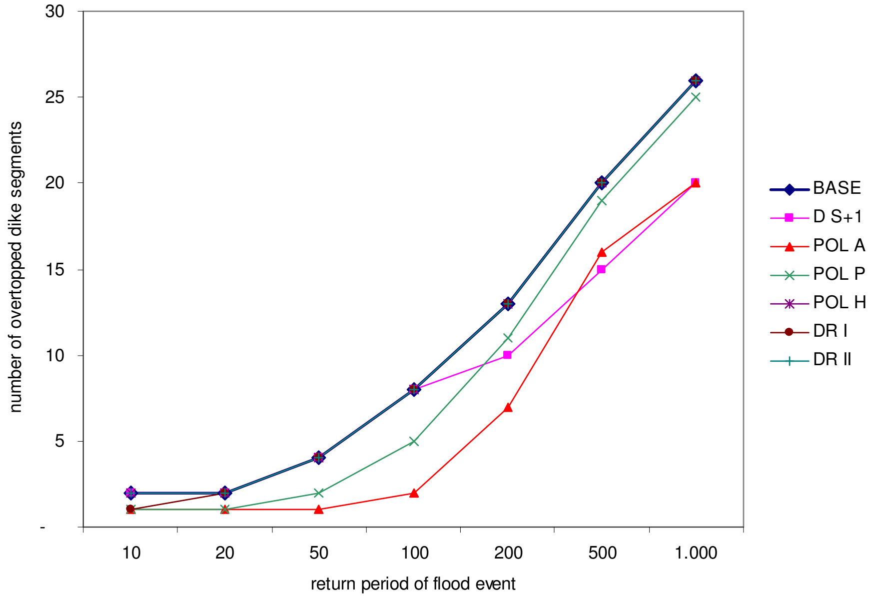

Figure 5: Effect of the assessed mitigation strategies on the number of overtopped dike segments for flood events with increasing recurrence intervals.

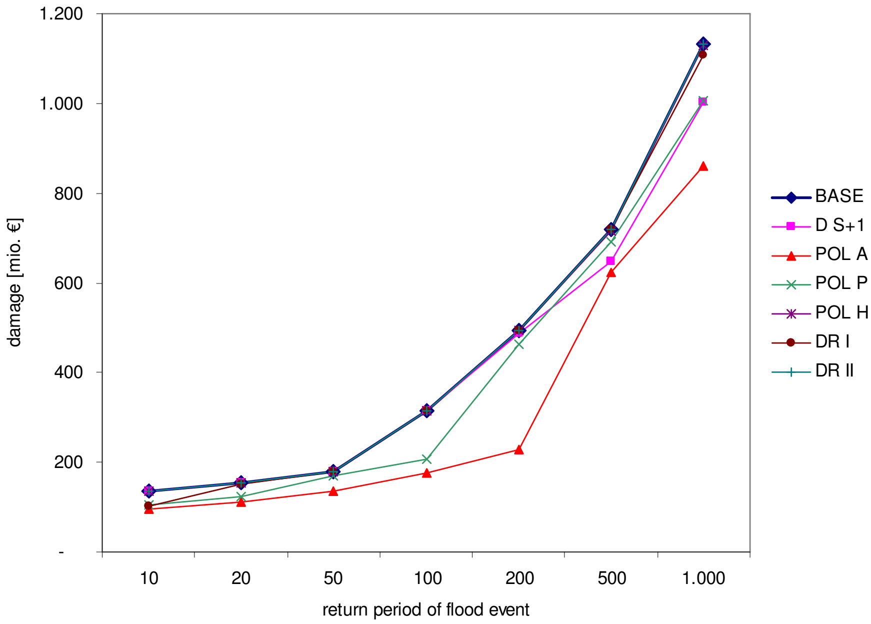

Figure 6: Effect of the assessed mitigation strategies on the flood damage for flood events with increasing recurrence intervals.

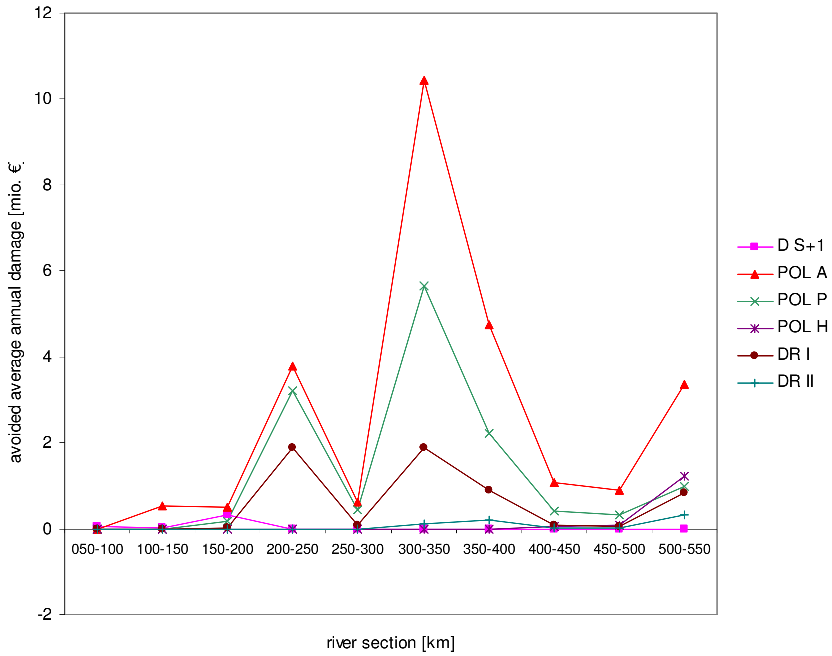

Figure 7: Benefits of the assessed mitigation strategies: distribution of the avoided Expected Annual flood Damage (EAD) along the river trajectory.

Furthermore, the avoided annual average damage is, as expected, lower for the two dike relocation programs with uncontrolled retention. The dike shifting projects DR I (large scale option) and DR II (small scale option) do not reduce the frequency of dike overtopping (Figure 5). The lack of an effect of uncontrolled retention on the peak discharge was already reported for the Elbe river (Helms et al., 2002b) and this study confirms this for even larger scale dike shifting projects such as DR I. However, an effect for flood events with lower recurrence intervals than 100 years can be shown for the dike shifting strategy DR I, which benefits especially those river sections that are to a lesser extent protected by dikes (Figure 8). The average annual damage for the small scale option (DR II) is not significantly reduced compared to the baseline scenario.

Raising the dikes in Sachsen to the protection level of a 100-year return period with an additional freeboard of one meter (D S+1) reduces both the number of dike overtoppings and total damage (Figures 5-6). Even though the number of reduced overtoppings is quite high in comparison to the other strategies, the total effect on avoided damage is not as large, because the areas protected do not contain large areas with high property value densities. A more detailed analysis of the effects along the complete river trajectory reveals that the damage reduction in the upstream sections is reduced at the cost of a slight increase of the flood damage downstream in river section Elbe km 300-350 and 400-500 (Figure 8). This effect is more pronounced, when lower dike breach width with resulting lower unintended retention in the vulnerable middle reaches are assumed.

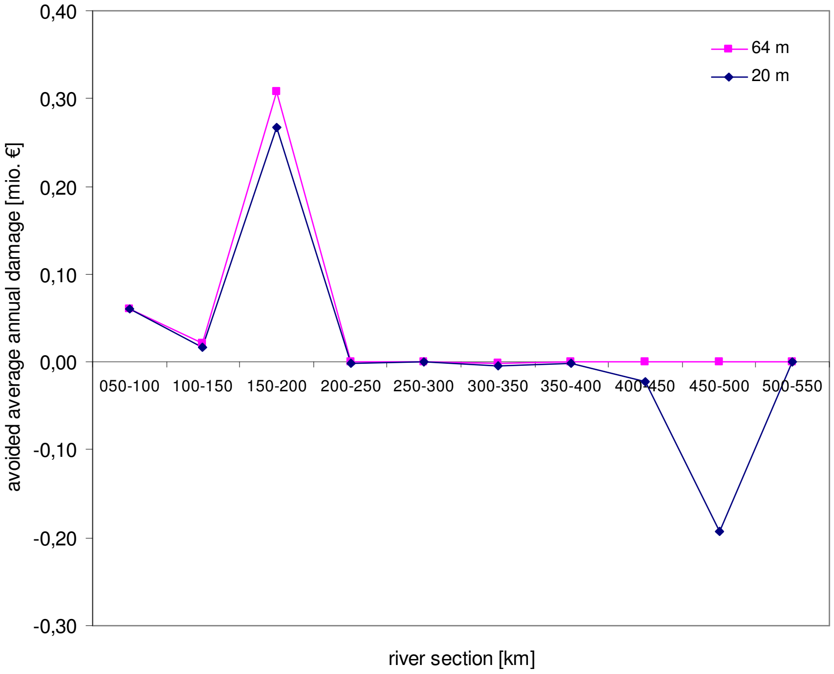

Figure 8: Effects of dike heightening in upstream sections of the river on the average annual damage along the river trajectory assuming average dike breach width of 20 m and 64 m.

5 Conclusions

The need to obtain insight in the downstream effects of flood risk mitigation strategies and necessity to develop flood risk management strategies that are effective at a basin-level scale call for a flood risk assessment methodology that can be applied at the scale of the trajectory of a large river. Although comprehensive 2D hydrodynamic models are very useful for in-depth studies for the planning of structural measures or risk assessment at the scale of individual dike breaches, these models are less suitable for incorporation in an integrated model network, interactive sessions with stakeholders, or repeated use in, for example, a Monte Carlo analysis. The River Elbe served as an example to demonstrate the usefulness of a rapid, GIS-based flood risk assessment methodology. The method allowed for the comparison of the flood risk at the scale of the main river trajectory, which has not been possible for the River Elbe to date. The flexibility of the approach enables rapid assessment of various sets of retention measures of different dimensions and locations. Whereas previous studies assess the water level reductions by various retention measures (Helms et al., 2002a), this work analyzes the effects on the flood risk by taking the spatial distribution of the property at risk along the river into account. Other studies for the River Elbe that incorporate flood risk (e.g.

Förster et al., 2005) have taken a local approach and not yet included the possible interactions of measures along the whole trajectory of the river.

The model was applied to a series of exemplary flood risk mitigation strategies that were developed from ideas discussed in the IKSE action plan (IKSE, 2004b). The downstream effects and the additive effects of combinations of measures on flood risk along the river can be observed clearly. Furthermore, the results demonstrate the increase of the downstream flood risk resulting from unilateral decisions to raise the dikes at upstream locations. As expected, the results underline the potential effectiveness of increased retention along the German river Elbe. The effects of controlled retention at the most upstream possible location and largest possible extent generate the most pronounced reduction of the expected average annual damage. The effect of uncontrolled retention (dike relocations) is significantly lower. However, the model implementation only considers the retention effects and does not consider the effect on the channel roughness of dike relocations. This is a topic for further research.

The results of the flood risk assessment have to be interpreted with caution because several assumptions had to be made. Ideally, a flood risk analysis should take into account all relevant flooding scenarios, the associated probabilities and possible damage. From these both the full distribution function of the flood damage and the annual expectation value of flood damage could be derived, ideally accompanied by uncertainty bounds. Although the role of several uncertainty sources was examined (dike breach width, economic framework of analysis), we did not investigate these systematically in a Monte Carlo analysis. A stochastic approach to modeling the probability of dike breaches as a function of water level and the width of the breach (NRC 2000; Apel et al. 2006) would further enhance the analysis of uncertainty of the inundation process. For recurrence intervals beyond 200 years the artificial flood events are based on an extrapolation of the peak discharge statistics, and the contribution of the tributaries to the discharge was assumed to be proportional to the discharge in the main channel. A conservative estimate was used for the parameters for the inflow that follows a dike breach (Eq. 2). The 1D hydraulic model (Otte-Witte, 2002) has been calibrated for discharges up to a 100-year recurrence interval only, which leads to an underestimation of the peak water levels for higher discharges. This, however, does not affect the general applicability of the risk assessment methodology, and the hydraulic model can easily be improved in this respect. For proper understanding, the hydrological conditions during a flood event should be varied in a Monte Carlo analysis as well to examine the role of uncertainty. The data that were used form another source of uncertainty. The dike overtopping locations and inundation patterns are sensitive to the absolute and relative dike heights, the elevation data for the floodplains and innerdike areas, and the delineation of the potentially flooded inner dike areas. The stage-damage functions are subject to uncertainty that is very difficult to estimate. The development of damage functions has in general received much less scientific attention than the development of models to asses the hydraulic aspects of flood hazards, so that very little is known about the associated uncertainty of the methods (Merz et al. 2004; Büchele et al. 2006; Apel et al. 2004). The macro-scale method in particular is limited by the highly aggregated description of the property elements at risk. Our comparison of different damage assessment variants indicates that the high level of spatial aggregation of our method to two land use classes (urban and agricultural) is not as much a cause for differences in the total damage as the assumptions made with regard to the value density within these classes. It is unclear to what extent results of a more detailed damage model would influence the ranking of results at the scale of analysis presented here. This would be an interesting field for model comparison. Furthermore, the assessment of damage in areas classified as built up land, such as promenades, harbors or buildings that are outside of the areas protected by dikes prove to be a source of error. The problem is aggravated by the inherent spatial inaccuracy of the digital elevation model and the CORINE land cover data. A possible improvement of the model is a separate treatment of protected and unprotected elements at risk.

When reflecting on the assumptions made and data inaccuracies, it has to be kept in mind that the key point that matters in economic assessment of flood risk management options is not so much the absolute magnitude of the flood damage but rather the extent to which a proposed plan will reduce that damage (NRC, 2000). The results demonstrate that a GIS-based rapid flood risk assessment approach can provide information on the relative dimensions and spatial distribution of the flood risk reduction of different combinations of measures along the trajectory of a large river. The information that can be generated with a rapid assessment method helps identifying problems and supporting discussions between and with riparians on flood risk management strategies that promise to be effective from a basin perspective and call for more detailed analysis with sophisticated models.

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[i] Using a net value of 7500 € per unit from IKSE (2004b) for eastern Germany and the statistical data on number of units and total floor area (Statistisches Bundesamt 2005b) yields an estimate of 102 €m^{-2}. Meyer (2005) uses a net value of 350 €m^{-2} for Western Germany, ICPR (2001) 392 €m^{-2} for Western Germany. Adjusted for Eastern Germany using the long term difference in average annual household expenditures of a factor 0.82, this yields values of 287 and 321 €m^{-2}. We use an average value of 200 €m^{-2} of residential building floor area.

[ii] We use the difference in capital intensity of production measured per employee of a factor of 0.77 between the Eastern and Western German federal states and a difference in average annual household expenditures of a factor 0.82 factor to adjust property density values for urban and industrial categories. This procedure further assumes a share of residential land use in the urban fabric of 60 %.

[iii] The specific value density for residential land use is 225 € m^{-2} and 25 €m^{-2} non residential land use. We assume a share of residential and non residential land use of 60% and 40% respectively in urban areas and apply a factor of 0.74 m^2/m^2 to correct from cadastral to CORINE land use. The value for transport infrastructure is calculated from 10 €m^{-2} and a share of transport infrastructure in the total area of 4 %. The value for household consumer durables is calculated from the stated specific value of 7500 € per household unit as described above. The value of inventories is calculated using the proposed 8 % share of total producing and commercial fixed assets. All other data are taken directly from the source.

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PAPER II

IMPACTS OF BOATING TRIP LIMITATIONS ON THE RECREATIONAL VALUE OF THE SPREEWALD WETLAND: A POOLED REVEALED / CONTINGENT BEHAVIOUR APPLICATION OF THE TRAVEL COST METHOD.

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Few studies have been conducted to date on the importance of water availability (in-stream flows, water levels) for demand for a recreation site in Europe. In this paper we combine data on actual trips taken to a site (revealed behaviour) with data on anticipated trips that are stated as a response to hypothetical scenarios constructed for survey respondents (contingent behaviour). We combine these two sources of data in order to assess whether and to what extent the maintenance of minimum in-stream flows for boating matter in demand for trips to a wetland recreation site. The data from the on-site survey is used to estimate an aggregate count data travel cost model. Our findings indicate that variations in navigability significantly affect demand and associated welfare measures.

Keywords: zonal travel cost method, count data model, water based recreation, wetlands, recreational boating, Spree River Basin

1 Introduction

Few studies have been conducted to date on the importance of water availability (in-stream flows, water levels) for demand for a recreation site in Europe. In this paper we combine data on actual trips taken to a site (revealed behaviour) with data on anticipated trips that are stated as a response to hypothetical scenarios constructed for survey respondents (contingent behaviour). We combine these two sources of data in order to assess whether and to what extent the maintenance of minimum in-stream flows for boating matter in demand for trips to a recreation site.

Our application is to a wetland site, the Spreewald in the Federal State of Brandenburg (Germany). The Spreewald is an inland delta within the middle reaches of the Spree River. The river splits into several branches that meander through a wide floodplain, whose landscape is a mixture of forest, grassland and traditional small-scale farming. Currently the wetland has the protected area status of a Biosphere Reserve. Over the centuries, the natural system of rivers has been canalized and regulated through the construction of weirs, both for the purpose of flood control and stabilization of water levels. The main rivers and canals in the Spreewald have the status of navigable waterways. To ensure navigability, a system of weirs and locks is in place. The waterways are mainly used by traditional wooden punts or barges, flat bottomed boats that are manoeuvred by long punt poles. Originally used for transport, they are now used to provide tourists with scenic trips through the wetland landscape. Tourism has a long tradition going back to the 18th century. Because of its vicinity to the metropolitan area of Berlin, the Spreewald has been and still is a popular outing destination for the urban population. In the 1930's the region drew almost 200,000 visitors a year; in 1960's, 500,000 were recorded. Currently, about 2 to 2.2 million visitors visit the area during the season from May to September each year. About 750 punts offer their service and it is estimated that roughly 1 million of the visitors participate in a punt trip. Trip duration ranges from 2 to 8 hours and is most often includes a stopover at a traditional village.

Until recently there was ample supply of surplus water provided from the drainage of opencast lignite coal mines in the headwaters. With the demise of coal mining after the post-socialist transformation, water has become an increasingly contested resource – both within the Spree River Basin as a whole and amongst different water uses within the Spreewald. To stabilise the flow regime of the Spree River, various management options are being considered or have already been partially implemented, including inter-basin water transfers, increase of upstream reservoir capacity or the redistribution of water within the wetland. Because of the high costs associated with many of these options and the outstanding importance of punt trips for the regional tourism economy, this study attempts to provide an estimate of the benefits from maintaining the minimum flows required for boating.

Previous studies have addressed the issue of recreational use value of water quantity changes especially in the US (cf. Eiswerth et al. 2000, Creel and Loomis, 1992, Cooper and Loomis 1993, Ward 1987, Ward et al 1996, Cordell and Bergstrom 1993, Fadali and Shaw 1998). However there are only few studies related to the benefits of maintaining access, in-stream flows or water levels at recreation sites from Europe (Hynes and Hanley 2006; Willis and Garrod 1991, Willis and Garrod 1999). For the headwaters of the Spree River, Lienhoop & Messner (2009) investigate loss of benefits associated with delays of the opening of restored coal mining pits for recreation as a result of insufficient water availability for the scheduled refilling.

Whilst we follow established methodology for travel cost analysis, this paper adds to the limited literature on valuation of recreational resources in Europe. The main features of our approach are that (a) it uses a combination of revealed and stated behaviour data, (b) it uses aggregate data (zonal approach to travel cost method) from an on-site survey and (c) it employs a count data framework with a Poisson log likelihood function.

The remainder of the paper is organised as follows. In the following sections we outline our approach and introduce the econometric specification of the travel cost model. We then present the design of the survey and the creation of explanatory variables. The following section presents the results of the travel cost analysis. Finally, we provide an application of the results to water management in the Spree River Basin.

2 Estimating the benefits of quality changes with single site models

Cost considerations are one of the reasons, as in this study, to rely upon intercept (on-site) surveys to collect information on recreation demand. On-site surveys guarantee that all respondents will be users of the resource. Both individual data and data aggregated by zones can be used to estimate travel cost models from on-site survey data. All travel cost models require significant variation in the number of trips taken. However the low dispersion in the dependant variable when the activity is not repeated frequently within the relevant time frame (individual visitation rates are around once or less than once per annum) may make the estimation of an individual travel cost model impossible.

Arguments favouring individual over zonal approaches are the higher theoretical consistency in modelling individual behaviour and better capacity to address heterogeneity among respondents (Haab and McConnell 2002). However individual models based on on-site sampling come at a cost of both truncation (excluding non users) and endogenous stratification (over sampling those individuals who are more frequent users of a site). As a result the sample is no longer representative of the broader population and failure to correct for on-site sampling will result in biased estimates of recreation demand and welfare measures. Unlike specifications based on individual data, zonal models generally do not need to be corrected for truncation and or endogenous stratification because information on non-participants in aggregate form is readily available from census data. Thus they do not require additional distributional assumptions on the dependant variable and avoid estimators that are highly sensitive to model misspecifications. On the other side, the use of aggregate data will generally not yield parameter estimates that accurately reflect individual behaviour, because they fail to systematically account for underlying individual heterogeneity. This is commonly referred to as aggregation or “error in variables” bias.

In defending zonal approaches, Hellerstein (1995) argues that in practice it is an empirical question as to which source of bias is worse: the error in variable type of bias that may occur in aggregation or the bias from model misspecification that individual models may induce. He concludes that with limited data budgets there can be an advantage in using aggregate data. This is especially true when the average and variance across individuals are small. In these cases aggregate data may contain a high degree of variability that may offset flaws in the aggregate model. Even though zonal models are theoretically less appealing, we opt to estimate an aggregate model because individual visitation rates per annum to our study site are low.

In the standard model of trips to a single site one cannot infer economic values of quality changes, because all individuals face the same quality, so that the model can be used to value access to a site but not the changes in quality of a site. If time series data were available, where the quality variables change over time, then a single site model could provide sufficient information to estimate effects of quality. Increasingly, authors have instead augmented single site models based on actual reported trips with information from contingent behaviour on travel plans under varying hypothetical price and quality scenarios. Whitehead et al. (2008) provide the most recent review on the combination of revealed and stated preference data in valuation of natural resources. Englin and Cameron (1996) and Eiswerth et al. (2000) provide applications within a revealed / contingent behaviour travel cost framework. With such single site cross-sectional data it becomes possible to deduce the effect of quality changes on trip demand and welfare. The main advantage of using contingent behaviour data is that scenarios can be constructed that lie outside of the historical experience of respondents for site qualities. This is especially important for cases where a large non-marginal change in environmental quality is expected in the absence of management interventions, such as unprecedented low flows that result in a loss of navigability.

If the observed and contingent behaviour data are collected through onsite surveys, the contingent data are also truncated and endogenously stratified, because the sampling procedure has excluded individuals who took zero trips in the past and over sampled individuals who took frequent trips. We circumvent these problems as above, by adopting an aggregate model. Using on-site survey data, this is possible because we are measuring a negative quality change only. In this case we do not need to consider increased participation by current non-users that would have to be expected with quality improvements. The use of contingent behaviour in an aggregate model framework has to our knowledge not been applied before. Using an aggregate approach requires a different formulation of the contingent behaviour questions. For an aggregate approach it is necessary to know if the observed trip would have also been taken under the conditions of the hypothetical scenario. While individual approaches need to ask for the expected number of trips per season under a hypothetical scenario, we asked respondents if they would also have taken their current trip under the conditions of the hypothetical scenario. Following Eiswerth et al. (2000) we combine the revealed and contingent behaviour data in a pooled Poisson specification. Pooled data studies stack the two types of data with errors assumed to be independent and identically distributed. Pooled data studies typically constrain the coefficients to be equal across data types but ignore the correlation in behaviour by the same individual across data sources.

3 Econometric specification of the travel cost model

In most modern single site travel cost method applications the model is estimated as a count data model that explicitly accommodate for the count nature of trip demand. As demonstrated by Hellerstein (1991) count data models are well suited to handle aggregate data. With this kind of data, the dependant variable is a non-negative integer and the frequency of zero (non participation) can make up a sizeable fraction of the observations. The theoretical foundation for using count data models in welfare analysis was developed by Hellerstein and Mendelsohn (1993), who show that the integer nature of number of trips taken can be accounted for by modelling the observed number of trips taken as the result of many discrete choices. In the repeated-choice model it is assumed that the individual makes a choice each day of the season about whether to visit or not. Under these assumptions, the distribution of trips will approach a count data distribution, such as the Poisson. The Poisson is a convenient distribution to work with, because it accommodates the presence of zero values and the integer values that trip data take. We use the econometric specification of the aggregate position model proposed by Hellerstein (1999) and Haab and McConnell (2002).

The aggregate model can be understood as a macro function, which assumes that the behaviour of individuals in a zone is identical. The estimation process then retrieves the demand parameters for a representative individual. The demand for trips to the site by individual i in zone j is x_{ij}. z_i is a vector of individual characteristics that such as travel and time cost to the site, costs of substitutes, and other variable that enter the individual demand function. In principle, it is desired to estimate a model of demand for an individual x_{ij}, but only aggregate data on number of trips X_j originating from zone j are available. Aggregate demand in zone j is then given by $X_j = N_j x_j^*$, where N_j is the number of potential users, often measured as population and x_j^* is the demand by a representative individual.

This assumes that all individuals are independent and identically distributed within the aggregate, so that $z_i = z^*$. However if they are not identical to the extent that the measure of z_j (for example the zonal mean), is not identical to z_j^*, an aggregation bias will be introduced.

We assume that the number of trips taken by an individual is generated by a Poisson process. The Poisson has the useful property that the sum of weight independent Poisson variables is also Poisson distributed, so that when the xj is distributed with parameter λ_j, $N_j x_j$ is also distributed Poisson with $N_j \lambda_j$ (cf. Hellerstein 1993). Thus the Poisson probability function for the aggregate trips in zone j becomes:

$$\Pr(X_j) = \exp(N_j \lambda_j)(N_j \lambda_j)^{X_j} / X_j! \tag{1}$$

with the expectation and variance both equal to λ. Parameter λ_j is the expected number of trips for the representative individual in zone j and is assumed to be a function of the variables specified in the demand model.

To allow for exogenous variables such as price to affect demand and to guarantee non negative number of trips, λ is modelled in exponential (or semi-log) form:

$$\lambda_j = \exp(\beta_{tc} tc_j + \beta_q q + \beta_z z_j) \tag{2}$$

where tc is the travel cost to the site from zone j, q is a vector of site specific (quality) attributes, z is a vector of demand shift variables and ß a vector of coefficients to be estimated.

Substituting (2) into (1) then gives an expression for the probability of observing X_j trips from zone j as a function of tc and z. The parameters in (2) are estimated by maximum likelihood. The log-likelihood function is:

$$\ln L = \sum_{j=1}^{J} N_j \exp(\beta_{tc} tc_j + \beta_q q + \beta_z z_j) + X_j (\ln(N_j) + (\beta_{tc} tc_j + \beta_q q + \beta_z z_j)) - \ln(X_j!) \tag{3}$$

The likelihood maximisation process recovers the parameters of the representative individual's demand for trips. However, in our application the sum of the aggregate count of the trips from a zone, X_j is not a census of all visitors but is a representative sample from the true (and independently known) total number of visits to the site per year, X^*. In order to retrieve the correct parameters for the annual individual demand for access to the site, N_j is the total population, N_j^*, scaled by the sample rate:

$$N_j = N_j^* \cdot (\sum_{j=1}^{j} X_j) / X^* \tag{4}$$

The expected consumer surplus (CS) per trip t is:

$$E(CS)/t = -1/\beta_{tc} \tag{5}$$

Aggregation is achieved by multiplying the consumer surplus per trip by the expected total annual number of trips, which is the product of the expected representative individuals demand for trips λ_j in zone j and the zones population N_j^* summed across zones:

$$E(CS) = \sum_{j=i}^{j} (N_j^* \exp(\beta_{tc} tc_j + \beta_q q + \beta_z z_j)) \cdot (-1/\beta_{tc}) \tag{6}$$

The change in total consumer surplus for a change in site quality is then calculated by evaluating (6) for different values of q.

4 Data sources

4.1 Survey

An onsite face-to-face interview survey was carried out on 8 days between 8.6.- 22.7.2002 at four major boating sites in the Spreewald (Burg, Lübben, Lübbenau, Schlepzig). Attention was given to drawing a random sample of tourists passing the interview station on their way to or from the boats, by approaching the next passing visitor after completion of a previous interview.

The design of the survey took place before the water resources management model was completed so that no detailed information on the expected effects of climate change and water management was available. Historical experience suggested that as a first effect of low water levels the number of persons per boats would be reduced. However as there is an overcapacity of punts, there would be no negative restrictions entailed for visitors. Increased water scarcity would then lead to problems in passing locks, so that trips would be limited to the regulated sections between locks with relative constant water levels. This would entail a limitation on the maximum duration of trips that could be taken by roughly a half, from currently 6-8 hours to then 2-3 hours. In a final stage, water levels would in general be too low for navigation, resulting in a complete closure of the waterway. We therefore decided to measure the effect of restrictions of maximal possible trip duration on the demand for trips to the Spreewald wetland. Whilst trip limitations may hypothetically also be the result of protected area management action or a deterioration of the canal system, we intend to use the reaction to measure the implicit impact of low flows on the recreational value of the Spreewald wetland.

After asking respondents details about their current trip, it was explained that a closure or limitation could be the outcome of reduced water availability. Respondents where then asked (a) if they would still have taken their current trip to the Spreewald if they would have known that they could not take a boot trip because of a total closure of the waterway and (b) if the would still have taken their current trip, if there was only the possibility of taking a limited boat trip, that would foreclose being able to see some of the major scenic attractions accessible with a long punt trip. Thus each respondent contributed three observations to the model: the actual trip, and two contingent trips under the hypothetical scenarios.

Table 1: Summary statistics for visitor sample

Variable	N	Mean	min	max	SD
Distance	483	228	1	765	189
Days	483	2.9	1	21	3.5
Days = 1	483	51			
Days = 2-3	483	27			
CB-p	483	0.58			
CB-t	483	0.55			
Persons	320	3.3	1	12	1.2
Duration	483	3.7	1	8	1.4
Trips	483	1.27	0.165	15	3.17

Definition of items	
Distance	Distance from home county in km
Days	Duration of trip in days
Days = 1	Percent of total trips that have a duration of 1 day
Days = 2-3	Percent of total trips that have a duration of 2-3 days
CB-p	Contingent behaviour: Percent of respondents who state the trip would also have been taken with limitation of maximum punt trip duration
CB-t	Contingent behaviour: Percent of respondents who state the trip would also have been taken with total closure of waterways
Persons	Number of persons per car
Duration	Duration of punt trip in hours
Trips	Average number of trips to the Spreewald per person per year (trips taken within the last three years)

The survey database contained 483 valid observations on participants in punt trips. Summary statistics of some of the response variables are reported in Table 1. The average number of trips per respondent to the Spreewald is 1.2 trips per year. The average duration of the punt trip is 3.9 hours. The mean distance to home is 230 km and on average the visit to the Spreewald is part of a 3 day holiday. Roughly 50 % of the respondents visit the Spreewald as day trippers, another 27 % as part of a weekend trip of max. 3 days / 2 nights. The remaining 22 % visit the Spreewald as part of a multipurpose holiday of more than 3 days. Roughly half of the respondents stated that they would not have visited the Spreewald, if they could not have taken a punt trip.

4.2 Variables

The dependant variable of the aggregate model is the count of visitors to the Spreewald per zone. The decision unit analysed in the travel cost model is in general trips of equal length. For trips of varying duration an extended model that can accommodate for choice of on-site time would have to be developed. We extract two datasets: counts of single day trips only and counts of short trips of one to a maximum of three days only. This second data set accommodates for typical weekend trips that combine two days of travelling with one day on site.

Of the 483 observations in the database, we extracted two sets of observations of punt trips originating from together 359 different local administrative units ("Gemeinden") for aggregation. The first data set contains only the 250 observations of visitors taking a day trip to the Spreewald, the second dataset expands this to encompass 383 observations of day and short trips. Because the basic travel model is only valid for single purpose trips of roughly the same length, we opt to exclude longer multipurpose trips (22 % of sample). These would require a separate analysis, also because assigning a consistent cost to the boating portion of such a trip is difficult. There are only observations from 0,025 % of the 13912 local administrative units that constitute the market area, so we subsequently aggregate all local data to means or sums for zones of the same distance, travel time and population density (cf. Lovett et al. 1997; English and Bowker 1996). We use classes of distance zones of 50 km and travel time of 30 min. The population density is used as an indicator to differentiate between rural and urban regions and we use classes of 0-500, 500-1000 and > 1000 inhabitants per km². This yields a set of 85 zones, 28 of which have an observed visitation rate of zero visits. Because the dataset contains three pooled data sets of observed and contingent counts of visitation, the final dataset used for model estimation contains 340 observations.

The independent variables included in the model consist of travel cost to the site (TC), travel cost to a substitute site (TCSUB), a site quality variable that describe the maximum possible trip duration (Q), a dummy variable to identify the contingent behaviour responses (CB) and additional socio-demographic variables of the zonal population.

The site quality is measured as the maximum possible trip duration as a fraction of current maximum possible trip duration, so that the resulting three point scale takes the values 1, 0.5 and 0. The distance and travel time to the Spreewald was calculated using a Geographical Information System. We use least cost path analysis across a cost surface of maximum travelling speed to identify the shortest duration path from every local administration unit. The cost surface was created from the road network assuming travelling speeds of 120 km/h for highways, 80 km/h for major roads and 50 km/h for all other roads. We also calculated the shortest distance and travel time to a substitute site. The treatment of substitution in site based recreation models remains unresolved and the choice of a substitutive variable remains arbitrary. Some authors, such as Hellerstein (1993), use an imputed substitution price based on the site nearest to the destination having similar characteristics. Other authors use a substitutive site that is the closest similar alternative site to an individual's origin. Others use indexes of accessibility of recreational resources within a certain time band from the origin (cf. Lovett et al. 1997, English and Bowker, 1996). Punting is an activity that is unique to the Spreewald. It is considered to be one of the outstanding tourist and day trip attractions in Germany. We therefore use a map of such sites to calculate the shortest distance path from each local administrative unit to the nearest substitute site.

We further include variables that describe per capita income and its distribution. We use average per capita pre tax income (INCOME) available on the district level as a proxy of income and the percentage of the total population that is liable for income tax (TAXPAYER) and the unemployment rate (UNEMPLOYMENT) as a indicator variables for the income distribution.

Table 2: Variables for the aggregate travel cost model

Variable	Definition
TC	Travel cost from zone j to site in €
TCSUB	Travel cost from zone j to substitute site in €
INCOME	Per capita pretax income in zone j in 1000 €
UNEMP LOY	Unemployment rate in zone j in %
TAXPAY ER	Share of population in zone j that is taxpayer
AGE60	Share of population in zone j that is older than 60 years
AGE20	Share of population in zone j that is younger than 60 years
URBAN	Dummy: 1= zone includes a major urban centre
POPDEN SITY	Population density in population per km²
N*	Population of zone j
Q	Site Quality: maximum possible trip duration as a fraction of current maximum possible trip duration with three levels (1, 0..5, 0).
CB	Dummy: 1 = source of data is contingent behaviour

Other factors that have been found to affect participation in recreational activity are age and urban/rural zones of origin. Two age related variables were included: the percentage of population older than 60 years (AGE>60) and younger than 20 years (AGE<20). We also calculated the population density of each local administrative unit. Units with a population density below 500 inhabitants per km² were classified as rural (RURAL) and units with a density above 1000 inhabitants per km² were classified as urban (URBAN).

All socio-economic variables are calculated using the local and district official statistics (StaBu 2003a, 2003b).

In recreation demand studies the price or cost of access to a recreational site is generally defined as a total of travel cost, entry charges and opportunity costs of time. The travel cost is typically estimated by multiplying the total distance travelled by a standardized average cost. The treatment of opportunity cost of time in travel cost methods has however been a consistent source of debate, because the consumer surplus estimate is highly sensitive to the treatment of time (Haab and McConnell 2002).

We calculate travel cost as the combination of distance costs, on-site pecuniary costs and distance travel time costs, using a third of the calculated hourly wage rate to capture the opportunity cost of time. We use the common procedure to divide annual reported pretax income by the average annual working time to obtain the wage rate. Specifically,

$$tc_{ij} = dc_{ij} + oc_{ij} + dtc_{ij} \tag{7}$$

with

$$dc_{ij} = (2d_{ij} * kc) / g \text{ , } oc_{ij} = bt * bc \text{ , } dtc_{ij} = 0.33 / 2080 * w_i * tt_{ij} \tag{8}$$

were dc are distance costs based on vehicle operating costs of 0,25 €/km based on the variable cost estimate for a middle class car, d is distance in km from residence site to the Spreewald and g is the average number of persons travelling in the same vehicle, oc are the on site costs, bt is the average duration of a boat trip estimated as 3,7 hours, bc are the average punt trip cost of 2,5 €/h, dtc are the distance time costs, w is the annual wage rate that is estimated using the average per capita pretax income of zone i, tt is the travel time from zone j taking various travelling speeds for different roads categories into account, 0.33 is the factor for downward adjustment of the wage rate, 2080 is the number of working hours per year.

5 Estimation of the travel cost model

The models are estimated using maximum likelihood technique and the count data poisson log linear model implemented in the SPSS 15 generalized linear model procedure. Although the Poisson provides consistent estimates of the coefficients, it provides biased estimates of the covariance matrix if the true trip distribution is not exactly Poisson. Since hypothesis testing requires accurate estimation of the covariance matrix, a robust estimator is used. The robust parameter estimate covariance provides a consistent estimate even when the specification of the variance function of the response is incorrect.

Table 3: Regression results using pooled aggregate count data model

Model (a)	D/RB/Q=1			S/RB/ Q=1			D/CB/ Q=1,0			S/CB/ Q=1,0			D/CB/ Q=1,0.5,0			S/CB/ Q=1,0.5,0		
	Coef.	sig	SE	Coef.	sig	SE	Coef.	Sig	SE	Coef.	sig	SE	Coef.	Sig	SE	Coef.	Sig	SE
(constant)	-15.887		14.850	-3.952		8.848	-17.000		11.728	-3.579		6.772	-16.283		9.931	-5.108		5.575
Q			.			.	0.616	**	0.090	0.601	**	0.087	0.583	**	0.090	0.576	**	0.087
TC	-0.051	**	0.009	-0.029	**	0.005	-0.050	**	0.008	-0.029	**	0.004	-0.053	**	0.007	-0.031	**	0.003
TCSUB	0.052		0.031	0.086	**	0.018	0.057	*	0.025	0.090	**	0.014	0.049	*	0.021	0.084	**	0.011
INCOME	0.339	**	0.120	0.216	*	0.093	0.319	**	0.099	0.203	*	0.072	0.314	**	0.081	0.209	**	0.058
UNEMPLOY	8.374		16.367	-7.258		8.549	9.388		13.033	-7.257		6.622	8.722		11.054	-6.379		5.499
TAXPAYER	-16.040		23.963	-29.831	*	12.456	-11.069		19.188	-27.609	*	10.311	-12.007		16.187	-25.555	**	8.087
AGE60	34.758	*	15.012	26.882	*	12.319	32.011	*	11.770	23.086	*	9.476	31.842	**	9.041	25.373	**	7.345
AGE20	8.092		17.328	1.069		14.554	6.934		14.380	-1.494		11.466	7.858		11.258	-0.357		9.096
URBAN	0.717		0.499	0.693		0.434	0.556		0.380	0.594		0.322	0.582		0.316	0.601	*	0.269
POPDENSITY	-0.061	**	0.021	-0.052	*	0.020	-0.047	**	0.016	-0.044	*	0.016	-0.051	**	0.014	-0.045	**	0.013
Count	250			383			500			766			750			1149		
LL	-102			-137			-183			-252			-274			-378		
E(CS)/t	19.8			33.9			19.8			34.5			18.7			32.1		
Ø Trip duration	1			1.59			1			1.59			1			1.59		
$t_{q=0,5} / t_{q=1}$							0.73			0.74			0.75			0.75		
$t_{q=0} / t_{q=1}$							0.54			0.55			0.56			0.56		

Dependant variable: number of trips (count) per capita per year from zone j.

a. Model: D = daytrips only, S = daytrips and short trips up to three days combined, RB = only revealed behaviour, CB = revealed and contingent behaviour combined (with behavioural response to total closure (Q = 1 and 0) or total and partial closure (Q=1, 0.5 and 0) entered into the analysis).

b. ** significant at 0,01 level, * significant at 0,05 level

Table 3 contains results from several model specifications. Results are presented for day trips only and day and short trips combined. For each of the two datasets, a revealed behaviour only, a pooled model including the total closure scenario and a pooled model including both contingent scenarios is reported. We find that all coefficients are of the expected sign and a stable pattern of significant variables. The estimated coefficients on the site travel cost (TC) are negative and significant at the 1 % level. The estimated coefficient on the site quality indicator, the maximum possible trip duration (Q) is also positive and significant at the 1 % level.

This indicates that, all else equal, limitations on the possible punt trips are associated with a lower level of visitation to the Spreewald wetland site. The travel cost to substitute sites (TCSUB), the average per capita income (INCOME) and the percentage of population aged over 60 (AGE60) have positive and in most cases significant effects on visitation rate. Positive, but not significant effects are also shown for the unemployment rate (UNEMPLOY), percentage of population aged under 20 (AGE20) and zone of origin that are urban centres (URBAN). The population density (POPDENSITY) was found to have a negative and significant effect, however this does not conform to the expectation that visitors would more likely originate from more densely populated areas.

We report two variants of the pooled model, because in contrast to the total closure scenario, the limited trip duration scenario is less well defined and more open to subjective interpretation. The results show that the coefficients on all variables are similar. This is also true in comparison to the revealed behaviour only model. As a further test of convergent validity, Table 4 compares the results of the pooled model that includes a variable to denote data derived from contingent behaviour (CB) with the restricted specification (also reported in Table 3) that does not include the variable. Following Eiswerth et al. (2000) we use this specification to test if the source of the data has a significant influence on other parameters of interest in the model. The estimated coefficient on the CB variable is not statistically significant. Comparison of the coefficient shows that the inclusion of the variable has no influence on the majority of the coefficients of the other variables, with the exception of the quality variable. We use a Wald Test to test the hypothesis that the CB variable has a significant effect on the model, which we find not to be the case. We conclude that convergent validity holds for our data set and that contingent and revealed behaviour both lead to a similar welfare estimate.

Table 3 also shows the estimated consumer surplus per trip. The consumer surplus for day trips is estimated to be around 19 € per trip and for trips from one to three days duration around 33 € per trip. Note that the estimates are per trip and not per day and that the average duration of the combined short trip data set is 1.6 days. This result indicates that there is a trade-off between travel distance and choice of on-site time. Table 3 further presents' estimates of the effect of limitations in maximum possible punt trip duration on the number of trips taken to the Spreewald. The estimates are very similar across all specifications with a total closure of the resource leading to a reduction of the total number of trips taken by ca 45 % and a partial closure to a reduction of 16 - 25 %.

Table 4: Test of effect of CB on estimated demand and consumer surplus.

Model	S/CB/ Q=1,0.5,0				S/CB/ Q=1,0.5,0			
	Coef.		Wald $\chi2$	P > ∣$\chi2$∣	Coef.		Wald $\chi2$	P > ∣$\chi2$∣
(constant)	-5.108		0.839	.36	-5.163		0.883	.347
Q	.576	**	44.031	.000	.842	**	24.342	.000
TC	-.031	**	97.969	.000	-.031	**	99.441	.000
TCSUBS	.084	**	56.485	.000	.084	**	57.017	.000
INCOME	.209	**	12.929	.000	.209	**	13.148	.000
UNEMPLOY	-6.379		1.346	.246	-6.379		1.395	.238
TAXPAYER	-25.555	**	9.987	.002	-25.555	**	10.201	.001
AGE60	25.373	**	11.935	.001	25.373	**	12.044	.001
AGE20	-.357		.002	.969	-.357		.002	.968
URBAN	.601	*	4.990	.025	.601	*	5.204	.023
POPDENSITY	-.045	**	12.374	.000	-.045	**	13.119	.000
CB					-.241		2.822	.093
							.	
LL	-378				-376			
E(CS)/t	32.092				32.091			
$t_{Q=0,5} / t_{Q=1}$	0.75				0.84			
$t_{Q=0} / t_{Q=1}$	0.56				0.55			
Wald Chi Square					3.043			
Df					1			
Sig.					.081			

* Dependant variable: number of trips (count) per capita per year from zone j.

** The Wald chi-square tests the effect of CB. This test is based on the linearly independent pair wise comparisons among the estimated marginal means.

6 Policy application: estimating welfare effects of water management options

We use the results of the travel cost model to assess water management options for the Spree River and to value the maintenance of minimum flows for boating. For this purpose we use the results of a water resources simulation model to analyse effects of changes in water availability and water management within the Spree Basin. This modelling framework was developed to address long term water resource planning under conditions of water scarcity (Koch et al. 2006, Dietrich et al. 2007). The simulation of the natural discharge and climate parameters follows a stochastic approach, whereby these input parameters are provided by a stochastic simulation of runoff conditions. The model describes the flow of the river system as a node link network. Key model elements are the balance profiles set along the watercourses, the catchments, water users, reservoirs and wetlands. The model operates on a monthly time step and balances water demand and water supply according to the physical capacities of the river and water management infrastructure and the water management rules in place. Water is allocated not only on a first come first serve basis, but according to the rank or priority accorded to a water use within the system of water use rights. The simulation of the water balance is carried out with 100 stochastically generated realisations of the climatic and discharge conditions over a period of 50 years. This procedure enables the estimation of water supply reliabilities by means of statistical analysis after completion of the simulation.

The effect of reduced flows on recreational boating in the Spreewald that is considered in the model is a disruption of the longer boating routes that require passing certain locks, because the water level in the locks is not sufficient to allow boats to pass. Monthly water levels in the locks are calculated on the basis of the wetland groundwater levels. A lock is considered impassable, when the water level is lower than the required minimal depth of approximately 0,3 m. Boating is considered to be disrupted, if the number of affected locks is higher than the threshold value. The share of the season that boating in the whole Spreewald is disrupted by low flows (M_{low}) is then calculated as follows:

$$M_{low} = \left(\sum_m if\left(\sum_l if(nD_l - dWL_{m,l} <= mD;1;0) > TH;1;0\right) \right) \Big/ S \tag{9}$$

where subscripts l denote locks, nD is the nominal depth of water in the lock under target conditions in meters, dWL is the difference of actual water level in a month m from target water level in meters, mD is the minimum required depth in meters, m is a month of the season, TH is an evaluative threshold for the number of locks above which boating is considered to be disrupted and S is the number of month in the season.

We then proceed to calculate the total annual recreational value (CS) of punting trips in the Spreewald as a function of annual low flow probabilities (M_{low}) as follows:

$$CS = (M_{low} \cdot X^* \cdot t_{Q=0,5} / t_{Q=1} \cdot CS/t) + (1 - M_{low}) \cdot X^* \cdot CS/t) \tag{10}$$

where CS/t is the consumer surplus per person per trip in €, M_{low} is the share of the season with trip limitations, $t_{Q=0,5} / t_{Q=1}$ is the share of visitors that would not have come to the Spreewald at low flows and X* is the estimated total number of participants in punt trips per year.

A rough approximation of the marginal benefit from water allocation to maintain minimum flows in the Spreewald can be derived from plotting the annual loss of recreational benefit against the summer water deficit of the wetland. The loss describes the difference between total recreational benefits without restrictions to navigability and the total recreational benefit with restrictions. The water deficit describes the difference between the water delivered to the wetland and the water demanded to maintain the target groundwater levels. Figure 1 depicts the loss as a function of the cumulative summer water deficit for current target water levels, using the value of 19,8 € for CS/t. Losses to recreation begin to appear at summer water deficits of ca. 25 hm³. Beyond this level, the marginal recreational loss is roughly € 0.08 per additional m³ of summer water deficit.

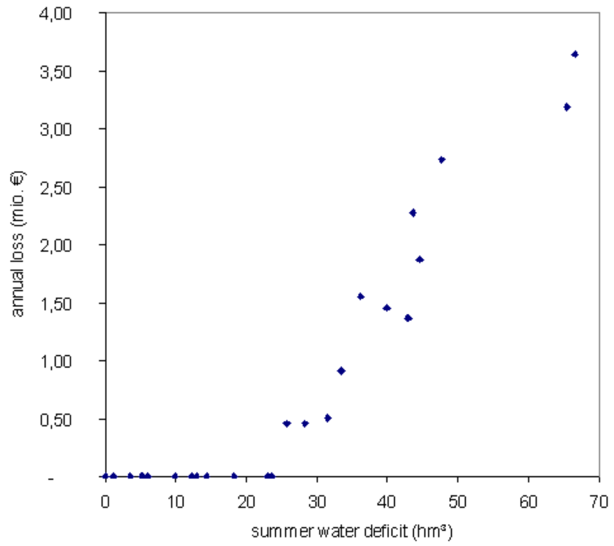

Figure 1: Estimated loss in annual recreational benefit plotted against the wetland summer water deficit.

With regard to water policy, the results indicate that there is a considerable benefit generated by the maintenance of minimum flows sufficient for boating. Augmenting low flows can be considered to generate a net benefit, if the additional water can be provided at a lower cost than the incremental benefit. Cost to be considered are costs for reservoir development, inter-basin transfers or the opportunity costs of reallocating water from other uses. Complementary benefits from maintaining minimum flows beyond recreation have also to be considered, that derive from other water level dependant wetland functions and additional downstream benefits from the non-consumed flows. Currently, the Federal States of Brandenburg and Saxony have an agreement under which inflows to Brandenburg at the Spreewald are supplemented from reservoirs in Saxony by up to 20 million m³ of water per year. Saxony has increased its reservoir capacity for this purpose, and in return Brandenburg agreed on a payment of half a million euros per year. This is equivalent to an implicit cost for the additional water of € 0.025 /m³. The additional water is intended to augment flows throughout the whole course of the Lower Spree including the flows in the metropolitan area of Berlin. From an economic perspective, however, the payment could be justified by the benefits created for the Spreewald alone. The results indicate that there is a favourable economic payoff to this long-term public investment for augmentation of low flows.

7 Conclusion

This study has shown, that a pooled revealed / contingent behaviour data single-site application of the travel cost method can be used to not only estimate the recreational benefit from access to an outstanding wetland site but also to estimate indirectly the effects of water availability on the demand for such a site. Few studies have been conducted on the importance of water availability for demand for recreation sites in Europe, because water scarcity has only been an issue in a few of the central European river basins to date. Measures of economic values associated with maintaining minimum in-stream flows are becoming more important in justifying investments in water infrastructure and assessing water management options. Despite the importance of recreational use of in-stream flows in industrial economies such as Germany, the associated benefits are not included in cost-benefit analysis because of a lack of value estimates. Our findings indicate that limitations in navigability significantly affect demand and associated welfare measures for an important wetland recreational site.

There is ample scope for improvements in the applied methodology. For complex relationships between in-stream flows and recreational use patterns, as is the case for the Spreewald, it is not always strait forward to create quality indicators that are amenable to both a plausible description of hypothetical scenarios for survey respondents and that can be meaningfully be incorporated into a hydrological modelling framework. The aggregate travel cost model can also be further improved. This pertains to the unrealistic assumption of zero variance for the representative individual that can be assumed to be a source of aggregation bias. An improvement of the aggregate model is possible, if intra-zonal covariance data of the variables describing the representative individual can be obtained and incorporated into model (cf. Hellerstein 1995, Moeltner 2003). More detailed attention in future surveys can also be given to the issue of the appropriate specification of the value of travel time (cf. Hynes et al. 2009). Finally, finding a way to accommodate for the substantial share of visitors who take a boat trip as part of a multipurpose visit to the Spreewald region at large is a further issue that has not been addressed in this paper.

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Manuscript

PAPER III

ACCOUNTING FOR SCOPE AND DISTANCE DECAY IN META-FUNCTIONAL BENEFIT TRANSFER: AN APPLICATION TO THE WILLINGNESS TO PAY FOR WETLAND CONSERVATION PROGRAMMES IN EUROPE

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This study presents a meta-analytical benefit transfer function for willingness to pay (WTP) estimates for wetland habitat and biodiversity conservation programmes generated with stated preference methods. In particular, it explores two key factors that need to be addressed in the benefit transfer process: scope and distance decay effects. As the number of empirical studies on wetland valuation has risen continuously, we extract a smaller, but more homogeneous dataset from the literature. In this way we are able to single out a theoretically consistent meta-function that estimates mean WTP for wetland habitat conservation as a function of scope, distance decay, and income. We find WTP to increase with the size of the conservation programmes (scope effect) and to decrease with increasing spatial extent of the sampled population (distance decay effect). These findings enhance the potential to use the results of the meta-regression for benefit transfer. Whether the remaining margin of error is considered large or too large depends on the use of the results. The results further indicate that methodological choices have significant influence on the mean value estimate.

Keywords: benefit transfer, wetlands, meta-analysis, valuation of ecosystem services

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1 Introduction

Wetlands provide a multitude of ecosystem goods and services that in turn give rise to private and social benefits (Turner et al. 2008). A wide range of methods is available for the valuation of the different market and non-market benefits provided by wetlands. In the context of wetland valuation, stated preference methods (contingent valuation and choice experiments) have primarily been used to assess the WTP of the population for the conservation of wetland habitats and biodiversity. The benefits captured by WTP for wetland conservation are potentially considerable; however are they are also extremely difficult to measure. Where the implementation of primary surveys is not possible or not justified because of budget and time constraints, benefit transfer is an alternative strategy to generate benefit estimates for policy and project appraisals. There are two broad approaches to benefit transfer: unit value transfer and value function transfer (Navrud and Ready 2007). Unit value transfer is the transfer of a single point estimate from a single study or the transfer of a measure of central tendency from several studies (average value) derived by meta-analysis. Value function transfer is the use of a value function derived from a single study or derived from a meta-analysis of several studies (a meta-regression function). The function is used to estimate the benefit for a transfer site by plugging in the appropriate parameters for this site, such as the socioeconomic characteristics of the population or the scope of the measure. The meta-analytical approach has advantages for benefit transfer compared to a transfer based on a single study, because it can control for effects of study specific methodological choices and because it provides a more rigorous measure of central tendency that is based on a broader empirical basis (Lindhjem and Navrud 2008).

In this paper, we present a meta-analytic approach to benefit transfer. With regard to environmental valuation, meta-analysis has primarily been used for systematic quantitative summary of evidence on methodological issues across empirical valuation studies. Meta-analysis is increasingly also used to develop benefit transfer functions (cf. Bergstrom and Taylor 2006, and Nelson and Kennedy 2009, for theoretical and methodological overviews). Five meta-analyses of wetland valuation studies already exist. However, only two of these studies are based on an analysis of stated preference studies (Brouwer et al. 1999 and Moeltner and Woodward 2009). The meta-analyses by Woodward and Wui (2001), Brander et al. (2006) and Ghermandi et al. (2008) pool value estimates from various types of wetland uses (e.g. agricultural, recreational, non-use) based on a range of valuation methods and different value concepts, such as cost based, producer surplus and consumer surplus based values[1]. While these studies reveal the substantial economic value associated with different types of wetland use, they are not suited for benefit transfer in a cost-benefit analytical framework, because they violate a key criterion for satisfactory benefit transfer: welfare measure consistency (Nelson and Kennedy 2009).

[1] The evaluated primary studies seek to elicit the value of a wide range of ecosystem service benefits including both market and non-market goods such as fuel wood, recreation, biodiversity conservation, water quality control, flood protection. In order to generate a common metric for the effect size across different methods, these meta-analyses convert the WTP estimates per capita from stated preference studies to an aggregate WTP per unit wetland area using additional assumptions from the primary study about the correct market size (population) over which to aggregate and the area of the wetland site by which to divide the value.

This paper differs from the previous wetland meta-analyses[2] in that it focuses exclusively on stated preference studies that elicit WTP for a narrowly defined good: wetland habitat conservation programmes. Two key concerns in transferring such WTP estimates across policy sites and populations of varying scale are scope and distance decay effects. Provided that wetland conservation is a normal good, economic theory would require willingness to pay for an environmental amenity to be an increasing function of the scope of the amenity (cf. Smith and Osborne 1996). The expectations are that for two wetland conservation programmes − one of which nests the other − the more ambitious or larger programme should be valued higher than the smaller one and that the marginal WTP would be decreasing. Measurements of the degree of sensitivity to scope are commonly proposed as a test for the reliability of contingent valuation studies. However, as Amiran and Hagen, (2010), argue, the limited sensitivity to scope often observed .in stated preference studies cannot be ruled out based on standard consumer theory and a failure to satisfy existing scope tests should not be used as a general argument for rejecting contingent valuation studies. Rather, the results of scope tests should be considered more carefully. There is an additional explanation from economic theory for observed scope insensitivity. Rollins and Lyke (1998) argue that this may be due to the diminishing marginal utility and that many studies are made over the range of a good where marginal utility approaches zero.

It is also likely that the mean WTP for wetland conservation programmes falls, the further the sampled individuals live from a site because individual's preferences are related to the intensity of use made of the valued resource. This phenomenon is referred to as distance decay (cf. Pate and Loomis 1997, Bateman et al. 2006). Whereas distance decay of use values has a clear explanation, as the share of users generally declines with increasing distance to a site because of the increase in travel costs, the explanation for non-use values is less clear. Non-use values may in principle be held by anyone irrespective of the distance from an individual's home to the relevant site. Distance decay may be related to the quality or uniqueness of a site. Unique sites such as national parks are likely to be well known and are likely to have fewer substitutes. Distance decay effects could be expected to be low (Pate and Loomis 1997). Less well known sites may have a more local or regional importance, so that WTP will decrease with distance or may even fall abruptly beyond a political border (Hanley et al. 2003). There may also be good reasons that WTP may be lower in the local area, since there may be opposition to conservation in local communities dependent on primary resource sectors. The conservation issue is often more important to urban populations (Lindhjem 2007). Distance decay effects are also a related to the availability of substitutes, which will normally increase with distance from a site. The higher the number of alternatives, the lower the WTP for a particular site will be.

[2] With the exception of Moeltner and Woodward (2009) in their study for the USA.

The two spatial concepts are highly relevant for benefit transfer across scales, because ignoring decreasing marginal WTP related to the scope of programmes and the spatial extent of the population over which benefits are to be aggregated is likely to overestimate the average and total WTP estimate (Bateman et al. 2006). Appropriate adjustments to account for spatial effects are therefore required.

The issue of scope has been addressed in several meta-analyses of stated preference studies. Scope effects have essentially been analysed along one of two dimensions: quality and quantity (cf. Lindhjem 2007). Scope effects have been analysed for the quality dimension where it is possible to translate the description of the good used in the primary studies into a common metric to describe the quality change. Examples are analyses of the WTP for air quality improvements measured as relative visibility (Smith and Osborne 1996) or for improvements in water quality measured along a water quality ladder (van Houtven et al. 2007). While these studies have been able to demonstrate scope effects regarding the quality dimension, they do not explicitly address the quantity dimension. In contrast, analyses of WTP for changes in land management have focused on the quantity dimension. It can be argued that this is because the spatial extent of land management programmes can readily be described in terms of area, while there is no unifying metric available to describe the impact of land management on the multiple quality dimensions that are impacted (such as biodiversity, scenic beauty, recreational opportunities). For example, Lindhjem (2007) or Barrio and Louriero (2010) investigate scope effects in WTP for forest management programmes (forest protection or multiple use forestry) based on the area covered by the programmes. However no significant scope effects were found. The meta-analysis of wetland studies by Brouwer et al. (1999) uses a size classification ranging from "very large" to "very small" but also finds no significant effects. The further three wetland meta-analyses by Woodward and Wui (2001), Brander et al. (2006) and Ghermandi et al. (2008) include the area of the wetland site as an explanatory variable and find that value estimates, transformed to a per unit area measure, decrease with wetland size. However, as these studies pool value estimates from stated preference studies with those from other methods, they cannot draw conclusions regarding the scope sensitivity of the included stated preference value estimates.

Distance decay has not been explicitly investigated in any meta-analyses of stated preference studies. A possible approach that has been used to capture relevant aspects is a classification whether the WTP for a good was elicited from a local, regional or national population (Lindhjem 2007). Other studies address the issue by explicitly basing their analysis on primary studies that sample populations of comparable spatial extension, for example US States (Moeltner and Woodward 2009).

With regards to the general public's WTP for wetland habitat and biodiversity conservation programmes, this paper argues that scope and market area are among the key moderator variables that need to be accounted for in a benefit transfer function. The approach outlined in this paper is based on the assumption that using a more homogeneous dataset that is based on a relatively homogeneous good definition and a single value concept but with intra- and inter study variation in both the economically important variables and the data treatment- and methodological choices, will allow to retrieve the parameters of such a benefit transfer function (cf. van Houtven et al. 2007 and Nelson and Kennedy 2009). This kind of approach is often foreclosed because there are too few primary valuation studies to allow a statistically sound analysis based on OLS regression approaches (cf. Moeltner and Woodward 2009)[3]. However, the number of empirical studies on wetland valuation has risen continuously and we are able to include many recent studies from Europe that are not included in any of the previously reported wetland meta-analyses.

In the next section we outline the process of extracting value estimates from the literature. We then discuss key determinants of WTP and how these are described as variables for the analysis. The following sections describe the meta-regression procedure and results. We then provide an application and discussion of the results for benefit transfer, before concluding with some general remarks on possible further improvements.

2 Data selection for meta-analysis

This meta-analysis is motivated by the need for a value transfer function to be applied in the cost-benefit assessment of management options for wetlands under agricultural or forestry use in the context of land use policy assessment in the European Union. We therefore restrict our study from the outset to studies from this political entity[4] and to semi-terrestrial wetland ecosystems. Semi terrestrial wetlands are floodplain, fen, bog and salt marsh wetlands. These are, in an unmanaged condition, only seasonally or intermittently inundated and are often converted to high intensity grassland or forestry land use. Restoration efforts are therefore typically accompanied by land use conflicts. We therefore exclude studies referring to permanently inundated and aquatic wetland types such as lakes, ponds, reed beds, lagoons, streams or rivers..

We searched the literature for primary studies that use stated preference methods (contingent valuation and choice experiments) to elicit WTP for wetland habitat and

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[3] An alternative approach to "make the most of small samples" is to use Bayesian modelling approaches (Moeltner and Woodward 2009).

[4] Comparable regional restrictions are also chosen in the cited meta-analyses by Moeltner and Woodward (2009) and van Houtven et al. (2007) (studies from US only) or Lindhjem (2007) (Scandinavia only).

biodiversity conservation programmes. It follows that we exclude studies that primarily seek to elicit WTP for other wetland related benefits, for example for flood protection or water quality improvements. We only include studies that estimate WTP of the general population (and therefore the WTP of both users and non-users of the wetland) and that are expressed as a value per time period (monthly, annual, single payment)5.

Based on these criteria, we conducted a literature search to identify as many studies as possible, using several databases and bibliographies on environmental valuation studies as well as internet search[6]. We included peer reviewed publications, books as well as articles from the grey literature such as reports, working papers and theses. We identified 26 studies reported in 33 publications published between 1990 and 2009 that included value estimates that conformed to our criteria[7]. Table 1 provides a summary of the studies.

[5] Specifically, we don’t include studies that elicit WTP per visit from site visitors because this can not be converted into an annual WTP of the general public.

[6] Based on these criteria, we conducted an extensive literature search to identify as many primary empirical studies for the European Union as possible, using several databases and bibliographies on environmental valuation studies (cf. Melichar 2004 for Czech, Hungary and Poland, Turner et al. 2008 for UK & Ireland & Netherlands, Sundberg & Söderquist 2004 and Navrud 2007 for Scandinavia, Meyerhoff and Elsasser 2007 for Germany, Austria and Switzerland). We identified further studies from scanning the reference lists of the aquired studies and systematic internet search based on the key words “economic valuation”, “stated preference”, “choice experiment” or “contingent valuation” and “wetland”, “floodplain”, “riverine”, “bog”, “fen”, “peat”, “marsh”, “estuary”, “coastal”.

7 Of the 26 studies in our sample, 7 have been included in Brouwer et al. (1999), 2 in Woodward and Wui (2001) and 1 in Gehrmandi et al. (2007).

Table 1: Overview of studies

Year	Publication (s)	Country	Wetland name	Type (1)	Valuation scenario / question	Measure (2)	Area of measure (ha)	Sampled population	VE (3)	Method (4)
1993	Kosz, 1996	A	Austrian Danube Floodplains	R	WTP for protection of floodplain forest from hydropower development by establishment of a National Park	P	11500 / 9700 / 2500	Austria	3	CV / C / F / O
1991	Bateman et al. 2000, Bateman & Langford 1997	UK	Norfolk Broads	P	WTP for protection of fen and marsh wetland from coastal erosion by maintenance of coastal protection infrastructure	P	16000	Zones of 20 40 60 km radius	3	CV / C / M / O
2001	Meyerhoff 2004, Meyerhoff & Dehnhardt 2007, Meyerhoff 2003.	D	Elbe Floodplains	R	WTP (a) to introduce low intensity use of floodplain grasslands by agri-environmental schemes and (b) to restore parts of the floodplains by dike relocation	R	55000	(a) Elbe, (b) Weser (c) Rhine River Basin	4	CV / C / F /O
1990	Hanley & Craig 1991	UK	Flow Country	P	WTP to prevent further afforestation of upland blanket peatland	P	40000	Scotland	2	CV / V / M / O
1994	Brouwer & Slangen 1998	NL	Alblasserwaar d	P	WTP for scheme to compensate farmers for continuation of low intensity land use system with high water levels on fen wetlands	P	15660	Zuid-Holland	2	CV / V / M /O
1994	Willis et al. 1996	UK	Pevensey Levels	P	WTP for scheme to compensate farmers and landowners for continuation of low intensity land use system with high water levels	P	3500	(a) Pevensey Levels (b) 60 km radius	2	CV / V / M / D
1997	Amigues et al. 2002	F	Garonne Floodplain	R	WTP for a project to restore floodplain wetland currently under agricultural land use.	R	10000	Toulouse Metropolitan Area	3	CV / C / F / O+D
1998	Degenhardt et al. 1998	D	Wangen Valley Floodplains	R	WTP for compensating farmers for introduction of low intensity land	R	650/1100	Wangen County	2	CV / C / F /O

					use system and partial restoration of valley floodplain grassland					
1998	Kenyon & Nevin 2001	UK	Ettrick Valley Floodplain	R	WTP for project to restore floodplain forests	P	90	Scottish Border Council	2	CV / V / F / O
1995	Ciszewska 1997, cited in Zylic 2000	PL	Biebzra River floodplain and fen peatlands	P+R	WTP towards protection and development of Biebzra National Park	P	59000	Poland	1	CV / V / F / D
2000	Ehrlich & Habicht 2001	EST	Estonian coastal and floodplain meadows	R+C	WTP for maintenance and restoration of semi-natural coastal and floodplain grassland habitats	P	30 000	Estonia	1	CV / V / M / D
2005	Lundhede et al. 2005	DK	Store Amose	P	WTP for restoration of fen wetlands currently under agricultural land use	R	270 / 1750	(a) Vestsjaelland (b) Denmark	4	CE / C / M
2007	Birol et al 2008	PL	Bobrek Floodplain	R	WTP to improve the biodiversity conservation status of an urban floodplain	R	60	City of Sosnowiec	1	CE / C / F
1993	Brouwer & Spaninks 1999, Spaninks 1993.	NL	Sneek peat meadows	P	WTP for scheme to compensate farmers for not intensifying the land use of fen wetlands	P	500	City of Sneek	2	CV / V / M / O
2005	Beaumais et al. 2007	F	Seine Estuary	C	WTP for conservation programme to protect Seine Estuary from conversion to industrial land use	P	14 000	Seine Estuary	2	CV /C / M / O
2001	Fucsko et al. (in prep.)	HU	Hungarian Danube Floodplain	R	WTP to prevent river regulation with hydropower station	P	-	-	-	CV /
2007	Turner et al. 2008	UK	Culm Grassland	P	WTP for a 10 % expansion of protected area through restoration of adjacent land	R	4400	Devon & Cornwall	1	CV / V / F / O
ca. 2005	Birol et al. (in prep)	UK	Tamar Valley Floodplain	R	WTP for restoration of river floodplain	R	-	-	-	CE
1986	Willis 1990, Willis 1989	UK	Derwent Ings, Skipwith	P+R	WTP towards protection of a floodplain meadow and a bog nature reserve	P	240 / 783	Yorkshire	2	CV /V / F / O

			Common							
2007	Luisetti et al. 2008	UK	Blackwater Estuary	C	WTP for an increase in salt marsh area and improved protection level for estuarine birds.	R	82 / 817 / 2404	32 mile radius	2	CE / C / F
2004	Birol & Cox 2007	UK	Severn Estuary	C	WTP for restoration of estuarine saltmarshes	R	500 / 7500	Worchestershire	2	CE / C / F
1992	Garrod et al. 1994, Willis et al. 1995	UK	Somerset Levels and Moors	P	WTP for a scheme to compensate farmers for maintaining current extensive use of wet grasslands	P	15250	(a) local (b) UK	7	CV / C / F / O+D
1998	Meyerhoff 2004b, Meyerhoff 2004, Hartje et al. 2002, Meyerhoff 2003	D	Waddensea Coast	C	WTP to maintain the availability of coastal habitats (salt marshes) for example by dike relocations to compensate for sea level rise	P	18500	Germany	3	CV /?/ F / O
2000	Hanusch et al 2000	D	Bavarian Danube Floodplain	R	WTP to prevent river regulation and associated negative effects on the floodplains of the Bavarian Danube River.	P	8600	(a) local (b) Bavaria (c) Germany	4	CV / C / F+M / O
2001	Danchev, 2002	BG	Bulgarian Danube River Islands	R	WTP to protect natural floodplain forest from transformation to poplar plantation.	P	11000	Bulgaria	1	CV /
2008	Lifran et al., 2008	F	Marais des Beaux	P	WTP for restoration of Marais des Beaux wetlands.	P	400 / 1000	10 km radius	2	CE/C/F
2007	Bliem & Getzner, 2008	A	Danube Floodplains	R	WTP for restoration (connection) of 50 or 90 % of modified floodplain wetlands in the Donau Auen National Park	R	3488 / 6277	Vienna and Lower Austria	2	CV/C/M /O
2007	Szerenyi et al. 2008	HU	Altal-er Floodplains	R	WTP for restoration (connection) of 50 or 90 % of modified floodplain wetlands in the Altal-er area	R	5000/9000	Komárom-Esztergom County	2	CVM/C/ O/F
2007	Vadineanu et al. 2008	R	Inner Danube Delta	R	WTP for restoration of 50 or 90 % of the former wetlands of the Braila Isalnds into their original state	R	42000/75600	Braila Island Complex	2	CVM/C/ O/F

2007	Liekens et al. 2008	NL	Saeftinghe / Hedwigiepolder	C	WTP for restoration of a coastal marsh area by dike relocation.	R	300	South Zeeland	1	CE/CF
2009	Meyerhoff et al., 2010	D	Floodplains and fen peatlands in Germany	R/P	WTP for implementation of national biodiversity strategy – specifically for large scale restoration of floodplain and peatlands in Germany.	R	500000 (floodplain)/ 1400000 (peat)	Germany	2	CE/V/M

(1) Wetland type: R = riverine, P = peatland, C = coastal

(2) Measure type: R = restore or improve wetland habitat availability, P = protect or maintain / avoid loss of habitat availability.

(3) VE: maximum number of value estimates extracted for the meta-analysis

(4) Method: CE = Choice Experiment, CV = Contingent valuation, C / V= coercive and voluntary payment vehicle, F/M = face to face and mail survey, O/D = open and dichotomous choice format.

3 Determinants of WTP for wetland habitat and biodiversity conservation

The basic assumption for the benefit transfer function is that the underlying variables of the implicit bid function for wetland conservation are assumed to be derivable from some unknown indirect utility function for a change in wetland quality (cf. Bergstrom and Taylor 2006). However, additional explanatory variables related to the elicitation methods are also introduced into the model. As the conceptual foundation for constructing the benefit transfer function we define the basic meta-model for mean WTP as:

$$WTP = f(Q, W, P, M) \tag{1}$$

The determinants of WTP for wetland habitat conservation can be grouped by the characteristics of the wetland conservation programme in terms of the scope of the quantity/quality of proposed change (Q), characteristics of the wetland site (W), the characteristics of the study population including income, distance to the site and availability of substitute wetland sites (P). In addition, the valuation method (M) that was used to estimate WTP is expected to have an effect on the resulting value estimate.

3.1 Effect size variable

The dependant variable is the summary statistic or effect size of the primary study. In this case it is the estimated mean of annual household WTP for a proposed wetland habitat and biodiversity conservation programme. In order to make estimates from different years compatible and to account for relative differences in purchasing power, the mean WTP estimate in national currency is converted to 2005 Euro 27 using a purchasing power parity exchange rate to convert national currency to Euro and subsequently applying a consumer price index to convert to 2005 purchasing power[8]. Primary studies variously elicit individual or household WTP. Many studies that elicit individual WTP then proceed to aggregate WTP by household, so that in the end WTP is most often interpreted as household WTP. We include a dummy variable (HH) to denote studies that elicit household as opposed to individual WTP. While some studies elicit single or one-off payments, most studies elicit annual payments. We include a dummy variable (SINGLE) to differentiate these values from other values. Strong insensitivity to

[8] We use data from Penn World Tables 7.0 for PPP exchange rates (Purchasing Power Parity over GDP in national currency units per USD), we then convert USD to EURO27 for the European Union of 27 based on PPP exchange rates from OECD Statistics and then adjust to the year 2005 based on the CPI for the European Union of 27 provided by EUROSTAT (HICP (2005=100) - Annual Data).

payment schedule indicates the inability of respondents to differentiate between a series of payments and a lump sum payment. Single, one-off payments are expected to be higher (Kim and Haab 2009).

3.2 Description of the valued good: wetland habitat and biodiversity conservation

The primary studies typically ask respondents' WTP for wetland conservation and restoration programmes or programmes to introduce more environmentally sensitive land management practices. The valuation scenarios used in the primary studies are summarized in Table 1. The values from these studies can be interpreted as the WTP to obtain a positive change or prevent a negative change in at least one element in the attribute vector describing the wetland ecosystem in an individual's utility function, for example the level of biodiversity, the scenic beauty or degree of human modification. In terms of the total economic value concept (Turner et al.2008), the valued benefits primarily encompass non-consumptive use benefits and non-use benefits arising from maintaining the diversity of habitats and organisms. Benefits from non-consumptive uses are related to amenity and recreational activities such as enjoying the scenery. Non-use benefits arise for instance from preserving natural heritage for future generations independent of any personal use of a site. Non-use values can not easily be separated from non-consumptive use values, because improvements of the wetland habitat and biodiversity quality may also improve the recreational use value of those members of the population that use the wetland areas for recreational purposes. A substantial share of the WTP can be assumed to be for non-use benefits, because the sampled general populace includes both users and non-users of the wetland sites.

Variables describing conservation programme characteristics (Q in Eq. 1) try to capture variation in the valued good. The wetland conservation programmes can in principle be characterised according to two dimensions: the proposed change in wetland quality and the quantity of wetland area that is to be affected by the quality change. It is difficult to create a common metric to describe the wetland quality changes implicit in the conservation programmes[9]. However, we include a dichotomous variable to describe whether a conservation programme describes a quality gain or an avoided quality loss (RESTORE). The framing of the valuation questions either imply the avoidance of a loss of the current wetland habitat quality or the restoration of the habitat quality of degraded wetland sites. The a-priori expectation is that WTP for maintaining the current quality status will be higher than for restoration of additional wetland sites (Bateman et

[9] We tried various approaches based on degree of human modification, but the sample is too small to create the required variation of the variable. Typical information provided in the studies relates to the description of measures such as agricultural extensification, restoration of sites, or the change in the abundance of certain key species or habitats.

al. 2006). A possible explanation for this would be loss aversion, describing people's tendency to strongly prefer avoiding losses to acquiring gains. A higher WTP could also reflect scope and substitution effects. In this case an increase in the availability of wetland habitats should be valued lower than maintaining the current availability as consequence of a decreasing marginal WTP for wetland habitat availability.

The key variable of interest for this study describes the area for which a change of wetland quality is proposed (WETAREA). In most primary studies, the proposed programmes only cover a share of the total area of a wetland site. Although great care was taken in compiling the area variable, the process inevitably required subjective judgments and involved uncertainty. A description of the area covered by the conservation programmes is generally part of the information provided to respondents, either explicitly in terms of area units or implicitly in form of a map or a description. Some primary studies specifically elicit WTP for programmes of varying area. Other studies, however, are not very precise about the scale of the programme and just specify a targeted wetland site. In these cases information on the area of programme was generated by augmenting descriptive information reported in the studies with additional information on the wetland sites available from supplementary sources. If wetland conservation is a normal economic good, the expectation is that average WTP and that marginal WTP decreases with increasing size of the programme area.

Besides the characteristics of the programme, WTP for wetland conservation may be influenced by characteristics of the wetland sites (S in Eq. 1). Based mainly on the land form and soil type, we distinguish three semi-terrestrial wetland ecosystem types: coastal and estuarine salt marshes (COASTAL), riverine floodplains (RIVERINE) and peat bogs and peat fens (PEAT)[10].

3.3 Characteristics of the sample population

To capture distance decay effects implicit in the mean WTP estimates reported by primary studies we define variables to describe the spatial extent of the sampled population. Lacking data on the mean distances from the respondent’s homes to the wetland site from the primary studies, we define a proxy to reflect the increase in average distance to a site with increasingly larger sample units. Assuming a uniformly distributed population, an increase in sample area will be correlated with an increase in average distance of the respondents to the site. We estimate the size of the sample area (POPAREA). As outlined above, the a priori expectation is that average WTP estimates are lower for studies that sample the population of a larger area than of a smaller area.

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[10] This is based on the classification of wetlands adopted by the contracting parties to the Convention for Wetlands of International Importance (RAMSAR).

Although great care was taken in extracting the sample area from primary studies, the process inevitably required subjective judgments. We define the sample area as the area that is defined by the maximum distance to the outer limit of the sample area[11]. Because in many studies only the sampled administrative unit is reported, we used internet search to complement data on the area. If only the radius of concentric sampling zones was given, we calculated the corresponding area taking the rough geometry of the sample area into account.

WTP will be also be affected by the availability of substitute wetland sites for the sample population. Again, if wetland conservation is a normal economic good, WTP will decrease with an increasing availability of wetland habitats. Following the approach of Gehrmandi et al. (2008), we include an index of substitute wetland site availability (WETINDEX) that describes the area of wetland in a 100 km radius from the wetland site[12]. However, there may be other substitutes to wetlands that may be important, such as other types of habitats or conservation areas. Also, not all wetlands may have the same quality or uniqueness and may therefore be unequal substitutes. The index can therefore only be a rough proxy.

An individual's ability to pay is captured by individual or household income. Income is expected to have a positive impact on WTP and the finding of a positive income effect provides evidence of theoretical consistency of the value estimates. However data on the income levels of the sample populations is not consistently reported in the primary studies, so we use mean annual individual income for the study year converted to USD_{2005} purchasing power parities as a proxy (INCOME). Using the national values as a proxy assumes that income levels of the sample population do not deviate substantially from average income level in the country. This approach was used for example by van Houtven et al. 2007. Most meta-analysis do not include any variables to describe income levels (Lindhjem 2007, Richardson and Loomis 2009, Brouwer et al. 1999). Gehrrmandi et al. (2008) and Brander et al. (2006) use GDP per capita as a proxy for income levels.

Finally, based on the exploratory data analysis that indicates relative high WTP estimates despite low income levels, we include a variable to differentiate eastern European transformation economies from the western European economies. Possible reasons for a higher WTP, besides of course a stronger preference for conservation, may also be

[11] In particular, we extract the actually sampled area, not the subsequent aggregation area. Some studies sample an administrative unit that does not include the site, for example by surveying a near administrative unit. According to our definition, the sample area is then adjusted to include the site.

[12] This index is proposed by Gehrmandi et al. (2007) and is based on a GIS analysis of the wetland area in a 100 km^2 radius of the site based on the digital wetlands map of the Global Lakes and Wetlands Database (Lehner and Döll 2004). It is measures in ha per $100km^2$.

systematic cultural differences in the perception of stated preference instruments or possibly also for the earlier studies in methodological weaknesses.

3.4 Valuation methodology and publication bias

Additional variables were created to describe potentially influential differences in the methodology (M in Eq. 1). First we distinguish between values estimates generated by contingent valuation and choice experiments. A dummy variable (CE) is used to identify values estimates generated by choice experiments. Choice experiments have been found to result in higher WTP compared to contingent valuation (cf. Richardson and Loomis 2009). We further include a dummy variable to characterise open ended contingent valuation elicitation formats (OPEN) in contrast to all other, that is dichotomous choice (yes or no to a given bid amount), iterative bidding (yes or no to a sequence of bid amounts) questions and choice experiments. Previous analysis indicates that the open ended elicitation format yield significantly lower WTP than dichotomous choice, because respondents do not have the incentive or the required routine to find their true maximum bid (van Houtven et al. 2007, Lindhjem 2007). Besides the elicitation format, the payment vehicle can be expected to have a marked effect on the WTP estimate. We use a dummy variable (TAX) to differentiate between coercive payment vehicles (general tax, local tax, earmark tax, water bill) from voluntary payments (donation, payment to a trust fund). Coercive payment vehicles have been found to generate both higher (Brouwer et al 1999) and lower (Lindhjem 2007) WTP estimates compared to voluntary ones. The credibility and incentive compatibility of the payment vehicle may be the decisive factor. Coercive payment vehicles such as tax may be considered more credible and with lower associated uncertainty because they constitute a plausible implementation mechanism with a guaranteed broad, albeit coercive, participation of all members of society. This would induce a higher WTP. On the other side, a rejection of coercive instruments may lead to a relatively lower WTP. WTP estimates may also be influenced by the survey format. A dummy variable (MAIL) was formulated to differentiate between mail surveys on the one side and in person interviews (mainly face-to-face interviews) on the other. It is difficult to formulate a priori expectation on the direction of the effect. Mail surveys are often expected to result in lower WTP, because of social desirability effects in the interview situation (Noonan 2003). However, in-person interviews may allow for more carefully considered responses, which may lead to a lower WTP (van Houtven et al. 2007). If the mail surveys have low response rates, a higher self-selection of respondents with high WTP may also be a reason for relatively higher mean WTP (Lindhjem 2007).

The way primary studies deal with protest and outlier bids also greatly influences the mean WTP estimate from a sample. There are two principal approaches to treat protest responses. The most common procedure is to exclude protest responses (Meyerhoff and Liebe 2006). The definition used to exclude protest answers from the sample varies between studies. While some studies exclude all zero responses, the most common practice is to exclude a part of the zero bids identified as protest by a set of debriefing questions. The alternative procedure is to retain all protest answers in the sample with a WTP of zero. Both procedures can strongly affect welfare estimates. We use a dummy variable (NONPROTEST) to denote those estimates that exclude protest values. We do not include WTP estimates that are only based on positive bids in our sample, even though many studies report these values. Truncation or trim removes an upper (and or lower) percentile of WTP responses from the sample so as to adjust for possible outlier or strategic responses that have a large leverage effect on the estimated mean WTP. We include a dummy variable (TRIM) to denote those WTP estimates that are based on a sample that is trimmed by an upper percentile.

Finally, there may also be a publication bias on WTP estimates because the published empirical literature is not an unbiased sample of the empirical evidence (Rosenberger and Johnston 2009).

Criteria for selection what research is published (for example significance of estimated coefficients, conformance of variables and sign with expectations) and the tendency not to publish valuation studies from policy applications that do not aim to generate methodological innovations may bias the WTP estimate in the published literature. For benefit transfer, it is therefore recommended to make use of the full range of evidence that also includes unpublished valuation studies (Rosenberger and Stanley 2006). We formulate a dummy variable (PUB) that denotes studies that have been published in peer-reviewed journals to control for publication bias. However, several of the studies in discussion paper format can be expected to be published at a later stage, so that this classification may not be very precise.

4 Data summary

Table 2 describes the variables used in the meta-analysis and provides summary statistics. Common spatial sampling strategies found in the primary studies are to survey either the local resident's that live in the vicinity of a site or the general population of the larger administrative (or national) region within which a wetland site is located. Many studies provide split samples for both such nested sample populations. In this case we extract from the primary study a WTP estimate for the small sample population (typically local population) and the large sample population (typically national or regional population). Some wetland studies attempt to explicitly examine distance decay effects beyond this simple two-fold stratification by stratified sampling using zones of increasing distance. For distance-stratified samples, we include the WTP estimate of the local stratum and the overall sample in the database.

Table 2: Meta-analysis variable description and summary statistics.

Variable	Description	Mean *	SE*
Effect size variable			
WTP	Annual per household / per individual WTP for biodiversity conservation measure in 2005 EURO 27 ppp.	47.1	14.3
Characteristics of good			
RESTORE	Dummy: 1 = proposed biodiversity conservation measure is a restoration / enhancement; 0 = proposed biodiversity conservation measure is maintenance / protection from loss of the current status	0.5	0.13
WETAREA	Project area for which the biodiversity conservation measure is to be implemented in ha	128,809	107,088
Characteristics of site			
COASTAL	Dummy: 1 = wetland is a coastal or estuarine salt marsh; 0 = other	0.08	0.50
RIVERINE	Dummy: 1 = wetland is a riverine floodplain; 0 = other	0.52	0.114
PEAT	Dummy: 1 = wetland is a peat bog or fen; 0 = other	0.46	0.114
Characteristics of sample population			
POPAREA	Sample area from which the representative population sample was taken in km²	134,449	40,337
WETINDEX	Availability of substitute wetland sites in a 100 km radius from the wetland site measured in ha per 100 km²	1.17	0.32
INCOME	Disposable net income per inhabitant in 2005 EURO 27 ppp	1,0706	1,676
EAST	Dummy: 1 = country is eastern European transformation country, 0 = other	0.176	0.080
Characteristics of method			
CE	Dummy: 1 = Choice Experiment; 0 = Contingent Valuation.	0.32	0.137
OPEN	Dummy: 1 = open elicitation format , 0 = other, mainly dichotomous choice	0.54	0.129
MAIL	Dummy: 1 = mail survey , 0 = other, mainly face-to-face	0.44	0.129
TAX	Dummy: 1 = coercive payment vehicle , 0 = non coercive payment vehicle	0.64	0.133
NONPROTEST	Dummy: 1 = value estimate is based on non-protest bids, 0 = other, estimate includes protest bids	0.35	0.118
TRIM	Dummy: 1 = value estimate is based on a trimmed sample, 0 = other	0.15	0.063
HH	Dummy: 1 = value estimate elicited per household; 0 = value estimate elicited per individual	0.50	0.130
SINGLE	Dummy: 1 = value estimate is based on a single payment; 0 = annual payment	0.03	0.020

* full sample means weighted by n of primary study and the inverse of the number of cases of sampled populations of different extent included in the dataset from each study

Further, if reported, we include in the database for each of the sample populations each available estimate for conservation programmes of varying size. For choice experiments, we include the WTP estimate for the highest biodiversity conservation level for the smallest and largest area offered in the choice set.

There is also considerable variation between studies in regard to the treatment of protest bids and outliers, with some studies reporting the one or the other treatment and many reporting more than one treatment. We included all reported estimates from the studies in the database, so that there is intra- and inter-study variation of these attributes. On average we obtain 3.3 observations per study and a maximum of 7 observations per study.

5 Meta regression model

There are three properties of the meta-data that have implications for the choice of the meta-analysis regression model: sample heterogeneity, heteroskedasticity and intra-study correlation (non independence) of effect size estimates (Nelson and Kennedy 2009). Because the data used in the meta-analysis are characterized by multiple observations from multiple studies, they are likely to violate the assumptions of independent and identical distributed errors. To account for the panel nature we use clustered robust regression with weights based on sample size and number of observations per cluster based on the approach developed by van Houtven et al. (2007)[13]. We cluster according to study. Regressions are estimated using the complex samples generalized linear model procedure in SPSS. This method generates robust standard error estimates that correct for potential error correlation within study clusters and unequal variance of errors across clusters[14].The approach of this paper is to reduce methodological and factual heterogeneity of the effect size estimate from the onset by limiting the data included in the meta-analysis to a more homogeneously defined good

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13 There are several possible statistical approaches to handle this type of data. Nelson and Kennedy (2009) provide a recent review. They recommend random effects, but consider the GLS with robust standard errors approach used in this paper as a valid, option. Of the meta-analysis described in this paper, the clustered robust regression approach is also used by van Houtven et al. 2007, Lindhjem (2007) uses random effects model, Brouwer et al. (1999) use a hierarchical/multilevel approach. Woodward and Wui (2001), Brander et al. (2006), Gerhmandi et al. (2008), Smith and Osborne 1996, Barrio and Louriero (2010), Richardson and Loomis (2009) all use OLS with (and without) robust standard errors.

[14] The Huber-White standard errors are standard errors that are adjusted for correlations of error terms across observations, especially in panel and survey data as well as data with cluster structure. This type of adjusted error is also called sandwich, robust or empirical standard errors.

(conservation programmes) and valuation methodology (stated preference) . We further attempt to account for remaining methodological heterogeneity by including various dummy variables in the analysis.

We combine two weighting factors in the analysis to account for variance and intra-study correlation (cf. van Houtven et al. 2007 and Nelson and Kennedy 2009). Due to different sample size and estimation procedures, the willingness to pay estimates are generally expected to have non-homogeneous variances. If variance estimates were available, these could be used to give more reliable estimates a greater weight in the regression by weighting the effect size variable with the inverse of its variance. Unfortunately, relatively few studies document standard errors or confidence intervals for any or all of the reported WTP estimates. As a proxy, we use the sample size on which each effect size estimate is based. This value can be extracted from nearly all studies. Second, the meta-data set contains many value estimates that are measured for the same sample population and wetland programme, but are estimated using different elicitation methods or data treatment. Because these are estimates for the same case, we weight each observation by the inverse of the number of estimates for a case (INV_CASE), so that the within case weights for each sampled population sum to one. All regressions are estimated using a combined weighting factor that is the product of N and INV_CASE.

We use a log-linear functional form specification. The log-linear specification converts the dependant variable (WTP) and the wetland area, population area, substitute availability and income terms into logarithmic form, but leaves the other dichotomous variables in linear form. In the log-linear model, the coefficients measure the constant relative change in the dependant variable for a given absolute change in the value of the explanatory variable. The log-linear approach has at least two conceptual advantages. First, it implies that, as the size of the wetland conservation measure approaches zero, WTP also approaches zero. The logged effect size has therefore been used in virtually all meta-analysis related to natural resource valuation to ensure non negativity of the welfare measure. Second, it implies that the coefficients of the variables expressed as logarithms can be interpreted as elasticities that describe the percentage change in the dependant variable given a one percent change in the explanatory variable. The log-linear model therefore assumes a constant elasticity.

Table 3: Meta-regression results (clustered robust weighted least square regression)

Dependent	LN(WTP)		
	Coef.		SE
(Intercept)	-5,256	**	2,259
LN(WETAREA)	0,210	**	0,052
LN(POPAREA)	-0,146	**	0,048
LN(INCOME)	0,599	**	0,273
EAST	1,898	**	0,329
MAIL	1,370	**	0,194
TAX	1,289	**	0,224
NONPROTEST	0,697	**	0,216
SINGLE	1,493	**	0,535
CE	1,225	**	0,229
N	68		
adj. R^2	0,88		
Wald F	40,3	**	
MAPE	73		

(1) Sampling and weighting scheme: clustered by study (n=29) and weighted by the N of the primary study and the inverse of the number of cases of sampled populations of different extent included in the dataset from each study.

(2) ** significant at P<0.05 level, * significant at P<0.1 level

(3) Dependant variable: LN (WTP) in EURO 272005 ppp

6 Meta regression results

Table 3 reports the results of the meta-regression model[15]. Only variables that are statistically significant at the 5 % level based on t-statistics are retained in the model[16]. The resulting model provides good fit to the data, with adjusted R^2 statistics indicating that 88 % of the variation in mean WTP estimates across primary studies can be explained by the variation in the explanatory variables[17].

Most importantly, the results indicate that the mean WTP estimates are sensitive to the scope of the wetland conservation programme measured in terms of area. The coefficient on log WETAREA is positive and significant at the 99 % level. Because we use a double log model, the estimated coefficient reflects the constant elasticity of mean WTP for scope. The coefficient is less than one, i.e. inelastic. A one percent increase in area therefore results in a 0.2 % percent increase of mean WTP. We do not find a significant difference between improvements in wetland quality compared to avoided losses, as the coefficient on the variable RESTORE is not significant.

Of the variables describing characteristics of the sampled population, three variables are significant. The log of the size of the sampled area (POPAREA) is significant and shows the expected negative sign. This confirms our hypotheses that mean WTP decreases with the choice of a larger sample (or market) area. Also the mean household net income has a positive and significant effect on WTP. Because we use a double log model, the estimated coefficient reflects the constant income elasticity of mean WTP. The coefficient is less than one, i.e. inelastic. A one percent increase in mean household income therefore results in a 0.6 % percent increase of mean WTP. The variable EAST, denoting eastern European transformation economies, also has a positive and significant effect on WTP. The interpretation of this variable may be related to cultural or methodological effects, as

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[15] Alternative combinations of explanatory variables, and other weighting schemes were also explored; however they did not indicate any change in sign or statistical significance of the reported explanatory variables.

[16] The dropped variables therefore are WETINDEX, RESTORE, OPEN, TRIM, HH, SINGLE, PUB, RIVERINE, COASTAL, PEAT.

17 Previous meta-analyses for wetland studies explain roughly 38 % (Brouwer et al. 1999), 58 % (Woodward and Wui 2001), 38 % (Brander et al. 2006) and 44 % (Ghermandi et al. 2008). Other meta-analyses focusing on stated preferences studies for forest management or water quality improvements have explained ca. 74 % (Lindhjem and Navrud 2008), 83% (Barrio and Loureiro 2010) and ca. 60 % (van Houtven et al. 2007). Nelson and Kennedy (2009) point out the danger that R^2 values may be inflated, if a large number of estimates are drawn from the same studies. However, we also test regressions based on a single estimate from each study only and find similar values for R^2.

discussed above. No significant effects where found for the wetland availability index (WETINDEX).

The variables MAIL, TAX and NONPROTEST describing key methodological influences are all significant at the 5 % level. The variables MAIL, TAX and NONPROTEST are found to have a positive sign. This indicates that in the sample, WTP estimates from mail surveys are higher than from in-person surveys. The results further indicate that coercive payment vehicles (TAX) produce higher WTP estimates than voluntary payment vehicles. As expected, the exclusion of protest bids (NONROTEST) has a positive effect on the estimated mean WTP of primary studies. WTP based on single payments (SINGLE) is, as expected, significantly higher compared to annual payments. The results further show that WTP estimates from choice experiments (CE) are higher than from contingent valuation. Other methodological variables were not found to have a significant effect.

7 Implications for benefit transfer

As a next step we explore the implications of the regression results for predicting WTP to support policy analysis. A key issue is how reliably the estimated equations can provide values for benefit transfer. As a first step we looked at the in-sample forecast performance of the model using the mean absolute percentage error (MAPE)[18]. The results are also reported in Table 3. The average MAPE of our sample is 74 % and is of a similar dimension to other meta-analyses, with common values ranging between 50 % and 80 %[19].

In order to utilise the regression function as a benefit transfer function the appropriate values of the moderator variables for the policy site have to be plugged into the model. In particular, the area of the proposed conservation measure in ha (WETAREA), and the assumed market size in km² (POPAREA) , the average individual income in EURO 27 $_{2005}$ (INCOME)[20] and the location in eastern or western Europe (EAST) have to be defined for the policy site[21]. The remaining variables, that describe methodological choices, should be set to reflect best practice[22].

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[18] MAPE is defined as $\left|(y_{obs} - y_{est}) / y_{obs}\right|$

[19] For example, Brander et al. (2006) report an average MAPE of 58 %, Lindhjem and Navrud (2008) of 52 %.

[20] Because all monetary variables are given in ppp EURO 27 2005, the income variable (and the resulting WTP estimate) have to be adjusted accordingly.

21 The appropriate market size cannot be deducted from the function, because the underlying data used for the regression represents average WTP for the population of a specific sample area size

To illustrate the implications for benefit transfer, we predict mean WTP for several combinations of the area of wetland conservation measures, the size of the sample area and the income of the sample population that cover the range of values found in the primary studies. The predicted mean WTP estimates as a function of wetland area and sample area in Figure 1. This serves to illustrate the key point of this paper, that for benefit transfer, WTP estimates need to be scaled based on the scope of the proposed conservation measure. The valuation estimates cannot simply be scaled linearly based on a per unit area WTP estimate. This would lead to a substantial overestimation of the WTP. To estimate WTP to use in cost-benefit analysis of single wetland projects that are developed as a part of a larger regional wetland conservation programme, we would therefore propose a two step approach. WTP estimates should first be estimated for the scope of the whole program. Only in a second step should the resultant WTP be converted into a per unit area WTP estimate that can subsequently used consistently in the analysis of the single projects within a programme.

However, the margin of error in terms of the inner sample forecast and the 95 % confidence interval of the mean estimates is still rather high. Figure 2 illustrates the range of 95 % confidence intervals based on asymptotic errors. Whether this margin of error is considered large or too large depends on the use of the results. For some projects and policy applications it is probably acceptable and uncertainty of the final results can be dealt with through sensitivity analysis.

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and not marginal WTP as a function of distance. Therefore, one cannot deduct at what distance the WTP is expected to approach zero and an independent choice for the appropriate market size has to be made.

22 We use following settings: The use of coercive payment vehicles (TAX = 1) and an inclusion of protest responses (NONPROTEST = 0) can be considered good practice. For estimates of annual WTP, annual payment needs to should be chosen (SINGLE = 0). The variable MAIL is set at 0, under the assumption that face-to-face interviews would be the preferred method. We set the variable CE differentiating CE and CV methods at 0.5. Other choices are possible.

Figure 1: Predicted mean annual WTP in $EURO_{2005}$ as a function of the area of the wetland conservation measure (WETAREA) for different maximum extent of sample (or market) area (POPAREA)[23].

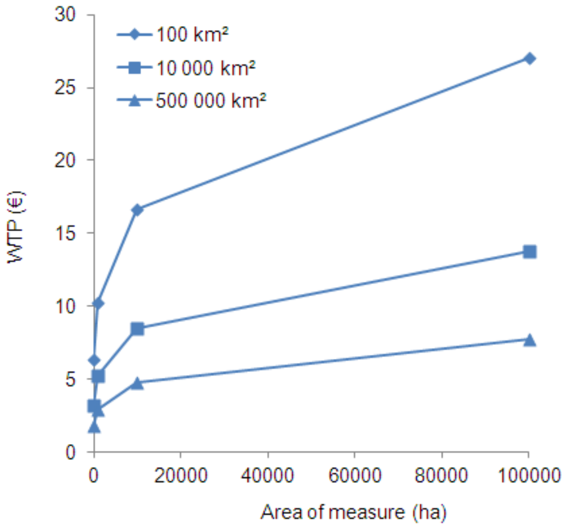
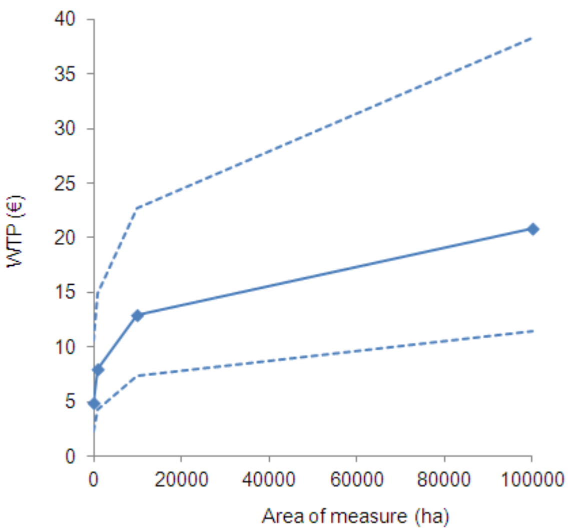


Figure 2: Predicted mean annual WTP in $EURO_{2005}$ as a function of the area of the wetland conservation measure (WETAREA) and 95 % confidence intervals based on asymptotic errors[24].



[23] other assumptions: INCOME = 10.000 €, EAST = 0, CE=0.5, SINGLE=1, MAIL=0, TAX =1, NONPROTEST=0

[24] other assumptions: POPAREA= 10000 km², INCOME = 10000 €, EAST = 0, CE=0.5, SINGLE=1, MAIL=0, TAX =1, NONPROTEST=0

8 Concluding remarks

The meta-analysis provides insight into key factors that prove to be influential and have to be accounted for when attempting to transfer values from stated preference studies for wetland habitat and biodiversity conservation programmes. Most importantly, we find evidence of a systematic and theoretically consistent pattern of WTP estimates. Particularly, we are able to describe WTP as a theoretically consistent function of scope of the valued good, the extent of the sample area and the sample population's income, which enhances the potential to use the regression for benefit transfer. We show that WTP estimates are significantly sensitive to scope of conservation programmes defined in terms of area.

However, the margin of error in terms of the inner sample forecast and the 95 % confidence interval of the mean estimates is still rather high. Whether this margin of error is considered large or too large depends on the use of the results. For some projects and policy applications it is probably acceptable and uncertainty of the final results can be dealt with through sensitivity analysis.

We find that beyond the framing of the quality change in terms of gain / avoided loss, it is difficult to develop a metric of the quality of wetland habitat change implicit in the conservation programmes, comparable for example to the widely used and simple scale for water quality ranging from boatable to swimmable. Our findings indicate that only the spatial or quantity dimension of the wetland conservation programme has a significant effect on WTP for wetland habitat conservation.. This could be because the valuation scenarios offered to respondents imply that the measures are suited to restore or maintain something like a "good wetland habitat quality status". Despite variations in the good descriptions in the primary studies, respondents may therefore on the large perceive wetland conservation as a relative homogeneous good. This is open to further analysis.

We also find that it is difficult to specify variables describing the sampled population. Whilst we find evidence of distance decay, it is clear that the mean WTP estimates reported in the primary studies for non-local samples are nearly all upward biased, because the sample designs over-sample the population close to the sites (cf. Bateman et al. 2006). The few studies that explicitly attempt to account for distance decay by stratified sampling, in most cases end up reporting uncorrected total sample means. For the type of meta-analysis proposed here, ideally distance and population weighted means should also be reported. A further impediment is that average household or individual income of the sample is not reported in several of the primary studies, so that a national average had to be used as a proxy.

The results also imply that the choice of method in general has a large influence on the mean WTP estimate compared to the variation in good and market specification.

Methodological variables are however not particularly useful as explanatory variables for benefit transfer. Despite the increase in number of primary studies, there is still a need for more valuation studies for wetlands. However, there is also a need for more precise definition and documentation of the valuation scenarios and more complete reporting of information on the samples (cf. Loomis and Rosenberger 2006). With a larger and yet more homogeneous dataset based on well documented and well specified primary studies, we assume that eventually more robust estimates for benefit transfer can be generated. In the meanwhile, careful interpretation of the available estimates as has been attempted in this study may provide estimates that can be used in cost-benefit assessment of wetland policies.

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PAPER IV

SOCIAL BENEFITS AND ABATEMENT COSTS OF GREENHOUSE GAS EMISSION REDUCTIONS FROM RESTORING DRAINED FEN WETLANDS - A CASE STUDY FROM THE ELBE RIVER BASIN (GERMANY).

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This paper presents estimates of the costs and benefits of reducing greenhouse gas (GHG) emissions through fen wetland restoration. This study takes previous research on GHG emissions from peat wetlands further by coupling water level dependent emission functions with a landscape and basin scale assessment of the hydrology and water management of wetlands. For this purpose we use a water management model for the Elbe River Basin that includes the major lowland fen wetland sites as water users. Based on the resultant estimates of the GHG emissions of wetlands and the reduction potential of management options under more realistic description of water availability, this paper provides improved estimates of the benefits of restoration in terms of the shadow price of carbon and the GHG abatement costs of wetland restoration measures. An econometric approach is used to develop abatement cost estimates for wetland restoration.

We find mean current emissions over 35 wetland sites (3,840 km²) to be in the range of 17.5 -25.5 $tCO2e\ ha^{-1}$. The median of estimated abatement costs for fen stabilisation scenarios are within a range of 10 – 20 € $tCO2e^{-1}$ and for restoration scenarios in a range of 7 -14 € $tCO2e^{-1}$. Fen wetland restoration can therefore potentially contribute to mitigation targets at low costs. An approach focused on restoration is a more efficient strategy compared to an approach centred on agri-environmental schemes, even though both components are required in a zoning approach. However the effects of climatic change may reduce the effectiveness of wetland restoration measures by roughly 50%.

Keywords: economic valuation, greenhouse gas emissions, peat wetland, water management, abatement costs, climate change.

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1 Introduction

The aim of this paper is to investigate the potential and the costs and benefits of reducing greenhouse gas (GHG) emissions through fen wetland restoration. Peat carbon sequestration is a result of low biomass decay rates under anaerobic conditions in waterlogged wetland sites. When wetlands are drained the peat is no longer conserved but decomposed, because lowering the water table stimulates aerobic decomposition of the peat. Next to sequestering carbon peat lands may also emit methane and nitrous oxide. In the case of peat land drainage, methane (CH_4) emissions decrease and nitrous oxide (N_2O) emissions increase. In the case of rewetting the opposite occurs: carbon dioxide (CO_2) and N_2O emissions strongly decrease, while CH_4 emissions increase, especially in the initial restoration phase. However, the overall balance is that restoration substantially reduces the net emissions of GHG into the atmosphere (Couwenberg, 2008). The rational for considering peatlands as potential mitigation options to reduce GHG emissions therefore is the preservation of the existing large carbon stocks in peat soils and the reduction of anthropogenic induced GHG emissions rather than an increase in the soil carbon stocks in the short term (Freibauer et al., 2004). The total area of peat wetlands in Germany is estimated to be 1.364 million hectares of which 76% are fen peat and 34% are bog peat. In Germany, roughly 95% of peat wetlands have been drained for agricultural purposes (Höper, 2007).

The restoration of peat soils can potentially provide a significant contribution to GHG emissions abatement efforts (Parish et al., 2008; Kat and Joosten, 2008). Greenhouse gas regulation is one of a large number of ecosystem services that are provided by wetlands, such as fodder production, recreational opportunities, habitat and biodiversity conservation or regulation of fluxes (Turner et al., 2008). The reduction of GHG emissions therefore is an important co-benefit of wetland restoration projects in general. In economic terms, the emissions of GHG induced by anthropogenic modification, especially drainage, are a negative external effect of the land use system, because the emissions impose costs upon society that are not reflected in the private costs of wetland drainage and water table regulation for agricultural production. These emissions impose opportunity costs on society by raising the overall emission level relative to target levels, in particular those imposed under the Kyoto Protocol. A decrease in emissions as a consequence of wetland restoration would imply less emission reductions elsewhere would be required to comply with emission reduction targets. In the economic appraisal of wetland restoration and river basin water management options, these externalities need to be taken into account. A growing number of governments and organisations have started to use a shadow price of carbon to value the externality for in cost-benefit and policy appraisals (cf. UK Department of Energy and Climate Change (DECC), 2009; Umweltbundesamt (UBA), 2007). The rational for this approach is to make policy and investment decisions across sectors comparable, regarding their effects on GHG emissions. A social price of carbon approach has been used for example to generate information on the value of carbon sequestration co-benefits generated from public investments into forest management (Brainard et al., 2009), wetland restoration programmes (Schäfer 2009; Worall et al., 2009) or biodiversity conservation (Bratt and Ten Brink 2008). There are basically two approaches to defining a shadow price of carbon, either based on the marginal damage costs of carbon or the marginal abatement costs. Due to uncertainties regarding the damage costs (Tol, 2009; Stern, 2007), a climate policy target consistent approach based on estimates of the abatement costs that will need to be incurred to meet specific reductions targets is considered to be more appropriate for valuing GHG emissions in project appraisals (Tol and Lyons, 2008; DECC, 2009). This paper presents an application the marginal abatement cost method to value the GHG abatement benefit of wetland restoration measures.

To determine how scare resources should be optimally allocate on activities that reduce atmospheric GHG it is also necessary to be able to compare the per unit costs of different activities that reduce GHG emissions. This helps to choose the best mix of sequestration and emission reduction options for mitigating climate change in an economy. We also use our model of landscape scale GHG emissions to investigate the costs of abating GHG emissions through fen peatland restoration. Potential methods to calculate the costs of abatement measures are sectoral optimisation approaches, econometric or statistical approaches and bottom up or micro engineering approaches. There is little published information on the abatement costs of wetland restoration measures in temperate areas. Under current United Nations Framework Convention on Climate Change (UNFCCC) reporting categories, emission from peatlands are reported separately under forestry and agricultural land use activities. Whilst there is a large body of evidence on the abatement costs from forestry related activities (cf. van Kooten et al., 2009) there is less information on cost in the agricultural sector (cf. Schneider and McCarl, 2006; Cara et al., 2009). However, these studies only indirectly address emissions from organic soils as part of the overall soil related sequestration potential. In contrast there are numerous studies that estimate costs of wetland restoration measures, for example based on farm level optimisation approaches (cf. Vogel, 2002 for agri-environmental schemes in the study area), activity budget approaches (cf. Schäfer, 1999 and 2005 for reed and alder production alternatives in the study area) or econometric approaches (cf. Söderquist, 2002 for an example). Published studies that relate the costs of restoration to the attainable emission reductions are however scarce (cf. Schäfer, 2009; Warell et al., 2009). With this paper, we provide an addition to the restoration cost literature based on an econometric approach that uses empirical data from implemented restoration projects.

In the framework of climate politics, GHG fluxes (emissions by sources and sequestration by sinks) must be quantified. While no finalized best guidance on estimating the GHG emissions from temperate fens is available to date, it is consensus that the main factors controlling GHG fluxes of temperate fen wetlands are largely related to aspects of water management and land use (cf. Byrne et al., 2004; Höper, 2007; Couwenberg et al., 2008;

Joosten and Couwenberg, 2009). Recent aggregated estimates for GHG emissions from peat wetlands in Germany are presented by Höper (2007) and Freibauer et al. (2009). For example Höper (2007) estimates the mean GHG flux from fen peat soils under current management conditions to be 25.6 $tCO2e_{GWP100}$ ha^{-1}. In addition the national inventory report (UBA, 2009) also provides estimates of emissions from peat soils under agricultural and forestry use. The estimated total GHG fluxes are equivalent to roughly 2.5 - 5% of total German GHG emissions and roughly 70% of the net flux from agricultural soils. The GHG emissions from wetlands compensate roughly half of the annual carbon sequestration in the forestry sector (Freibauer et al., 2009). Several authors have also presented estimates of the abatement potential. Hirschfeld et al. (2008) assume that the emission reduction potential from peatland restoration is equivalent to the emission factors of 40 and 18 tCO2e ha^{-1} for fen and bog peat under arable and grassland cover that are used to estimate the emissions for the national inventory report. Freibauer et al. (2009) estimate an average reduction potential of 30 t ha^{-1} for fen and 15 t ha^{-1} for bog peatlands. For the state of Mecklenburg-Vorpommern, Schäfer (2009) reports the emission reductions attainable from fen restoration projects to be estimated in a range of 10.5 - 12.5 tCO2e ha^{-1}.

The aggregation of emissions from peatlands at the landscape level (a scale larger than a single site) is in most applications based on per unit area estimates for habitat - and land use types. While it is acknowledged that water levels are a key determinant of emission and are suitable as a proxy indicator to estimate GHG emissions (Joosten and Couwenberg, 2009), for lack of spatial data few landscape level assessments take the spatial variation of water levels explicitly into account (cf. Warell et al., 2009 for an example from upland peatlands). Whilst our work builds on available approaches to describe the GHG emissions from peat soil hydrological response units based on water levels, this study takes previous research on GHG emissions further by coupling such water level dependent emission functions with a landscape and basin scale assessment of the hydrology and water management of wetlands. For this purpose we use a water management model for the Elbe River Basin that includes the major lowland fen wetland sites as water users. This modelling approach allows for a spatial assessment of ground water levels as a function of basin - and wetland water management as well as basin water availability. In modelling water availability we take projected climatic trends for the Elbe Basin into account. Based on the resultant estimates of the GHG emissions and reduction potential of wetlands under more realistic description of water availability, this paper provides improved estimates of the benefits of restoration in terms of the shadow price of carbon and the abatement costs of wetland restoration measures.

The rest of this paper is organised as follows. We begin with an overview of the study area. We then describe our methodological approach for the water management model and the estimation of GHG emissions, the shadow price of carbon and the abatement costs. After a description of the restoration scenarios we present key results. We close with a discussion of policy implications.

2 Study area

For this study we consider the lowland wetlands of the Elbe Basin. These are areas with groundwater levels near to the surface and include both sandy and fen peat soils. We consider the water resources management of the whole Elbe basin (148,000 km²), but we restrict our analysis of wetland water balance and GHG emissions to lowland wetlands with an area larger than 1,000 ha and with active water management systems. A total of 35 major wetlands with an area of 1,200 ha to 40,000 ha (3,840 km² in total) were selected (Figure 1) (Dietrich et al., 2010). Roughly 50% of the selected wetland sites are groundwater-influenced sandy soils and 35% are peat soils. The peat soils are differentiated into shallow peat above sand (20% of the total area) and deep peat (15% of the total area). The main land use type is grassland (54% of the total area). Arable land uses (34%) are primarily found on sandy soils. All of the selected wetlands have water regulation systems in place. During the last century, the majority of the wetlands have been drained in order to intensify agricultural production. Due to the low levels of precipitation, the drainage systems were augmented with weirs in the 1970s and 1980s to regulate water levels and to enable sub-irrigation. This was a prerequisite for intensive agricultural production. As a result this region today has a complex water regulation system that is an integral part of the regulation system for the whole river basin. Within the wetland sub-basins, short term decisions on water levels and water distribution are made in watershed advisory committees on the basis of existing formal and informal water use rights. Water allocation within the river basin is in the responsibility of the state water authorities. Major changes in land use and water management that are associated with the restoration of wetlands require a formal land use planning process. Restoration initiatives are often contested by agricultural land owners, as changes in land and water use rights have to be negotiated and compensated. State governments in the Elbe Basin have developed various programmatic approaches to support the restoration of peat wetlands (cf. Kowatsch, 2007). These wetland conservation programmes typically encompass two types of measures: support to low intensity agricultural management practices adapted to high water levels and support to permanent rewetting and reconversion of agricultural land. Due to the hydrological interdependencies at the landscape level, these two types of measures are often combined in a zoning approach.

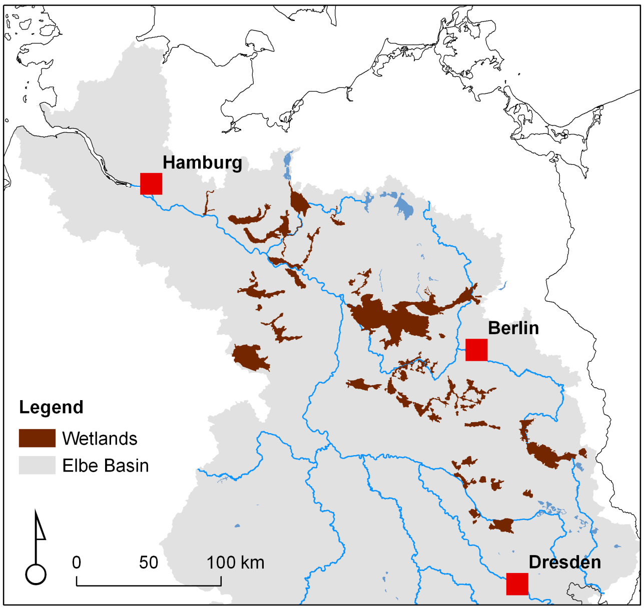

Fig. 1: Location of the selected wetland sites within the lowland of the Elbe Basin

3 Methods

3.1 Water resources management model

The hydrological analysis is based on the stochastic simulation of water management for each of the wetlands using the WBalMo model system (Kaden et al., 2008). The model balances water availability and water abstraction by the most important water users in a river basin. These users can be characterised by their position in the river network, their monthly abstraction and return flows and their priority ranking in relation to other users. The model for the Elbe basin was complemented by the implementation of detailed sub-models for the 35 major wetland sites that are composed of 457 regulation entities or water users, in order to describe the complex water use process in these major wetlands. This sub-model is described in detail in Dietrich et al. (2007).

In the WBalMo Elbe model, each wetland sub-area is designated as a water user for which monthly water balances can be calculated. The model therefore explicitly takes the effects of wetland restoration at upstream sites on downstream flows into account. The sub-areas are delineated based on the underlying concept of hydrologic response units (HRU). The main soil types (peat, sand, and loam), a digital elevation model, land use and sub-areas are intersected by means of GIS to define the HRUs. The sum of hydrological responses from all the HRUs of a sub-area gives the reaction of the sub-area being investigated, and the sum of these gives the reaction of the whole wetland. At the same time it is possible to couple ecosystem processes with hydrological processes on the basis of the HRU concept. Mean depths to groundwater are derived for all HRUs for every month of the simulation period. The GHG emissions are modelled by coupling the emission functions to the simulated groundwater levels below floor of each HRU.

The model uses a stochastic approach to calculate the respective probabilities for different water levels and GHG emissions for every sub-area and aggregated wetland sites. This is done by calculating 100 statistical realisations of climatic and water availability scenarios over a time period of 50 years. The time step of the model is one month. The expectation (or average annual) of the global warming potential (GWP) for a wetland in year t is then calculated as follows:

$$E(GWP_t) = \sum_{r=0}^{1} \left[P_r \cdot (\sum_{wl} GWP_{wl} \cdot A_{wl,r,t}) \right] \tag{1}$$

where P is the occurrence probability of a realisation r and where $\sum_{r=1}^{n} P_r = 1$ to ensure normalisation. Wl is the mean annual groundwater level below floor and A_{wl} is the fen area with water level wl in realisation r, and GWP_{wl} is the global warming potential at water level wl.

3.2 Estimation of the greenhouse gas emissions and warming potential

Global warming potentials (GWP) are conventionally used to compare the relative contribution of GHG fluxes to the earth's radiative balance. The GWP we use is based on Intergovernmental Panel on Climate Change (IPCC) (1995) where $GWP_{CO2\text{-}C}$ =1, $GWP_{CH4\text{-}C}$ = 7.6 and $GWP_{N2O\text{-}N}$ = 133.

Following the approach by Höper (2007) and Drösler (2005) the GWP balance of the GHG exchange for wetlands is calculated as:

$$GWP_{CO2e} = (NEE_{CO2-C} \cdot GWP_{CO2-C} + F_{CH4-C} \cdot GWP_{CH4-C} + F_{N2O-N} \cdot GWP_{N2O-N}) \cdot 44/12 \tag{2}$$

where GWP is measured in CO2e, $NEE_{CO2\text{-}C}$ is the net ecosystem exchange of carbon measured in CO2-C, $F_{CH4\text{-}C}$ is the mean annual flux of methane measured in CH4-C, $F_{N2O\text{-}N}$ is the mean annual flux of nitrous oxide measured in N2O-N and GWP are the corresponding elementary global warming potentials. The factor 44/12 converts from Ce to CO2e which is the usual accounting unit. The GWP balance is an atmospheric balance, so that emissions from wetlands have a positive and sinks a negative sign.

Carbon sequestration by carbon accumulation occurs at high water levels under anaerobic conditions. The difference between the carbon fixation by photosynthesis and respiration from the ecosystem is the net ecosystem exchange (NEE). The carbon surplus gained in the process is available for long term carbon accumulation. However a share is emitted again by anaerobic respiration as methane (CH4) or lost as dissolved organic carbon (DOC) through the subsurface water flow. Under aerobic conditions carbon is no longer accumulated in peat but emitted as a result of peat degradation (cf. Kluge et al., 2008). Following Strack et al., (2008) and Höper (2007) we proceed to estimate NEE flux into the atmosphere simplifying as follows:

$$NEE_{CO2-C} = \Delta C - F_{CH4-C} \tag{3}$$

and

$$\Delta C = LORCA_{CO2-C} \text{ for WL < 10 cm below ground} \tag{3a}$$

$$\Delta C = F_{CO2-C} \text{ for WL > 10 cm below ground} \tag{3b}$$

where ΔC is the net change in soil carbon storage[1], F_{CH4-C} is the methane carbon flux, $LORCA_{CO2-C}$ is the long term rate of carbon accumulation and F_{CO2-C} are the soil carbon emissions from peat degradation.

Peat accumulation can be expected at water levels higher than 10 cm below ground (Couwenberg et al., 2008). A possible method to estimate the long term rate of carbon accumulation (LORCA) is to estimate the average annual peat accumulation rate, the peat bulk density and its carbon content. We use such an approach provided by Blankenburg et al., (2001), who estimate a water level dependent function of annual peat accumulation for northern German fen soils. To estimate the emissions commencing at water levels below 10 cm we use water level dependent carbon emission functions provided by Renger et al. (2002) for north-east German fen soils.

The emissions of methane depend on water levels and are virtually zero at water levels lower than 20 cm below surface but rise rapidly with high water tables. Nitrous oxide emissions are restricted to mean water levels below 20 cm and are negligible in case of natural peatlands (Höper 2007; Couwenberg et al., 2008). For methane emissions we use a water level dependent function provided by van Dasellar et al. (1999) for fen soils in the Netherlands. The trajectory fits well with the more recent meta-analysis reported by Couwenberg (2008), however methane emissions subside with the duration of

[1] Note that NEE and ΔC are viewed from the atmospheric carbon balance, so that LORCA has a negative sign and F_{CH4-C} and F_{CO2-C} have a positive sign.

restoration, so we also conduct sensitivity analysis for a 50% lower trajectory that roughly corresponds to a lower bound of the reported emissions. No water table dependent functions are available for nitrous oxide emissions. For lack of better approaches, we resort to a simple step function with zero emissions at water levels above 20 cm and mean emissions of 8 kg N_2O-N ha^{-1} for water levels less then 20 cm. This is the default emission factor given for 'temperate organic crop- and grassland soils' that is used in the German national inventory (UBA, 2009).

Finally, we combine the approaches to generate a water level dependent function of GWP as described in (2) over the whole range of average annual water levels from 0 – 120 cm below soil surface. The functions and parameters used are summarised in Table 1. We use two combinations of parameters to generate a high and a low impact trajectory for the resultant GWP function. The high impact trajectory combines higher CO_2 emissions at low water levels with lower estimate of CH4 emissions at high water levels, so that the net effect of wetland restoration is greater than in the low impact alternative that combines a lower estimate for CO2 emissions at low water levels with higher estimate of CH4 emissions at high water levels.

Figure 2 provides an overview of the resultant greenhouse gas emission functions. Both of the resultant trajectories assume that restored fen wetlands do not have a negative GWP balance, but that significant reduction of the GWP balance are possible compared to a drained status.

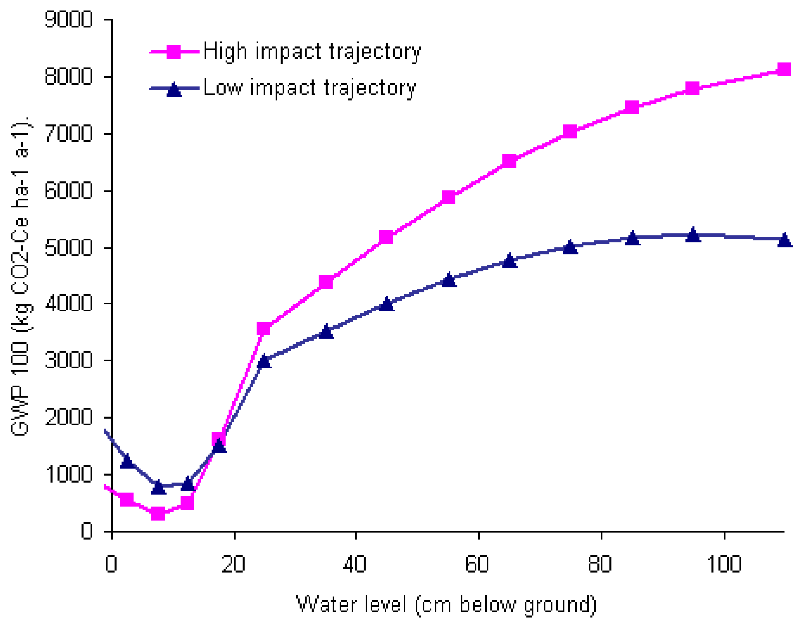

Figure 2: Water level dependent emission of greenhouse gases (high impact and low impact trajectory) measured in global warming potential.

Table 1: Water level dependant functions used to estimate greenhouse gas balance of fen peat soils.

GHG balance component		Valid for water level range	Unit	Emission trajectory	Equations a	Source
Longterm rate of carbon accumulation	LORCACO2-C	-10 - 10 cm	CO2–C kg-1 ha-1	High and Low	(0,143*(WL-10))*CD*10*CC	Blankenburg et al. 2001
Soil carbon emissions	FCO2-C	10 – 150 cm	CO2-C kg-1 ha-1	High	(WL*121)-(0.482*WL^2) -121	Renger et al. 2002
				Low	(WL*96)-(0.5*(WL)^2) -170	
Methane emissions a	FCH4-C	-10 – 150 cm	CH4-C kg-1 ha-1	High	EXP(3.57-0.08*WL)*10 * CF *0.5	Van Dassler et al. 1999
				Low	EXP(3.57-0.08*WL)*10 * CF	
Nitrous oxide emissions	FN20-N	-10 -150 cm	N2O-N kg-1 ha-1	High and Low	IF(WL>20;8;0)	Höper 2007

a WL: water level below ground in cm

b CD: peat density in peat of 80 g l-1 and CC: carbon content fraction of 0,43

c for the high impact trajectory we apply a CH4 emissions estimate reduced by 50 % compared to reference function

d CF: we use a conversion factor of 12/16 to convert from CH4 to CH4-C.

3.3 Social costs of greenhouse gas emissions

We apply an abridged version of the method developed by DECC (2009) to price carbon emissions. This is a target consistent approach using estimates of the abatement costs that will need to be incurred to meet specific emissions reduction targets. The approach is designed to make sure that the appraisal of carbon related benefits are consistent with policy targets at the national, European and global level and differentiates between a short and along term price of carbon. In the short term, it is argued that policy targets up to 2020 are distinct and non-fungible for the targets in sectors covered by the EU Emissions Trading Scheme (ETS) and those in non traded sectors. In the long term, from 2030 onwards, it is assumed that consistent with the development of a more comprehensive global carbon market the prices of carbon will converge into a single global traded price of carbon.

However, as robust data on marginal abatement costs for the non traded sector targets for Germany has not been compiled officially, in this paper, we resort to using price projections for the EU Emissions Trading Scheme (ETS) traded sector only. Such a pragmatic approach based on market price projections has for example also been suggested by Tol and Lyons (2008) as the appropriate method for incorporating costs of GHG emissions in the economic appraisal of investment projects. The DECC approach uses model based estimates both of the short term and the long term price of carbon for 2008, 2020, 2030 and 2050. Linear interpolation is used to form a price series and a sensitivity range of +/- 50% is proposed. The resultant price series are summarised in Table 2.

Table 2: Projected price schedule for carbon traded under EU Emissions Trading Scheme (ETS) from 2008 – 2050 in $€_{2008}$ $tCO2^{-1}$

	2008	2020	2030	2050
Low	14	18	42	120
Central	26	30	84	240
High	31	37	126	360

Source: DECC 2009

The initial price of carbon rises from roughly 26 € $tCO2^{-1}$ in 2008 to 30 € $tCO2^{-1}$ in 2020. These values are close to the prices suggested by Tol and Lyons (2008), who argues for a carbon price to be based on a combination of the price of carbon futures within the European Trading Scheme in the short run and IPCC-based projections of carbon price on the long term. This would lead to an initial carbon price of roughly 26 € $tCO2^{-1}$ for 2008 that rises to 32 € $tCO2^{-1}$ in 2020. To put this further into relation, in its guidance on pricing external environmental effects, the German Environmental Protection Agency

(UBA 2007) argues that the marginal abatement cost for measures to comply with the targets under the Kyoto Protocol for Germany of a 21% reduction compared to 1990 are expected to be in the range of 20 € $tCO2e^{-1}$. This is also the finding of McKinsey (2007), who estimate that GHG emissions in Germany can be reduced by 26% compared to 1990 if all available abatement measures with abatement costs up to 20 € $tCO2e^{-1}$ are implemented. We conclude that the proposed range for sensitivity analysis of 50% will incorporated the central estimates of carbon prices generated with different methods.

The benefit of a restoration measure in year t in terms of the avoided marginal abatement costs of carbon is then calculated as:

$$SC_t = (E(GWP_t^{baseline}) - E(GWP_t^{measure})) \cdot spC_t \tag{4}$$

where spC is the social price of carbon in € $tCO2^{-1}$ and E(GWP) is the expectation value of global warming potential.

3.4 Estimation of the costs of wetland restoration

Most fen wetland conservation programmes in Germany include two major types of measures (cf. Kowatsch, 2007): stabilisation of fen peat through adapted agricultural management practices and complete restoration of fen peat sites by rewetting that involves permanent conversion of agricultural land use and water management infrastructure. These measures generally need to be combined in a zoning approach, so that a cost estimate for both types of measures is required..

Table 3: Fen wetland restoration projects: descriptive statistics

Variable		Unit	Mean	Range
Total project expenditure	PC	€ ha^{-1}	3193	826 - 8783
Restored wetland area	A_{total}	Ha	3282	70 – 20000
Share of restored wetland area that is purchased from the total project expenditures	$A_{purchase}$	ha/ha	0.38	0.01 – 0.77

N = 21 projects implemented between ca. 1998 and 2008.

We estimate a cost function for the investment and opportunity costs of wetland restoration based on the reported expenditures for 21 large scale lowland wetland restoration projects that have been implemented in the Elbe River Basin in the last ten years[2]. A data base was compiled from project summary reports. The reported items are

[2] Data on projects was collected from the databases on current and completed projects available at http://ec.europa.eu/environment/life/ for EU-Life Projects and http://www.bfn.de for projects of the German Federal Agency for Nature (BfN). The data was augmented by additional review of the individual project websites and fen conservation programmes of the Federal States of Germany

total project expenditures and expenditures for planning and project implementation, the purchasing of land and for the removal of water regulation and drainage infrastructure and embankments. The expenditures do not include recurring payments under agri-environmental schemes. We interpret the resources required for land purchase as the opportunity costs of forgone benefits from agricultural land use. The collected data further includes information on the restored wetland area and the area of land that was purchased from the total budget. In most cases, not all of the restored land was purchased because public land was provided at no cost to the project. Table 3 summarizes the descriptive statistics.

We tested various specifications and find that the per unit area expenditures for lowland wetland restoration is best explained by the share of the wetland area that was purchased from the total budget. Exploratory data analysis suggests the following cost equation:

$$PC = \beta_0 + \beta_1 (A_{purchase})^2 \tag{5}$$

where PC are the total project costs in € ha^{-1}, and $A_{purchase}$ is the share of total land area that is purchased from the total project expenditures. Table 4 reports results for a OLS regression model. We conclude that the cost function can be used for a rough estimation of the investment costs of lowland restoration measures at different sites within the Elbe basin. The abatement cost estimate for management options in this paper assumes that land from the public sector project nearly always has an opportunity cost. We therefore set $A_{purchase}$ at 80%, which is the upper limit of the observed range of values; this yields average investment and opportunity costs of restoration of ca. 7000 € ha^{-1}.

Table 4: OLS estimation results for the cost equation

Coefficient	Estimate	SE	Sig.
ßo	989	345	0.010
ß1	11508	1358	0.000

Adjusted $r^2 = 0.78$, n = 21

In addition we estimate the opportunity costs of the reduced agricultural productivity under the stabilisation target on the basis of payments offered under agri-environmental schemes. The payments are mostly funded or co-funded from the second pillar of the common agricultural policy of the European Union (EU-CAP). These payments are granted to farmers for maintaining high water levels and are paid in addition to compensations for low intensity grassland use. The payments are calculated to compensate average income losses and include an additional incentive component to cover transaction and risk costs. We therefore consider these payments to be a rough estimate of the opportunity costs. Table 5 summarizes payments granted in the German

Federal States in the current period. For our analysis, we use a central estimate of 200 € ha^{-1}.

We use the cost estimates to calculate the cost-effectiveness of wetland management measures regarding the reduction of GHG emissions. This is defined as:

$$CE = \sum_{t=0}^{T} \frac{1}{(1+r)^t}(C_t - oESB_t) \Bigg/ \sum_{t=0}^{T} \Delta E(GWP_t) \tag{6}$$

where C are the costs of the measure in year t, r is the discount rate, oESB are the co-benefits from other ecosystem services in year t and $\sum \Delta E(GWP_t)$ are the lifetime tonnes of GHG saved compared to the baseline, measured in CO2e. For our analysis, we focus on the GHG benefits alone and do not value non greenhouse co-benefits. The resultant estimate of the cost-effectiveness therefore has to be considered as an upper bound estimate, as the consideration of further co-benefits would reduce the abatement cost estimate. We use a social discount rate for public investments recommended by the German government of 3% (UBA, 2007).

Table 5: Compensation granted under agri-environmental payments schemes for fen wetland conservation and maintenance of high water tables in the German Bundesländer of the Elbe Basin.

Bundesland	Payments (€ ha^{-1} a^{-1})
Mecklenburg-Vorpommern	175
Lower Saxony	286
Brandenburg	200
Schleswig Holstein	245 – 320
Saxony-Anhalt	195 – 250

Source: ELRE programmes for the period 2008-2013

4 Climate scenario and water management options

4.1 Climate scenario

For the projection of future climatic conditions and water availability we use the climate scenario "STAR T2". This baseline projection and methods used for downscaling are described in detail in Wechsung et al. (2008). This climate scenario is based on a mean temperature increases of 2 K by 2050 compared to 1960-1990 that was downscaled with the regional climate model STAR (Werner and Gerstengarbe 1997; Orlowsky et al., 2008). The key effects for the study region are a shift in intra-annual precipitation distribution. While there is a reduction of precipitation in summer, winter precipitation increases slightly. In combination with an increase in potential evapotranspiration this leads to an increased deficit in the climatic water balance in the summer. The statistical regional climate model STAR allows the calculation of many realizations by using the Monte Carlo simulation technique. In this manner, we use 100 realizations of meteorological (precipitation, potential evapotranspiration) and hydrological (discharge) input data for the WBalMo Elbe model for the time series 2003 to 2052. The discharge from the sub-basins into the wetlands was calculated with the precipitation runoff model SWIM (Conradt et al., 2007).

4.2 Water management options

Water level control and control of flows within the stream network of the wetlands are the key elements of water management options within the wetlands. The model system WBalMo Elbe allows to simulate different management options by variation of input parameter sets for target water levels and water distribution rules at nodes (Dietrich et al., 2007). In this paper we compare a baseline and four alternative management options (Table 6). The baseline replicates the current management approach in terms of water level control targets and distribution of flows between the sub-areas in the wetland. The alternative options are hypothetical options defined in such a way that the maximum abatement potential under different management strategies can be explored.

Table 6: Definition of the wetland management options

Management option		Fen water level targets			Required changes in land use		
		Wi	Su	Duration of winter target	Area [a]	Extensive use	Conversion
		cm	cm		ha	%	%
baseline	Base	30	45	April	-	-	-
stabilisation / less ambitious	Stab A	20	40	May	17,418	74	26
stabilisation / more ambitious	Stab B	20	40	June	81,864	79	41
restoration / less ambitious	Rest A	0	0	All year	61,277	38	62
restoration / more ambitious	Rest B	0	0	All year	89,681	36	64

a the total fen area is 132.674 ha and the total wetland area is 385.500 ha.

The description of current water management (baseline) is based on information from the Water and Land Management Associations. The target water levels depend on the current land use. There are regional differences, but in principle grasslands have higher water levels than arable land use and the water levels for more extensive grassland management are higher than for intensive grassland. For all land uses the target water levels are higher during the winter and are lowered in spring to regulate soil moisture conditions suitable for agricultural production. The mean values for grassland are 30 cm below surface in winter and 45 cm below surface in summer and 45 cm below surface for arable land use in winter and 60 cm in summer.

The alternative management options build on the target categories of fen conservation programmes - fen stabilisation and fen restoration (cf. Landesumweltamt Brandenburg (LUA-BB), 1997; Ministerium für Landwirtschaft; Umwelt und Verbraucherschutz Mecklenburg Vorpommern (MLUV-MV), 2009). They build on spatial combinations of these target categories. Table 6 summarises the target water levels and the required extent of land use change for the defined management options. Two options focus on large scale fen stabilisation by low-intensity land uses and two further options on large scale fen restoration.

The management options are built on following assumptions:

- the management option "stabilisation / less ambitious" (Stab A) assumes low intensity land use for fen dominated sub-areas (fen area > 20% of the sub-area). This is implemented by changing all arable land on peat and peat over sand soils to grassland. In addition the winter water level targets for grassland on fen soils are raised to a minimum of 20 cm and summer targets to a minimum of 40 cm below ground;
- the management option "stabilisation / more ambitious" (Stab B) assumes low intensity land use on fen soils combined with higher summer water level targets and a longer duration of winter water level targets than Stab A. The duration of winter water level targets is prolonged to end of May and summer water levels are to be implemented from July. In addition summer water targets are raised to 30 cm below ground;
- the management option "restoration / less ambitious" (Rest A) assumes restoration of fen soils. Target water levels for fen dominated sub-areas (fen area > 50% of the sub-area) are raised to surface level throughout the year. All affected arable land and grassland are converted to natural wetland habitats;
- the management option "restoration / more ambitious" (Rest B) assumes restoration of fen soils. Target water levels for fen dominated sub-areas (fen area > 20% of the sub-area) are raised to surface level throughout the year. All affected arable land and grassland are converted to natural wetland habitats.

5 Results

5.3 Effects of climate change and water management on water tables

The model system was used to calculate the water balance and groundwater levels below floor for the period from 2003 to 2052. In Figure 3 we present selected results of the share of the wetland area with groundwater levels below surface above an evaluative threshold for all management options. Groundwater levels below surface are the water balance indicator that is most closely related to the GHG emissions. We present the July values because this is one of the months where the water levels are at their lowest level during the year. The values represent a mean hydrological year (50^{th} percentile of 500 values) of period 2 (2008/2012) and period 10 (2048/2052).

The results clearly illustrate that current water availability is insufficient to maintain the water levels targets under the baseline or alternative management options. This is aggravated in the future by the effects of climatic change. The share of area targeted to have water levels higher than 50 cm below floor increases as the management options become more ambitious regarding the wetland restoration targets. While under the baseline management option this target applies to roughly 15% of the area, it increases to roughly 40% for the larger restoration option. Under current climatic conditions this range is reduced to 10 - 20% and under future climatic conditions to 5 - 10%. In the baseline and all management options roughly 60 - 70% of the wetland area is targeted to have water levels higher than 100 cm below floor. However under current climatic conditions, this is only achieved for roughly 35 - 50% and under future conditions only for 18 - 35%.

While all management options lead to larger areas with higher groundwater levels, there are differences in the impact of the options. If we look at the differences between water management options within each of the two periods, we see that the effect of the less ambitious stabilisation option is small compared to the baseline. The effect of the other three options is similar, despite clear differences in the target water levels. The reason for this difference is that there is insufficient water available to maintain the target water levels. The results imply, that while there is a potential to increase the share of area with higher water levels compared to the baseline in the short term, in the long term it is only possible to maintain the current conditions. Prospects for restoration are therefore likely to be severely limited by water availability.

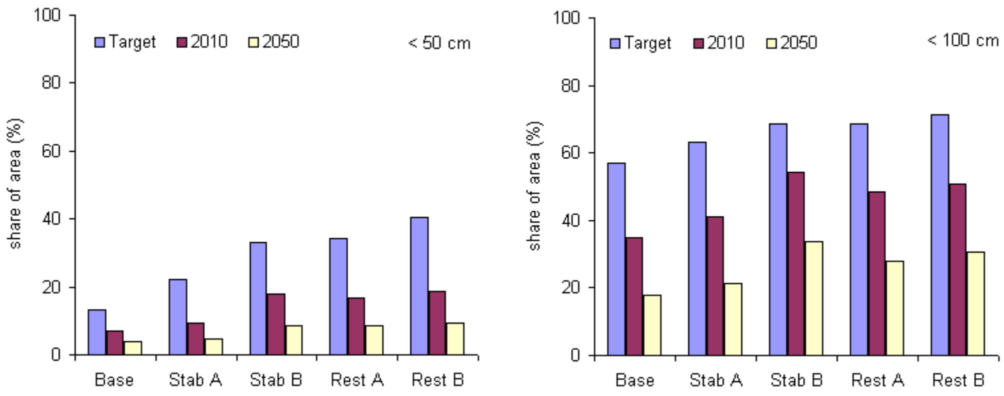

Fig 3: Share of the total wetland area with water levels below ground above a threshold of 50 cm (left) and 100 cm (right) in July for the target and mean conditions of the simulation period 2008/12 and 20048/52 for different water management options.

5.4 Effects of climate change and water management on greenhouse gas emissions

The estimates of GHG emissions for the management options for the first and the last simulation period in terms of the expectation value of emissions per unit area and in total are summarised in Table 7. We estimate that mean emissions in the baseline option are in a range from 17.5 tCO2e ha^{-1} to 25.7 tCO2e ha^{-1} for the low and higher impact trajectories[3]. We show that emissions for the baseline option increase by 0.5 – 1.4 tCO2e ha^{-1} from the first to the last simulation period as a result of climatic change. This is an increase in the range of 2 - 5%.

The less ambitious stabilisation option (Stab A) reduces the mean annual GHG emissions under current conditions by 0.9- 2 tCO2e ha^{-1}. The emission reductions by the more ambitious stabilisation option (Stab B) of 3 - 6.1 tCO2e ha^{-1} are in a similar range to those attainable from the less ambitious restoration option (Rest A) of 3.1 - 6.3 tCO2e ha^{-1}. The reductions are slightly higher for the more ambitious restoration option (Rest B) of 3.9 - 7.8 tCO2e ha^{-1}. However, the initial reduction of emissions induced by management measures is partly compensated by increases in emissions due to reduced water availability towards the end of the fifty year simulation period. The avoided emissions are approx. 50% lower towards the end of the simulation period. For example, emissions in the less ambitious restoration scenario (Rest A) increase from 14.4 – 19.4 to 16.1 - 23.2 tCO2e ha^{-1}, which is almost as high again as the initial emissions of 17.5 - 25.7 tCO2e ha^{-1}. The mean emission reductions of the Stab A, Rest A and Rest B options are similar in the

[3] In all following sections the reported ranges refer to low and high impact trajectories.

last period, even though the reduction in the initial period is largest for the more ambitious restoration option (Rest B).

Table 7: Greenhouse gas emissions and avoided emissions from the case study fen wetlands for different management options and for current and future climatic conditions.

Management option	Emission function	Simulation period 2003/2007		Simulation period 2048/2053	
		Mean [a]	Total [b]	Mean	Total
		tCO_2e ha-1 a-1	Mio tCO_2e	tCO_2e ha^{-1} a^{-1}	Mio tCO_2e
Emissions					
Base	High	25.7	0.914	27.1	0.975
	Low	17.5	0.626	18.0	0.650
Avoided emissions compared to baseline					
Stab A	High	2.0	0.056	1.1	0.029
	Low	0.9	0.025	0.4	0.013
Stab B	High	6.1	0.215	3.9	0.131
	Low	3.0	0.110	1.8	0.064
Rest A	High	6.3	0.235	3.9	0.141
	Low	3.1	0.120	1.9	0.069
Rest B	High	7.8	0.261	4.8	0.153
	Low	3.9	0.134	2.3	0.075

[a] mean of the expectation value for the fen soil area for each of 35 wetlands measured in GWP_{100} CO_2e

[b] for the calculation of the aggregate value the total fen area is 132,674 ha.

5.5 Social benefits from greenhouse gas emission reductions

The reductions in emissions attainable under the different management options in comparison to the baseline are valued using the marginal abatement cost approach to derive an estimate of the carbon co-benefit of wetland restoration. Table 8 presents the calculated present value for the central, high and low price schedules using a discount rate of 3% over the 50 year simulation period. The present value is converted to an average annual value (annuity) per unit area to facilitate comparison.

The mean annual benefit from GHG abatement of the less ambitious stabilisation option is in the range of 44 - 98 € ha^{-1} a^{-1} and for the more ambitious stabilisation option in the range of 163-330 € ha^{-1} a^{-1}. For the restoration options these are in the range of 167 - 337 € ha^{-1} a^{-1} and 208 - 415 € ha^{-1} a^{-1}. It is worth noting, that the uncertainty range stemming from the +/- 50% range on the central estimate of the shadow price of carbon is about twice as large compared to the uncertainty range resulting from the high and low trajectories of the GHG emissions.

Table 8: Average annual value of the avoided greenhouse gas emissions from the case study fen wetlands for different management options.

Management option	Emission function	Mean average annual carbon co-benefit (€2008 ha-1 a-1) a		
		Central	- 50 %	+ 50 %
Stab A	High	98	51	137
	low	44	23	61
Stab B	High	330	171	466
	low	163	85	231
Rest A	High	337	175	476
	low	167	87	236
Rest B	High	415	216	586
	low	208	108	293

[a] mean of the expectation value for the fen soil area of each of 35 wetlands. For the calculation of an aggregate value, the total fen area is 132,674 ha. Estimates are based on a GWP_{100}, a 50 years appraisal period and a 3% discount rate.

5.6 Abatement costs for wetland restoration measures.

We estimated the abatement costs for the different management options for a 25 year and a 50 year appraisal period. We present the mean and median of the estimates across the 35 sites (Table 9). Generally, the median provides a better estimate of central tendency, because the mean is strongly influenced by few sites with a very low cost-efficiency.

Table 9: Greenhouse gas abatement costs for wetland restoration scenarios for a 25 and 50 year appraisal period.

Management option	Emission function	Greenhouse gas abatement costs (€$_{2008}$ $tCO2e^{-1}$)			
		Appraisal period / discount rate			
		25 / 3%		50 / 3%	
		Mean	Median	Mean	Median
Stab A	High	17	10	22	10
	Low	39	19	67	20
Stab B	High	14	11	16	12
	Low	29	21	33	22
Rest A	High	11	8	11	7
	Low	23	14	23	13
Rest B	High	16	12	15	10
	Low	31	21	31	19

[a] mean and median of the expectation value for the fen soil area of each of 35 wetlands.

The median abatement costs over all options are in a range of 7 - 22 € $tCO2e^{-1}$. The abatement costs are highest for the more ambitious stabilisation option (12 - 22 € $tCO2e^{-1}$) and lowest for the less ambitious restoration option (7 - 13 € $tCO2e^{-1}$). The abatement costs for the more ambitious restoration option are higher, mainly because the restoration effort rises whilst the attainable reductions do not increase due to a lack of water. For the stabilisation options the abatement costs are slightly higher for a longer appraisal period due to falling emission reductions caused by an increase in water shortage. Whilst emissions also increase over time in the restoration option, the costs fall because the large initial investments are spread over a longer appraisal period.

6 Discussion

The inclusion of wetland restoration as an abatement measure in a climate policy framework requires GHG emission baselines to be specified and reductions to be amenable to monitoring, reporting and verification. As GHG emissions are difficult to measure directly, indirect methods based on proxies need to be used. Water levels are one of the possible proxies for measuring and monitoring peat GHG emissions (Joosten and Couwenberg, 2009). This paper has demonstrated how an approach based on water level proxies can be implemented into a water resources modelling framework and can be used for ex-ante spatial and temporal assessments of GHG mitigation potentials at a landscape scale.

We find that our landscape level estimates of current emissions in the baseline lie within the range of previous estimates from the literature that was reviewed above. We show that without any further action, decreasing water availability resulting from climatic change can lead to a 2 - 5% increase of emissions over the next 50 years. However, we estimate lower emission reduction potential associated with fen wetland restoration. One of the key differences between the reviewed studies and our estimate is that the first are based on scenarios assuming a complete restoration and unlimited water availability. In contrast, our estimates are based on more realistic management scenarios for large fen wetlands sites that take both zoning of land use according to topography and basin water availability into account. While the emission reduction for some of the sub-areas of our model may be as high as the maximum potential reduction assumed in the reviewed studies, the estimate for each aggregate wetland sites is based on the water availability for the whole wetland. The resultant estimates therefore describe, under more realistic management scenarios, the emission reduction potential of large lowland wetlands that are dependent on the inflow of river water from the upstream basin.

Our abatement cost estimate is in a similar order of magnitude as the abatement costs for restoration projects in Mecklenburg-Vorpommern of 7.5 - 12.5 $tCO2e^{-1}$ estimated by Schäfer (2009). We find that the abatement costs are within a range of 10 - 20 € $tCO2e^{-1}$ for the stabilisation scenarios and, with a range of 7 - 14 € $tCO2e^{-1}$, are lower for the restoration scenarios. However, we are able to show that water availability is a key factor determining the effectiveness of wetland restoration measures– in the short and the long term. With increasing wetland area targeted for restoration the abatement costs increase, because the limited water availability does not allow maintaining high water tables over larger areas. We also show that the effects of reduced water availability as a consequence of climatic change may reduce the effectiveness of wetland restoration measures regarding the avoided emissions by roughly 50% in the long term.

The findings have several policy implications. First, our findings confirm that wetland restoration can be a relatively low cost option for GHG abatement. The abatement cost estimates we provide indicate that many projects can reduce emissions at costs less than the projected market price of traded carbon in a range of 18 - 37 € $tCO2e^{-1}$ (cf. Table 2). Therefore there is potential to activate resources allocated for GHG abatement towards the restoration of wetlands. For example emission reductions from restoration of peatlands can be rewarded with carbon credits in a certification process. This confers a market value on the certified emission reductions. Our estimate of abatement costs indicates that a price for carbon credits above 7 - 20 € $tCO2e^{-1}$ could be sufficient to refinance a majority of fen wetland restoration projects.

Second, the case study shows that the benefits from a single ecosystem service - in this case GHG abatement - may be large enough to justify wetland restoration. However wetlands provide a multitude of ecosystem services. Wetland restoration projects should therefore be justified and assessed based on their impact on the whole range of potential benefits - such as biodiversity conservation or landscape amenity for recreation - that are not considered here. The consideration of further co-benefits would reduce the abatement cost estimate.

Third, we find that an approach centred on fen restoration is a slightly more efficient abatement strategy in comparison to an approach centred on fen stabilisation and extensive agricultural production. This implies that abatement policy efforts should prioritise the support to permanent rewetting of wetlands and put lower priority on supporting continued, albeit extensive, grassland management practices. However both approaches are required, for example for the implementation of zoning concepts within the landscape where only parts of a wetland site can be rewetted.

Fourth, water availability is a key limiting factor that may lead to decreasing effectiveness with increasing size of the restoration effort for a wetland site. This implies that water availability needs to be factored into the design of any wetland restoration strategy. This may also require according a higher priority to water allocated to wetlands from the available river water. From both a hydrological and an economic point of view, wetlands must be understood as multi-functional water users that compete with other water users upstream and downstream for sufficient water supplies.

Finally, this paper has presented a practical method for the economic assessment of the benefits of increased GHG abatement from changes in water allocation or wetland water and land management. This enhances the systematic inclusion of wetland benefits in the cost – benefit analysis of water management options. The value of ecosystem service benefits are most often not included in such analysis because of the lack of appropriate assessment methods. The application of the shadow price of carbon approach would be greatly enhanced, if more scientifically sound, regularly updated and binding guidelines on the appropriate shadow price of carbon for Germany for use in cost-benefit analysis would be available (cf. UBA 2007). The water level proxy based approach to estimate changes in GHG emissions can easily be adapted as improved calibration of the proxy becomes available.

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PAPER V

ECONOMIC VALUE OF THE NUTRIENT RETENTION FUNCTION OF RESTORED FLOODPLAIN WETLANDS IN THE ELBE RIVER BASIN

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This paper presents an application of an indirect method, the alternative or replacement cost method to value a regulatory ecosystem service: the retention of river nutrient loads by floodplain wetlands. The paper presents a cost-minimisation model for nutrient abatement measures for the River Elbe. The model is applied to estimate the shadow price of phosphate and nitrogen nutrient retention services by restored floodplains. It is shown that the shadow price of restored floodplain area is a function of the nutrient load reduction target for the river basin. The scope of the floodplain restoration projects are shown to have a lesser impact on the estimated shadow price.

In addition, this paper presents an empirical cost function for the costs of floodplain restoration measures in the Elbe Basin. In conjunction with the shadow prices, this allows for a rapid strategic assessment of the costs and benefits of 45 potential restoration sites along the Elbe trajectory. In spite of the large investment costs for dike realignments, the nutrient retention effects alone may in many cases generate sufficient benefits to generate an economic efficiency gain. Floodplain restoration may therefore, under advantageous circumstances, constitute a cost effective nutrient abatement measure. However the key thrust of the argument is that floodplain restoration projects have to be assessed as multifunctional projects, with the positive impacts on water quality being one of several benefit dimensions.

Keywords: ecosystem services, economic valuation, floodplain wetland restoration, cost minimisation model, nutrient management

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1 Introduction

This paper addresses the issue of valuation of the nutrient retention function associated with a restoration of floodplain wetlands. The retention of nutrients is one of the many ecosystem services provided by floodplain wetlands subject to regular inundation. Through ecosystem processes such as denitrification and sedimentation nutrients are removed from the river water. The removal of nutrients contributes to the provision of clean water that in turn has social benefits, for example in terms of enhanced recreational benefits. However, many rivers have been disconnected from their historic floodplains by dikes and the floodplains no longer contribute to nutrient retention. The recognition of the potentially large benefits associated with the provision of ecosystem services by active floodplains has in many countries led to a revision of floodplain management policies: away from a historical focus on gaining and securing agricultural land through the construction of dikes and towards a gradual process of floodplain restoration, for example by dike realignments.

The concept of ecosystem services has become an important model to systematically link functions of ecosystems to human welfare. It is an important concept for the valuation of negative and positive external environmental effects associated with changes of land use in floodplains (Turner et al. 2008). It is well acknowledged in the environmental economics literature that the public goods characteristic of many ecosystem services has traditionally caused many ecosystem services to be undervalued in the assessment of floodplain management options. Sectoral appraisal of floodplain restoration measures, for example in cost effectiveness analysis of the flood risk reduction or nutrient abatement potential, tend to omit the co-benefits from multiple ecosystem services. This is largely due to lacking information on the value of such benefits. As a result neither flood risk nor nutrient management plans tend to consider floodplain restoration as economically advantageous options, because the unit costs of restoration are generally high and the benefits with regard to a single target dimension comparatively low. However, both full cost benefit analysis and correctly specified cost-effectiveness analysis of floodplain restoration measures need to take co-benefits into account. Both types of analysis require information on the value of ecosystem service benefits provided by floodplains. This paper addresses the valuation of one such benefit: nutrient retention.

The evidence regarding the contribution of floodplain wetlands to nutrient retention is limited but field studies of sedimentation and denitrification on floodplains indicate considerable potential retention of nitrogen and phosphorous. To assess the importance of sediment deposition and denitrification in floodplains for nutrient retention, annual nutrient fluxes out of the river water by these processes have to be compared with the annual loads of nutrients that are transported in the river. In a study for the lower Rhine, van der Lee et al. (2004) scaled up the results from measurements on inundated floodplain sites to the entire river stretches of the Waal and the Ijssel and compared the resultant fluxes with annual nutrient loads in the river. They concluded that N-retention was low (less than 3% of annual load), whereas P-retention was significant (5 - 18% of annual load in Waal and Ijssel). Walling and Owens (2003) calculate annual conveyance losses from overbank events for total phosphorous in floodplains bordering the main channel of the rivers Swale and Aire as 14 and 9 %. Other authors have compared nutrient loads and fluxes not on an annual, but an event basis. For example Engelhardt

(1999) report retention of 50 % of particulate P and 16 % of total N load during flood events for controlled polders on the Odra River. Kronvang et al. (2007) estimate a storage efficiency for total phosphorous during overbank events of 4 -7 % for the River Gjern. Brunet and Brian Astin (1998) report retention rates of particulate P during flood events on the River Adour of 26 – 28 %.

Several authors have attempted to estimate the economic value of the nutrient retention ecosystem services of restored floodplain wetlands. All of these studies use the replacement or alternative cost method, whereby the value of the retention capacity is estimated from the savings of abatement costs from alternative measures to achieve nutrient reduction targets for the basin. The most common alternative measures are either some form of improved waste water treatment or measures to reduce nutrient emissions from agricultural production. The replacement value is based on estimates of the marginal abatement costs of these measures that are generated either with micro-engineering approaches (Gren et al. 1995 for the Danube River floodplains, Dubgaard et al. 2005 for Skjern River, Meyerhoff and Dehnhardt 2007 for the Elbe River) or with cost minimisation models (Jenkins et al. 2010 for the Mississippi River).

Several other studies based on spatial cost minimisation models have attempted to quantify the possible costs savings fromincluding wetland restoration in programmes of measures to reduce nutrient loads. Such programmes to date often focus on measures that reduce emissions, like improved waste water treatment, and less on measures that improve nutrient retention. For example Bystrom (2000) and Ribaudo et al. (2001) use abatement cost models to investigate the savings in total abatement costs that are made possible by investing in wetland construction or restoration projects. Other major abatement cost model studies in the European context, such as Gren et al. (1997) or Schou et al. (2006) for the Baltic Sea drainage basin, Lise and van der Veeren (2002) or van der Veeren and Tol (2001) for the Rhine include wetland measures as cost effective abatement options. All of these studies argue that wetland construction and restoration, depending on the contextual conditions, can be a cost effective abatement measure and that there are opportunity costs associated with omitting wetland restoration from the catalogue of abatement options.

The present paper attempts to value the benefit of the nutrient retention ecosystem services that are reactivated in the course of wetland restoration. In a first step, we present a novel cost minimisation model for nutrient abatement measures for the River Elbe. We apply the model to generate estimates of the value or shadow price of the nutrient retention services provide by floodplain restoration. None of the previous cost minimisation approaches have addressed floodplain restoration, but have focused on constructed or pond like wetlands (cf. Byström 2000). We not only consider nitrogen but also phosphorous retention.

In a second step, using the estimated shadow prices, we conduct a strategic cost - benefit analysis to assess floodplain restoration along the Elbe River. In order to be able to carry out this assessment, this paper also presents an empirical cost function that can be used to estimate the costs of floodplain restoration in the Elbe Basin. Such functions have been presented for constructed wetlands for example by Söderquist (2002) or Byström (1998) but not for floodplain restoration.

The rest of the paper is organised as follows: after an introduction to the study site, we present the methodological approach. The focus is on the replacement cost method, the cost-minimisation modelling approach, the method to estimate the nutrient retention capacity and the empirical estimate of the costs of floodplain restoration measures. We proceed to present selected results for the shadow price of the nutrient retention capacity of restored floodplains and close with a discussion of the implications for the cost-benefit analysis of wetland restoration and the valuation of ecosystem services.

2 Study area and management scenarios

The German part of the Elbe River catchment covers an area of 97175 km^2 and has 18.5 million inhabitants. The Elbe River has characteristics of a lowland river with a wide alluvial valley downstream of Dresden. Approximately 84% of the floodplains along this river stretch are protected by dikes. The loss of active floodplains in the Upper and Middle Elbe differs according to the width of the alluvial valley. The narrow valley of the southern section generally has lower losses of active floodplain. After entry to the wider lowland valley, 50 - 90 % of the floodplains have been diked (Brunotte et al. 2009).

Despite these large reductions in active floodplain area, the Elbe still has a very long free flowing river stretch. The designation of large parts of the remaining active floodplains as a UNESCO Biosphere Reserve and other categories of protected areas highlights the importance these habitats have been accorded for habitat and biodiversity protection in Germany. An early analysis of potential dike relocation sites for large scale conservation programs was conducted by Neuschulz and Purps (2000). The public debate on dike realignments gained momentum in the aftermath of the major flood on the Elbe in 2002. The International Commission for the Protection of the Elbe commissioned an action plan (IKSE, 2004), whose purpose was to develop a comprehensive flood risk management strategy for the river. The proposed measures include amongst others, reactivation of the retention capacity along the river floodplains and reconstruction of dikes to the desired safety standard. Several federal state governments have since commissioned detailed studies to evaluate potential sites for dike relocations and retention polders. The number, exact location, area and retention volume of potential sites is the subject of public debate and constant review.

The Ministry of Environment (BMU) and the Federal Agency for Nature Protection (BfN) actively promote the concept of an integrated approach to management and development of floodplains (BMU/BfN 2009). Such an approach seeks to harness multiple benefits for flood protection, water resource management, nature and biodiversity conservation and climate change mitigation. The strategy is based on the three principles of strict protection of remaining natural floodplain habitats, restoration of modified floodplains in agricultural use that are still subject to regular flooding and increased efforts to regain historic floodplains by dike realignments where feasible.

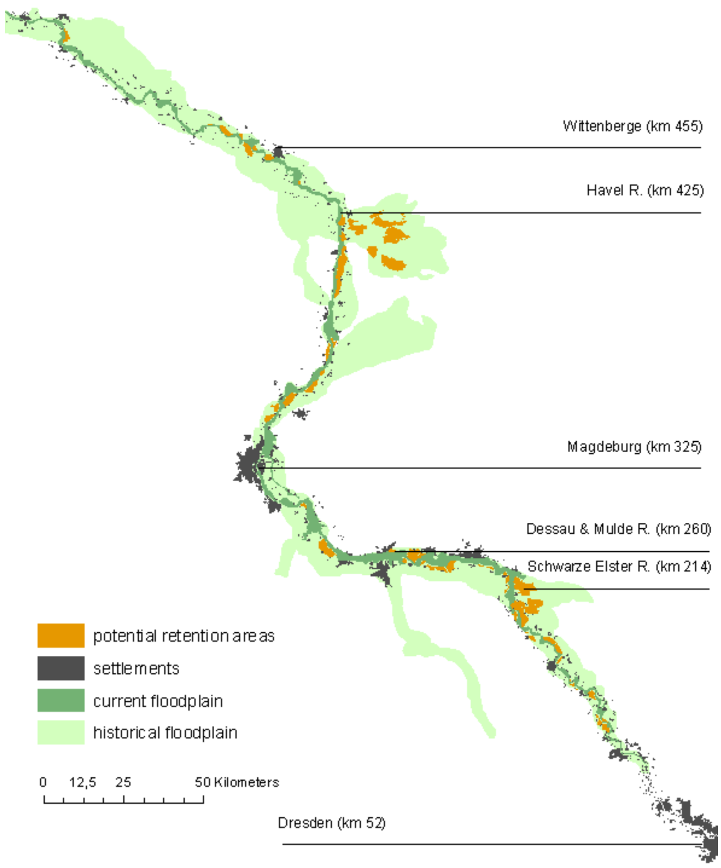

Figure 1: Map of the study area showing the location of the potential retention areas and the extent of the historic floodplain

Nutrient management goals for the Elbe River Basin are set out under the management plan developed in compliance with the EU Water Framework Directive (FGG Elbe 2009). The reduction targets are deemed necessary to comply with general target of achieving a good ecological status of the river and coastal seas. The current goal is to reduce both phosphorous and nitrogen load to the sea by 24 % by the year 2027. This is to be achieved in a stepwise approach, with a third of the commitment to be achieved over three reporting periods ending 2014, 2021, 2027. The Federal Agency for Nature Conservation

(BfN) has commissioned studies to explore potential overlaps and synergies of conservation oriented floodplain management with water resource management as mandated by the European Union Water Framework Directive (cf. Korn et al. 2006). Whilst river management plans have taken up the issue of restoration of river morphology including the restoration of the natural floodplains on minor rivers, major floodplain restorations on the main river trajectory, as discussed in this paper, have not featured in river basin management plans to date.

For this assessment, the proposed sites and dimensions from several data sources were combined to generate a large scale floodplain restoration scenario. A more detailed description of the sites can be found in Grossmann et al. 2010. The location of sites is shown in Figure 1. Table 1 provides a summary of key characteristics of the sites included in this study. The total area (A_{tot}) of the 60 potential restoration sites is 20749 ha, with an average annual inundated area ($E(A_{in})$) of 1717 ha. This difference is due to the different topography of the sites that will lead to different inundation frequencies. The mean ratio of total area to average inundated area for the proposed project sites is 1:13.

Table 1: Summary of key characteristics of the potential restoration sites

	Total area	Average annual inundated area	Ratio
	A_{tot}	$E(A_{in})$	$A_{tot} / E(A_{in})$
	Ha	ha	(-)
Mean[(a)]	482	40	13
Median	350	15	10
Total	20749	1717	0,08

(a) N = 60 sites

3 Methods and Data

3.1 Valuation concept

Floodplains are ecosystems that provide a number of market and non-market benefits. The net benefit or net present value of implementing a floodplain restoration programme can be written as:

$$NPV_t = \sum_{t=0}^{T} \frac{1}{(1+r)^t} \left[(NRB_t + oESB_t - PC_t - OC_t \right] \tag{1}$$

where NPV is the net present value of a floodplain management option in comparison to the baseline, PC are the project investment, maintenance and operation costs, OC are the opportunity costs in terms of the loss in economic rent from the initial use of floodplains (mainly loss of benefits from agricultural and forestry land use), NRB are the benefits from nutrient retention and oESB are all other ecosystem service benefits, for example from habitat and biodiversity conservation or reduced flood risk.

The paper presents approaches to estimate both the benefits from nutrient retention (NRB) and costs of restoration (PC and OC). The value of the nutrient retention services are estimated based on the replacement or alternative cost method. The alternative cost values are estimated as the shadow price of floodplain restoration measures in a cost-minimisation model of measures to reduced nutrient loads in the Elbe River basin. The costs are estimated from an analysis of the costs for floodplain restoration projects along the Elbe River. Finally, the resulting shadow prices estimated by the cost-minimisation model (NRB) and the cost estimates (PC and OC) are combined in the cost-benefit analytical framework described by equation (1). This allows for a rapid strategic assessment of the costs and benefits of 45 potential restoration sites along the Elbe trajectory. The analysis is only partial, because we only focus on a single benefit of floodplain restoration – nutrient retention – leaving out other co-benefits (oESB) such as flood risk reduction.

3.2 Valuing ecosystem services using the replacement cost approach

In terms of the ecosystem services approach, nutrient retention is an intermediate service (Fisher et al. 2009)[1]. In the case of nutrient retention the final ecosystem service is the provision of clean water, which in turn gives rise to various benefits, such as improved recreational opportunities or clean drinking water. The direct estimation of benefits as a function of restored floodplain area would require estimates of the demand for clean water, where demand is a function of the water quality and quality is a function of the floodplain restoration.

Lacking the extensive data required for a direct approach, the nutrient retention service can be valued using an indirect method for benefit estimation: the replacement or alternative cost method. Replacement cost values do not constitute a direct estimate of the benefit from the ecosystem service to society (i.e. the value of clean water); they represent the value of having the ability to provide the benefit in demand through an ecosystem service rather than through an alternative method. Shabman and Batie (1978) were the first to suggest that this method can be used for an indirect valuation of ecological services if the following conditions are met: (1) the alternatives considered can provide similar services as the natural wetland (2) the alternative used for cost comparison should be the least cost alternative (3) there should be substantial evidence

[1] Nutrient retention can therefore also be considered to constitute an indirect use value.

that the service would be demanded by society if it were provided by that least cost alternative.

Nutrient retention by restored floodplains can in principle substitute for other measures to reduce nutrient loads in a river basin. The cost savings compared to a least-cost alternative combination of alternative measures (such as waste water treatment or land management) required to reach a reduction target are then the alternative or replacement costs. The validity of the alternative cost approach in this context hinges on the implicit assumption that the water quality targets set out under public environmental policies reflect the underlying water quality preferences (aggregated demand) of the general populace. In this case the marginal benefit from a policy to achieve the targets will be equal to the marginal costs of the required measures. Under this assumed equality, the replacement cost approach allows one to value the marginal benefit from nutrient retention using the marginal costs of nutrient abatement.

In this paper we estimate the replacement costs based on the shadow price of floodplain nutrient retention measures in a spatial cost minimisation model framework (cf. Samuelson 1952) for measures to reduce nutrient loads.

3.3 Estimation of the shadow price of floodplain retention with an integrated economic-ecohydrological model.

The setup of the cost-minimisation model for the Elbe River considers 950 sub-basins and 2105 individual waste water treatment plants. The number of decision variables in the model is the product of measures and corresponding spatial model units – sub-basins or waste water treatment plants. The types of measures available for each spatial unit are summarised in Table 2. The measures represent all major types of intervention to reduce nutrient emissions and improve nutrient retention. They cover improved sewage treatment in central waste water treatment plants, improved waste water treatment for population not connected to central treatment plants, storm water treatment, agricultural water management, agricultural land management. Finally, also measures to restore wetlands and in particular floodplain wetlands are included. All measures are characterised by the maximum number of units that can be implemented in each spatial modelling unit. These constitute the constraints to the cost-minimisation model. These limits can be explained by the fact that each measure has a limited feasibility range within each drainage basin. Further, the implementation range has to ensure that the estimates of costs and emission reductions remain coherent with the assumptions on prices and technologies.

In a constrained cost minimisation model, the shadow price is the change in the objective value of the cost minimal solution of an minimisation problem obtained by a change in the constraint by one unit – it is therefore the marginal cost savings from relaxing the constraint, or equivalently the marginal additional cost of strengthening the constraint.

The floodplain restoration measures described above are included in the model. The 45 restoration sites are aggregated to 16 based on their location within the spatial sub-units of the model. For the analysis presented in this paper, we set the costs of these measures at 0 €/ha, because we want the shadow price to reflect the marginal cost savings (shadow price) induced by restoring one further unit of floodplain area, independent of the cost of the measure[2]. The shadow price in this case therefore reflects the marginal cost savings (or replacement value) from the nutrient retention co-benefit induced by floodplain restoration projects.

As the shadow price can be expected to decrease with increasing implementation of floodplain restoration, we estimate the shadow price for two scenarios. In the scenario without the implementation of the floodplain restoration programme, the constraint on the maximum floodplain area available is set at 0 ha. The model then yields the shadow price of implementing the first unit area into the model. The second scenario explores the shadow price with the implementation of the floodplain restoration programme. In this case the constraint is set at 1718 ha. The shadow price now reflects the marginal cost savings that could be induced by extending each of the sites by a further unit area.

The cost minimisation problem is formulated as a choice of the cost minimising mix of policy measures within drainage basins (cf. Bystrom 2000 or Schou et al. 2006 for comparable applications). The cost minimising mix consists of an optimal mix of measures and an optimal allocation of the measures so that the specified goal for reductions of loads at the outlet or at specific sub-basin is obtained for the least total abatement cost. As explained, the minimisation problem is constrained by the exogenously set potential of each measure.

The cost minimisation problem is described by:

$$\min_{x_{i,m}} \sum_i \sum_m c_{i,m}(x_{i,m}) \tag{2}$$

s.t.

$$\sum_i \sum_m q_p(x_{i,m})\Omega_{i,s} = TR_{s,p} \tag{3}$$

$$x_{i,m} \leq x'_{i,m} \tag{4}$$

[2] Note that this procedure is different to the approach presented by Bystrom (2000), who estimate the marginal abatement cost savings of wetland measures that are included at their cost.

where indexes denote

i spatial model units (sub-basins and waste water treatment plants)

m type of measure

p pollutant / nutrient (phosphorous and nitrogen)

s location for nutrient reduction goals (river outlet, upstream location)

and

x is the implementation level of a measure

c is a function describing the total costs of implementing a measure

q is a function describing the reduction of nutrients emitted by implementing a measure

$\Omega_{i,s,p}$ is the retention coefficient describing the share of nutrient p emitted from a basin i reaching the sea

TR is the targeted load reduction

x' is the upper limit to the possible implementation level of a measure in a basin

In the model, the functions q is linear in x saying that there is a positive linear relationship between the extent of a measure in a basin and the reduction of nutrient loads reaching the sea. Therefore the first derivates for q exists and is continuous. For a solution to the problem the necessary conditions for optimality are the Kuhn-Tucker conditions that are given in (5) – (9). To obtain optimal solutions the general micro-economic theory on cost functions is applied, whereby all separate cost functions are convex and total abatement costs (objective function) are quasi-convex. The first order conditions for cost minimisation can then be written as:

$$\frac{\partial C_{i,m}(x_{i,m}*)}{\partial x_{i,m}} - \sum_{s,p}\left(\mu_{s,p} \cdot \Omega_{i,s,p} \cdot \frac{\partial q_p(x_{i,m}*)}{\partial x_{i,m}}\right) - \lambda_{i,m} = 0 \tag{5}$$

$$\sum_i \sum_m q_p(x_{i,m}*)\Omega_{i,s,p} = TR_{s,p} \tag{6}$$

$$x_{i,m}* \leq x'_{i,m} \tag{7}$$

$$\lambda_{i,m}(x_{i,m}*) = 0 \tag{8}$$

$$\lambda_{i,m} \geq 0 \tag{9}$$

The condition in (5) ensures optimality and (6) and (7) are feasibility conditions, (8) is the complementary slackness condition and (9) is a non-negativity condition. According to economic theory, the Lagrange multiplier μ_s can then be interpreted as the marginal cost of increasing the nutrient reduction target by one unit and the Lagrange multiplier $\lambda_{i,m}$ as the shadow price of the constraints on the capacities of each of the measures. The Lagrangian multiplier $\lambda_{i,m}$ for the floodplain restoration measure (m = floodplain restoration) and for each of the 16 aggregated sites (i) can therefore be interpreted as the shadow price of an additional unit of floodplain area, which is identical to the marginal change in total abatement costs induced by a marginal change in floodplain area.

The model is implemented using GAMS and is solved using the solver CONOPT. The data for the model parameterisation, specifically the river retention coefficients ($\Omega_{i,s}$), the upper limit to the possible implementation level of a measure in a basin (x'), the reduction of nutrients emitted by implementing a measure and the costs for implementing a measure are directly extracted and imported from an existing simulation model for the Elbe River. This model system (MONERIS - Modelling Nutrient Emissions in River Systems) and its applications are described in detail in Behrendt et al. (2003 and 2005) and Venohr et al. (2009 and 2010). The model has been widely used to estimate nutrient emissions, nutrient retention in the river system and resultant nutrient loads to the sea. In a recent version of the model abatement measures have been included. The set of abatement measures that were included in the cost minimisation application presented here were selected from the set of available measures so that (a) the marginal costs for all measures are constant, i.e. the total abatement costs are a linear function of the abatement level and (b) that the effects of the measures are independent. Increasing marginal costs are instead captured by two mechanisms: a very high spatial disaggregation and differentiation of the effectiveness of measures (by waste water treatment plant and sub-basin) and a stepwise segmentation of cost functions with increasing marginal costs in particular for measures to reduce nitrogen surplus in agriculture. Measures that are interdependent would change the effectiveness of other measures as their implementation level increases, and their inclusion would increase the model complexity. For example, increased erosion control through tillage would reduce the benefits of buffer strips, because less sediment would be eroded and therefore retained by the strips. The selected types of measures and the assumptions regarding their costs are summarised in Table 2. The abatement potential of each measure is imported directly from the analysis of these measures in the MONERIS model.

Table 2: Measures included in the model

Measure	Spatial modelling unit	Price: unit	Price: value
Improvement of sewage treatment in central waste water treatment plants			
N-Elimination: optimisation of denitrification (by size class 6- 1 **)	WWTP	€ / kg TN	2 / 2.5 / 3 / 6 / 9 / 14
P Elimination: simultaneous precipitation (chemical treatment) (by size class 6- 1*)	WWTP	€ / kg TP	3.3 / 3.6 / 4/ 6.4 / 10.8 / 25.7
P Elimination: optimisation of simultaneous precipitation	WWTP	€ / kg TP	2.3
P Elimination: additional filtration	WWTP	€ / kg TP	135
P Elimination: membrane technology	WWTP	€ / kg TP	1000
Wastewater treatment for population not connected to central waste water treatment plants			
Upgrading and construction of small waste water treatment plants < 50 PE	Basin	€ / person	85
Construction of waste water treatment plants (< 2 000 PE)	Basin	€ / person	58
Connection to sewage system	Basin	€ / person	125
Storm water treatment			
Increase of the urban area with storm water overflow tanks in combined sewage systems installed	Basin	€ / ha	1188
Increase of the urban area with storm water sedimentation tanks and soil filters installed	Basin	€ / ha	2190
Agricultural water management			
Tile drained land: construction of wetlands in outlet of drainage systems	Basin	€ / ha	500
Agricultural land management			
Erosion control measures for arable land with slope > 4 %	Basin	€ / ha	50
Reduction of agricultural nitrogen surplus (by reduction level) ***	Basin	€ / kg	1.7 / 6.6 / 13.7 / 23
Wetland restoration			
Floodplain restoration	Basin	€ / ha	0 ****

* the nutrient reduction capacities and effectiveness of measures regarding is calculated using the MONERIS method and model (Venohr et al. 2009) for the specific conditions of each basin and WWTP and cannot be presented here.

** size classes: 1 = < 1000; 2 = 1000 –5000; 3 = 5000 – 10 000; 4 = 10 000 – 50 000; 5 = 50 000 – 100 000; 6 = > 100 000 connected person equivalents

*** marginal costs for reduction of N surplus by 0-10, 10 - 20, 20 - 30 or 30 - 40 kg N ha.

**** for this analysis, the costs are set at 0 €/ha in order to estimate replacement value of the nutrient retention function

3.4 Estimation of the nutrient retention capacity of floodplain wetlands

The nutrient retention by floodplain inundation have been studied at various river sites, however robust assessment methods for the landscape or watershed scale are not available. In order to capture the future nutrient retention capacity of restored floodplain wetlands, we estimate the total retention of a site as the product of the additional average annual inundated area and the specific nutrient retention rate per inundation day. A comparable approach based on daily retention rates and inundation modelling is used for example by van der Lee et al. (2004) to estimate nutrient retention of the Rhine floodplains.

Estimation of the inundated area

We estimate the average annual inundated area for each of the project sites. Discharge-exceedance and discharge-stage functions for each river section are available from a study of the morphodynamics of the Elbe River (cf. Nestman and Büchele 2002). We use the discharge exceedance curve for each site along the river trajectory and divide these into ten classes of equal duration. We then interpolate the corresponding stage for each of the eleven breakpoints (Qmin, Q10,..., Q90, Qmax) from the discharge-stage function. The corresponding inundated area (A_{in}) for each stage is estimated by extrapolating the water level over a digital elevation model of the floodplain using GIS analysis. The expectation value or average annual inundated area $E(A_{in})$ for each project site is then calculated by sum of the product of inundated area and the corresponding occurrence probability to ensure normalisation as follows:

$$E(A_{in}) = 0.05*A_{in}(Q_{min}) + 0.1*A_{in}(Q_{10}) + .. + 0.1*A_{in}(Q_{90}) + 0.05*A_{in}(Q_{max}) \qquad (11)$$

Estimation of the specific nutrient retention capacity of lowland floodplains

In eutrophic rivers, a large part of the phosphorous load is adsorbed to silt particles (50-70 %) whereas the largest part of the nitrogen load is transported in solution (98 %) (Ventrink et al. 2003)[3]. At high stages in the river the floodplain will therefore be inundated with water having high concentrations of nitrate-N, sediments and particulate phosphorous.

Retention of phosphorous in floodplain wetlands is mainly controlled by sedimentation processes. Retention of dissolved P is not as pronounced as retention of particulate P. In contrast to the storage of fluvial sediments on inundated floodplains the evidence of P deposition is not as widely documented. Hoffmann et al. (2009) provide one of the most recent reviews of the phosphorus retention capacity of wetlands. Annual deposition rates of particulate P range from 1 to 128 kg P ha^{-1} a^{-1}. However desorption of high P concentrations from wetted soils previously under agricultural use may for some time

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[3] In the following P and N are used to denote phosphorous and nitrogen. TP and TN are used to denote total phosphorous and nitrogen.

lower the effectiveness of restoration measures. The observable net retention is attributed to the fact that the sedimentation of P is generally higher than the remobilisation of P from the soil matrix. For the approach selected for this study, estimates of deposition per inundation day are required. Table 3 provides a survey of estimates of particulate phosphorous deposition rates per inundation day based on field measurements on periodically inundated floodplains from European rivers. We find that values range between 0.2 – 3.5 kg TP ha^{-1} d^{-1}.

Table 3: Estimates of deposition of particulate phosphorous per inundation day based on field measurements on periodically inundated floodplains.

River	Country	Daily TP deposition	Duration of inundation	Annual TP deposition [c]	Reference
		kgTP ha^{-1} d^{-1}	d a^{-1}	kg TP $ha^{-1}a^{-1}$	
Maas	NL	3.3[a]	40[b]	130	Lee et al. 2004. Olde Ventrink et al. 2006
Issel	NL	2.5[a]	45[b]	114	
Adour	F	2.9	45	129	Brunet et al. 1998
Gjern	DK	3.5	17	58	Hoffmann et al. 2009
Gjern	DK	1.0	76	76	Kronvang et al. 2007
Brede	DK	0.5	35	17.5	
Odense	DK	1.2	20	24	
Skjern	DK	0.2	60	12	
Odra	DE	0.32*	60[b]	19.2	Engelhardt et al. 1999

a calculated from data provided
b estimate from provided description
c inundation days x daily deposition rate

The most important processes controlling nitrogen retention in floodplains is denitrification in the water column and alluvial soils (Pinay et al. 2002). The surface contact area of water with sediment as well as the duration of contact determines the rate of nitrogen retention, because the nitrogen cycle is driven by processes that occur on or at the interface of particulate matter. Denitrification in alluvial soils contributes to retention of the river nitrate load if nitrate originating from the river water is denitrified.

Behrendt and Opitz (2000) estimate an empirical relationship between the hydraulic load as an indicator for the intensity of water – surface contact and nutrient retention in large river systems[4]. The approach is implemented in the ecohydrologic model system MONERIS to quantify nutrient retention in river systems. We applied the MONERIS model version for the Elbe River basin (Venohr et al. 2010) to estimate the additional nitrogen retention resulting from an increase in the water surface area by the average

[4] This approach was also used for an assessment of the nutrient retention potential of two restoration sites along the Elbe by Meyerhoff and Dehnhardt (2007).

annual inundated area of all project sites at their respective location along the trajectory. For the discharge and nutrient loads of the period 2000-2005 this gives an average retention of 1.4 kg N ha^{-1} d^{-1}. This value compares well with other estimates from the literature. For example, Hoffmann & Baattrup-Pedersen (2007) give recommendations for estimating the additional nitrogen retention of restored floodplain wetlands with regular inundation in Denmark. For rivers with nitrogen concentrations > 5 mg N l^{-1} they propose a value of 1.5 kg TN ha^{-1} d^{-1} per inundation day and for nitrogen concentrations of < 5 mg N l^{-1} a value of 1 kg TN ha^{-1} d^{-1}. For the branches of the Rhine estuary, Lee et al. (2004) and Olde Ventrink et al. (2006) estimate retention rates per inundation day in the order of 0.1 – 0.35 kg TN ha^{-1} d^{-1} for the alluvial soils of the Maas and Issel floodplains. Engelhardt et al. (1999) observe retention rates of ca. 1.46 kg TN ha^{-1} d^{-1} for flood polders along the Odra River.

Based on an evaluation of this evidence, we use a retention rate of 0.8 kg TP ha^{-1} d^{-1} for phosphorus, that we consider to be a lower reliable estimate (0,33 percentile) of the sample. We use the recommended value of 1.5 kg TN ha^{-1} d^{-1} for nitrogen. Because of the uncertainty of the estimates we conduct sensitivity analysis for a 50 % lower nutrient retention capacity.

3.5 Empirical estimation of the costs of floodplain restoration

We estimate a simple cost function from the total investment costs reported for 27 floodplain restoration projects at various stages of preparation along the Elbe River (Table 4). The data refers to the period 2000-2005. The largest share of the investment costs are the construction costs for the relocated dike and the costs for acquiring the floodplain land from current land owners prior to conversion[5]. Other major cost elements are project planning and transaction costs.

Table 4: Data on floodplain restoration projects: summary statistics

Variable	Unit	Mean	Range
Total project cost	T. € ha^{-1}	34.8	3.2 - 93.3
Restored area	ha	342.5	30 – 860
Dike line construction	km	4.2	0.5 – 10
Ratio dike length to area	km ha^{-1}	0.018	0.001-0.039

N = 27

We estimate the total project investment costs as a function of the length of the constructed new dike and the restored floodplain area. We tested various specifications

[5] The opportunity costs of agriculture land restored to wetlands is roughly equal to the value of lost production in perpetuity. This can be considered to be roughly equivalent to the purchase price for land.

and find that the cost per unit area is best explained by the required length of new dike line per unit area. The costs per unit area decrease with increasing restored area in relation to the length of necessary dike construction. This suggests the following cost equation:

$$TC = \beta_0 + \beta_1 (L_{DIKE})^2 \tag{12}$$

where TC are the total costs in € ha^{-1} and L_{dike} is the length of required dike construction per total restored floodplain area in km ha^{-1} and ß are the coefficients to be estimated. Table 5 reports ordinary least square estimation results.

The function has an explanatory power of 81 %. The variable and constant are significant at the level < 0.001. We conclude that the cost function can be used for a rough estimation of the investment costs of lowland restoration measures at different sites within the Elbe basin.

Table 5: OLS estimation results for the cost equation

Coefficient	Estimate	SE	Sig.
ßo	15792.50	2664.6	0.000
ß1	45866633.6	4462642.1	0.000

Adjusted r^2 = 0.81 . n = 27

4 Results

4.1 Value of nutrient retention ecosystem service

We solve the model defined by equation (2-9) for various levels of abatement requirements defined in the constraint to the objective function and for two implementation levels (with and without) of the proposed floodplain restoration programme[6]. The average annual inundated area of the 20749 ha total area of the floodplain restoration programme is 1718 ha. We consider load reduction requirements that range from 5 - 35 % and differ according to the combination of the targeted nutrients and locality of the abatement requirement. This range encompasses the current policy target of a simultaneous 24 % reduction of the load of both nutrients by 2027. The

[6] In the without implementation case, the constraint ($X_{i,m}'$) on the available average annual inundated floodplain area is set at 0 ha, for the with implementation case the constraint is set at 1718 ha.

reference nutrient loads for the main river at the outlet at Hamburg (Seemanshöft) for the period 2000-2005 are 3500 t TP a^{-1} and 98500 t TN a^{-1} (cf. Venohr et al. 2010).

Table 6: Load reduction, abatement costs and shadow prices for restored floodplain area for increasing abatement targets before and after implementation of a floodplain restoration programme

	Restoration programme [a]	Unit	Load reduction target [b]			
			5 %	15 %	25 %	35 %
Load reduction	With & without	t TP	175	525	875	1,225
	With & without	t TN	4,925	14,770	24,630	34,480
Total abatement costs	Without	Mio €	9	221	688	1,529
	With	Mio €	7	202	656	1,484
Marginal load abatement costs	Without	€ kg TP^{-1}	5.3	7.6	48.6	137.3
	Without	€ kg TN^{-1}	3.8	37.1	59.6	125.2
	With	€ kg TP^{-1}	4.4	5.9	16.9	29.7
	With	€ kg TN^{-1}	3.5	36.5	58.6	121.2
Shadow price of floodplain area [c]	Without	€ ha A_{in}^{-1}	1,716 (835)	12,218 (5,067)	23,416 (10,689)	52,914 (24,2099)
	With	€ ha A_{in}^{-1}	1,531 (736)	11,849 (4,862)	19,809 (8,331)	40,407 (16,831)

(a) with / without implementation of floodplain programme. In the without case, the constraint on available inundated floodplain area (A_{in}) is set at 0 ha, for the with case the constraint is set at 1718 ha.

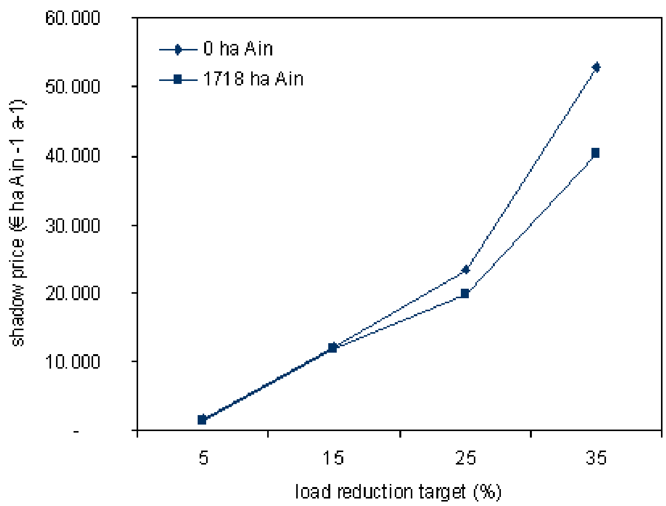
(b) for simultaneous reduction of TN and TP loads at river outlet at Hamburg -Seemanshöft

(c) mean shadow price (and SD) across 16 river sections for a unit of average annual inundated area

In Table 6 we present results for a joint and proportional reduction of the load of the two nutrients at the outlet to the North Sea with and without implementation of the floodplain restoration programme. The abatement target constraint for each nutrient is binding so that there is a marginal value for the abatement requirement that rises as reduction efforts are increased from 5 % to 35 %. The marginal costs for load reductions without the floodplain programme rise from 5.3 to 137 € kg^{-1} for phosphorous and from 3.8 to 125 € kg^{-1} for nitrogen. The marginal costs are lower when including the nutrient retention generated by the floodplain restoration programme: they rise from 4.4 to 29.7 € kg^{-1} for phosphorous and from 3.5 to 121 € kg^{-1} for nitrogen.

The corresponding estimates of the shadow price for restored floodplain area are also shown in Table 6. The shadow price reflects the change in total abatement costs if one additional unit of “average annual inundated area” is made available (or removed). The shadow price increases with increasing abatement requirements. It decreases with an increasing area of restored floodplains (from without to with case). Figure 2 and data in Table 6 shows the effects of increasing abatement requirements and the size of the implemented restoration programme on the shadow price. The shadow price for the first additional unit of inundated floodplain area rises from 1716 € ha^{-1} for an abatement target of 5 % to 52914 €ha^{-1} for a target of 35 %. The shadow price for an additional unit of restored floodplain after the restoration of 1718 ha is lower, and rises from 1531 € ha^{-1} for an abatement target of 5 % to 40407 € ha^{-1} for a target of 35 %. The reduced shadow prices for increasing floodplain area are explained by the fact that the first additional unit of floodplain replaces the most costly alternative measure and latter units replace increasingly less costly measures in the model.

Figure 2: Shadow price of inundated floodplain area (Ain) for increasing load reduction requirements before and after the implementation of a floodplain restoration programme of 1718 ha average annual inundated area (Ain).



We investigate the effects of separate abatement requirements for phosphorous and nitrogen compared to joint abatement requirements (Table 7 and Figure 3). The results show that the shadow price for separate reduction targets are lower for phosphorous than for nitrogen. However the shadow prices for a joint and proportional reduction are only slightly higher than the shadow price for nitrogen reduction alone because of the measures that have combined effects on both nutrients. In addition we look at the effect of introducing an additional and proportional abatement requirement for phosphorous for an upstream location (at the Czech-German border at Schmilka) of the main river trajectory. This is motivated by the fact that phosphorous is the limiting nutrient in the river ecosystem. Any effects on the biological quality status of the river would require phosphorous loads to be reduced already before the entry to the main river trajectory. We find that the shadow price in this case is almost identical to that for the separate nitrogen reduction target at the outlet because the required upstream load reductions are already part of a minimum cost solution for the downstream target. Taken together, the results show that the nitrogen load reduction component in the joint reduction target is the determinant of total abatement costs.

Figure 3: Shadow price of inundated floodplain area (Ain) for various combinations of nutrient reduction targets

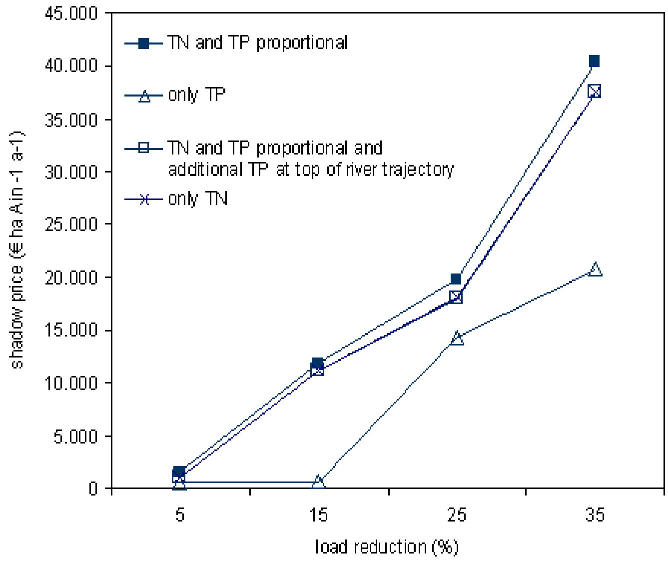


Table 7: Shadow price of inundated floodplain area (€ haA_{in}^{-1}) for various specifications of the load reduction targets

Load reduction target valid for …		Load reduction requirement			
Nutrient	Location [(a)]	5 %	15 %	25 %	35 %
only TN	downstream	1071[(b)]	11,231	18,072	37,500
only TP	downstream	460	698	14,385	20,721
TN and TP	downstream	1,531	11,849	19,809	40,407
TN and TP	downstream plus TP also upstream	1,071	11,227	18,063	37,500

(a) downstream = Hamburg Seemanshöft, upstream = Schmilka

(b) mean shadow price after implementation of a program with additional 1718 ha of inundated floodplain area.

This is also shown in the sensitivity analysis of the effects for lower nutrient retention capacities of floodplains on the shadow price (Table 8). A proportional reduction of the nutrient retention capacity of both nutrients by 50 % reduces the shadow price by 50 %. A reduction of the retention capacity only for phosphorous by 50 % reduces the shadow price by roughly 10 %, while a 50 % reduction of the retention capacity for nitrogen reduces the shadow price by roughly 40 %.

Table 8: Sensitivity analysis for the effects of lower nutrient retention rates of floodplains on the shadow price for inundated floodplain area (€ haA_{in}^{-1}).

Nutrient retention capacity (in % of baseline)		Load reduction target			
TN	TP	5 %	5 %	5 %	5 %
100	100	1,716	12,218	23,416	52,914
50	50	858	6,109	11,708	26,457
100	50	1,440	11,817	20,877	45,721
50	100	1,134	6,510	14,247	33,650

4.2 Cost benefit assessment of floodplain restoration as a nutrient retention measure

We combine the cost estimates for floodplain restoration measures and the estimates of nutrient retention benefits to assess the net welfare effects of restoration within a partial cost-benefit analytical framework. This analysis is necessarily incomplete, because it does not consider the full range of ecosystem services (eg. flood risk mitigation or biodiversity conservation) that are associated with floodplain restorations. It therefore provides a lower bound estimate. The annual costs are calculated using the restoration cost function (Function 12 and Table 5), site specific values of A_{tot}, A_{in},, L_{dike} for each of 45 potential sites and a perpetual lifetime of the project. The benefits are calculated using the site specific values of A_{in} and the shadow prices from Table 6 for a 5 % and 25 % load reduction target. We calculate the cost benefit ratio for each site using the usual discount rate of ca. 3 % for public investments in Germany.

Table 9: Partial cost benefit analysis for 45 potential dike realignment projects along the Elbe trajectory.

Discount rate	Load reduction target	Implemented scope of floodplain restoration programme	Shadow price	Benefit cost ratio		BCR >1
%	%	[-]	€ haA_{in}^{-1} a^{-1}	Mean [a]	Median [a]	% [a]
3	5	Minimum	1,716	0.19	0.13	0
3	5	Maximum	1,531	0.09	0.15	0
3	25	Minimum	23,416	2.65	1.78	84
3	25	Maximum	19,809	2.24	1.51	79

[a] for n = 45 restoration sites

The results (Table 9) indicate that with a load reduction target of 25 % (corresponds to the current 24 % policy target), the benefits from nutrient retention alone are large enough to generate a benefit-cost ratio larger than 1. This is the case for ca. 80 % of the sites. In contrast, at a lower load reduction target of only 5 %, none of the sites would have a benefit-cost ratio larger than 1. Generally, the net benefits are higher for projects with a lower ratio of length of new dike line to total restored area and with a higher ratio of average inundate area to total area.

5 Conclusions

This paper has presented an application of an indirect method, the alternative or replacement cost method, to value nutrient retention ecosystem services. The presented cost minimisation approach provides a tool for estimating a shadow price of an unpriced benefit provided by wetland ecosystems. The estimated values can be used for subsequent economic appraisal of floodplain and integrated water resource management projects in the Elbe River basin. They also provide further empirical evidence on the economic value of restoring wetland ecosystem services.

We have shown that, as expected, the shadow price or nutrient retention benefit of restored floodplain area increases with increasing nutrient load reduction targets. Scope effects have a smaller impact, but marginal benefits decrease with increasing scope of the floodplain restoration projects. The findings underline the fact that value estimates for regulatory ecosystem services are highly dependent on the contextual conditions of the service benefit area, such as the availability of substitute abatement options and abatement targets. We also conclude that the existence of clearly defined policy targets, such as for phosphate and nitrogen load reductions to the sea, in principle enhance the applicability of the replacement cost method. Implementing the replacement or alternative cost method is not difficult in concept, although detailed empirical analysis requires considerable effort. The process of imputing shadow prices that takes the various interdependencies in a river basin context into account is best addressed within the framework of optimisation models.

The data on shadow prices presented in Table 6 can be used for benefit transfer for dike relocations along the Elbe River. The appropriate marginal benefit needs to be selected based on the appropriate load reduction target. It can be adjusted for scope effects. Additional adjustments will be required to transform from average annual inundated area to total area. The data given in Table 1 gives some indication on the appropriate ratio, even though this is an important site specific piece of information. Such adjustments are also necessary for a comparison of the results of this study to the results of other studies, for example an earlier study for the Elbe by Meyerhoff and Dehnhardt (2007). These authors assume an average retention rate of 200 kgTN $ha^{-1}a^{-1}$ of total restored area. They proceed to value this retention using estimated emission abatement costs at source in the range of 2.5 - 7.7 € $kgTN^{-1}$. This yields average benefits per unit of total restored floodplain area of 440 – 1540 € ha^{-1}. Using a mean ratio of inundated area (Ain) to total area (Atot) of 1:13 (Table 1), the comparable retention rate for an average unit of total restored floodplain in the present study would be in the range of 42 kgTN $ha^{-1}a^{-1}$. This is much lower. The adjusted benefit estimates rise from ca. 117 to 3100 € ha^{-1} of total restored area. Thus, whilst the resulting order of magnitude of benefits found in the two studies is similar, the estimate of the present study is based on a lower retention rate and on evidence of higher marginal abatement costs for TN load reductions.

A factor we could not account for in our analysis is the expected reduction of retention rate with increasing abatement efforts and decreasing nutrient load. Whilst this might be conceptually appealing, we did not find that the available empirical data on nutrient retention is good enough to warrant such an approach. Large uncertainty regarding the nutrient retention capacity of restored floodplains remains and there is continued need for the development of a robust method.

Given the large investment costs for dike realignments, it is a more surprising result of this study that the nutrient retention effects alone may generate sufficient benefits to provide an economic efficiency gain, if the long term target of a 24 % load reduction is the relevant target. This is not the case for the incremental target of a 7 % load reduction until 2014. It can therefore be considered likely that floodplain restoration will receive more attention as potentially cost-effective abatement measures as it becomes apparent that other low cost options for nutrient abatement become scarcer. However the key thrust of the argument developed in this paper is, that floodplain restoration projects need to be assessed as multifunctional projects, with the positive impacts on water quality being one of several co-benefit dimensions.

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Manuscript

PAPER VI

STRATEGIC COST-BENEFIT ANALYSIS OF AN INTEGRATED FLOOD PLAIN MANAGEMENT POLICY FOR THE RIVER ELBE

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This paper addresses strategic land use choices for floodplains from an integrated floodplain management perspective. It applies the ecosystem services approach to explore the economic effects of various options for large scale floodplain restoration in an extended cost-benefit analytical framework. Particular attention is given to scope effects in the assessment of benefits from flood risk reduction, nutrient retention and wetland habitat conservation for programs of increasing scale.

The choice of the appropriate strategy for floodplain management is contested between stakeholders of nature conservation and flood risk management. Whereas flood risk management interventions have focused on dike strengthening ("hold-the-line" strategy), nature conservationists are arguing for an integrated approach that includes large scale floodplain restoration and realignment of dike lines ("space for the river" strategy). The key empirical result is that large scale restoration of floodplains of the River Elbe provides an economic efficiency gain. The results therefore support the general policy shift in floodplain management from a "hold-the-line" to a "space for the river" strategy. It is argued that an extended cost-benefit analysis should be one component of a wider strategic policy appraisal process that integrates targets of river basin -, flood risk - and floodplain land use policies.

Keywords: integrated floodplain management, cost-benefit analysis, ecosystem services, flood risk management, Elbe River

1 Introduction

This paper addresses strategic land use choices for floodplains from an integrated floodplain management perspective in a cost-benefit analytical framework. Floodplain management inevitably involves trade-offs: trade-offs between the benefits of conducting activities on the floodplain against the risk and adverse consequences to these activities caused by flooding and trade-offs between the benefits and costs of reducing this flood risk, for example by the conversion of the active floodplain by construction of dikes. Floodplain economics can therefore be understood in terms of balancing the marginal benefits of converting or protecting (and restoring) natural floodplain land. Both protection and conversion generate public as well as private benefits (cf. Heimlich et al. 1998).

The net marginal private benefits that can be realized from protecting (or restoring) active floodplains that are regularly inundated may be relatively low, since there are few benefits of wetland protection that landowners can capture. These may include economic returns from extensive land use such as haying, grazing or timber harvest. In contrast, private returns to construction of river dikes may be relatively high as conversion makes possible intensive agricultural production and settlement development. Historically in central European countries such as Germany, the private and public benefits of floodplain conversion where perceived to be large and public incentives and investments were made to encourage floodplain conversion in order to promote food self sufficiency and economic growth.

As a result of this historical development, about 80-90 % of the floodplain of the German river stretches of the Rhein, Elbe, Donau and Odra has been protected from flooding by the construction of dikes (Brunotte et al. 2009). Along the Elbe, the majority of the dikes where constructed in the late 19th and early 20th century. However, embankment continued well to the second half of the 20th century and no new dikes have been conducted since the 1970’s.

In contrast, most benefits from protection and restoration of natural floodplains are public in nature. Examples include flood control, water quality improvement, fish and wildlife habitat and recreational opportunities. Over the course of the 20th century the public benefit of wetland protection came to be more fully appreciated. This can partly be attributed to the increased scarcity of the remaining natural floodplain landscapes and habitats. In addition, it is only relatively recently that the significance of the many regulating services provided by floodplain ecosystems has been clearly recognized (cf. Turner et al. 2008).

Public policy regarding floodplain land use in Germany is currently at a turning point (Monstad and Moss 2008). It is generally accepted that there is no further benefit to additional embankments; the last major flood protection projects were carried out 30 to 40 years ago. Public controversy now surrounds the merits of a restoration of natural floodplains by relocation of dikes (dubbed “giving space for the river” strategy) compared to a hold-the-line approach. As a consequence all policies proposed or project schemes designed to increase public benefits through restoration of floodplain functions are being contentiously debated. While in general there is a large public support for the restoration of natural floodplains, local land users and inhabitants often oppose restoration because it affects their interests.

From an economic perspective, the difficulty in determining whether a floodplain restoration policy is an appropriate policy goal lies in the difficulty of determining the value of the public benefits. It is this problem which is examined in this paper. It is well acknowledged in the environmental economics literature that the public goods characteristic has traditionally caused ecosystem services of wetlands to be undervalued in the assessment of management options. However, the concept of ecosystem services has become an important model to systematically link functions of ecosystems to human welfare (cf. Turner et al. 2008, Posthumus et al. 2010). This concept builds on the conceptual differentiation of ecosystem functions (processes and structures), the uses and benefits that these functions support (goods and services) and the economic values of these goods and services. Various methods of economic assessment have been developed to address the valuation problem.

This paper utilizes the ecosystem services approach to explore the economic effects of a large scale floodplain restoration in a cost-benefit analytical framework. The use of CBA in decision making context where these non-market impacts are expected to be significant has stimulated an extensive debate and literature (Turner 2008, Hanley and Barbier 2009, Brouwer and Pearce 2005, NRC 2005). When including also "non-priced" external effects or public goods in monetary units this is often referred to as an extended CBA. The application of extended CBA in the context of integrated floodplain management is rare. However, there are numerous examples from the scientific discourse. For example Gren et al. (1995), Kosz et al. (1996) and Schönbäck et al. (1997) estimate the benefits of the protection of Danube River floodplains. Brouwer and van Ek (2004) conduct a CBA of different flood management strategies for the Rhine that includes environmental benefits. Meyerhoff & Dehnhardt (2007) conduct a cost benefit analysis for floodplain restoration projects along the Elbe River and Dubgaard et al. (2005) for restoration projects along the Skjern River. Turner et al. (2007) present an extended CBA for managed realignment in the Humber Esturay and Jenkins et al. (2010) value ecosystem services from wetlands restoration in the Mississippi Alluvial Valley.

This paper combines the result of three other studies that address the valuation of specific ecosystem services at a basin scale for the Elbe River (flood risk: de Kok and Grossmann 2010, nutrient retention: Grossmann accepted, and willingness to pay for conservation: Meyerhoff 2003 and Meyerhoff and Dehnhardt 2007) for an integrated assessment of the cost and benefits for floodplain management programs of various dimension and composition. The analysis presented in this paper is novel, in that it explicitly accounts for aspects of scale of the restoration effort and upstream-downstream interdependencies in the valuation approach.

The following section will look briefly at the policy appraisal methodology at the strategic and river basin scale. The next section introduces the study area, the policy process and the considered floodplain management options. A further section outlines the assessment and valuation methodology, with a focus on the valuation of three ecosystem services: reduction of flood risk, nutrient retention and non-consumptive benefits of natural habitat and biodiversity conservation. The outcomes of the integrated assessment are presented before the paper ends with a conclusions section, elaborating on the effectiveness of measures, potential bottlenecks of the method, and room for future research.

2 Cost-benefit-analysis and integrated floodplain management

The focus of this paper is on the economic appraisal of strategic options for floodplain management. Strategic approaches are useful when the decisions involve problems of a large scale and solutions of a long term nature. Strategic studies or assessments can contribute to the development of a coherent strategy and large-scale plans that determines the framework within which management options can be selected and assessed in detail. The strategic options that are assessed in this study are large-scale restoration of floodplain functions ("give space to the river strategy") compared to a maintenance of the current dike line ("hold-the–line strategy").

A strategic assessment can help to explore the potential synergies between different policy goals. Major policy fields that are affected or affect floodplain land use are rural development, agricultural and forestry policies, water resources and flood risk management and nature conservation policies. Specifically the EU Flood Directive, the EU Water Framework Directive, the EU Fauna-Flora-Habitat Directive and the EU Common Agricultural Policy are important pillars of European environmental policy whose policy fields overlap in water dependant habitats like floodplain wetlands (Hasch and Jessel 2004). In order to develop a coherent approach that balances the various targets of public policy, floodplain management policy therefore generally needs to be appraised across a more full range of criteria than has typically been the case in the past, where investments in the dike infrastructure have been assessed exclusively from a flood risk mitigation perspective. It is therefore crucial to set out the floodplain land use policy objectives and then compare the alternatives in terms of their contribution to the achievement of these objectives.

Cost-benefit analysis can be a useful tool for determining the appropriate strategic approach. The central goal of economic appraisal is an efficient use of public resources. In Germany, like in most European economies, almost all capital works on the system of flood protection infrastructure are effectively funded out of general taxation. A well designed cost-benefit analysis should ensure that the strategy represents best value and that uneconomic schemes or policies are identified at an early stage. In Germany, cost-benefit analysis has been relevant in determining the worthiness of conversion and optimal protection levels for floodplain sites (Meyer and Messner 2005, Holm-Müller and Muthke 2001). Also the expenditure of public funds on major infrastructure investments is often justified by reference to a rank ordering system underpinned by standard economic cost-benefit-analysis (or cost-effectiveness analysis) applied on a project by project basis. However this kind of analysis in practice generally tends to focus on a single, tangible benefit dimension, namely flood risk reduction. Secondary environmental effects, for example with regard to nature conservation or nutrient retention benefits are ignored.

The problem of secondary environmental benefits also arises in other sectoral planning systems that are relevant for floodplain development. For example, the implementation process of the EU Water Framework Directive requires the analysis of the cost-effectiveness of measures to reduce nutrient loads proposed in the river basin management plans (Engelen et al. 2008). However, in practical applications of cost-effectiveness analysis, wetland restoration measures are only assessed in regard to their contribution to water quality targets. As a result of ignoring co-benefits, neither flood risk nor water management plans tend to consider floodplain restoration as economically advantageous options, because the unit costs of restoration are generally high and the benefits with regard to a single target dimension comparatively low. To remedy this situation, secondary environmental benefits should be introduced as additional co-benefits into sectoral, one dimensional cost-effectiveness analysis. The alternative option is to proceed with an integrated assessment or full cost-benefit analysis. Both type of analysis require the same type of information on the value of ecosystem service benefits. One of the roles of strategic economic assessments, as presented here, is therefore to provide such information that can then be used in more detailed project appraisal or sectoral analysis and thus facilitate a coherent evaluation framework.

3 Case Study

3.1 Floodplain restoration policies for the Elbe River

For this study we focus on an integrated analysis of floodplain management options focusing on the targets for river corridor development set out in three policy fields: flood risk management, nature conservation and water resources management. The following section outlines some of the relevant policy debate and describes the scenarios.

The German part of the Elbe River (Figure 1) has characteristics of a lowland river with a wide alluvial valley downstream of the City of Dresden. Approximately 84% of the floodplains along this river stretch are protected by dikes. The proportionate loss of active floodplains in the Upper and Middle Elbe differs according to the width of the alluvial valley. . The narrow valley of the southern section generally has lower losses of active floodplain. After entry to the wider lowland valley 50 -90 % of the floodplains have been diked (Brunotte et al. 2009). Despite these large reductions in active floodplain area, the Elbe still is one of the largest free flowing rivers in Central Europe because the German part of the Elbe is largely without weirs. The designation of large parts of the remaining active floodplains as a UNESCO Biosphere Reserve and other categories of protected areas highlights the importance the river landscape has been accorded for habitat and biodiversity protection in Germany.

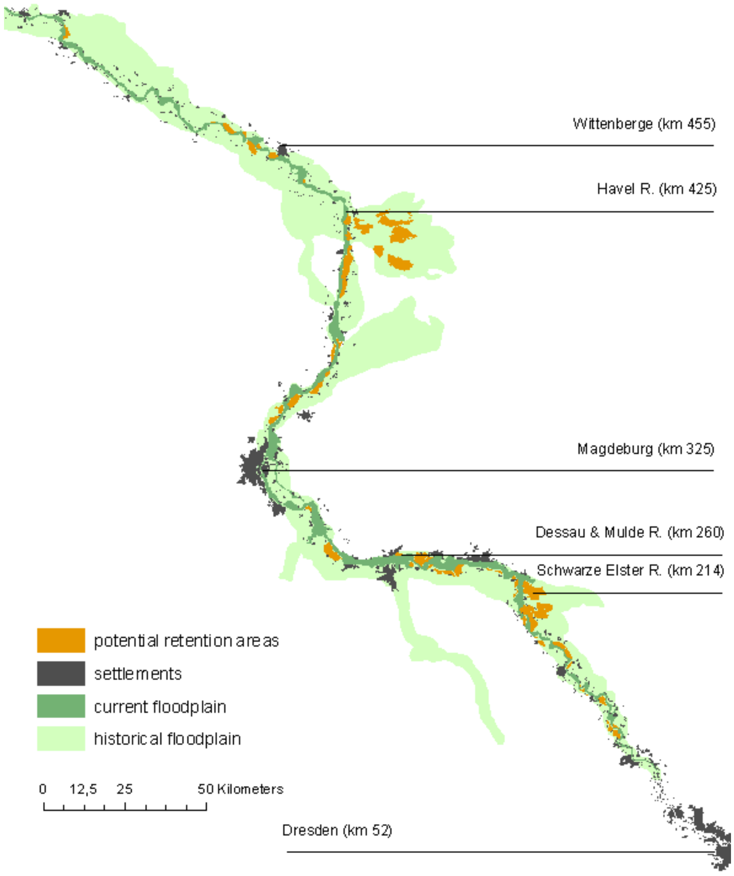

Figure 1: Map of the study area showing the extent of floodplains and location of the potential retention areas along the Elbe River.

The Federal Ministry of Environment (BMU) and the Federal Agency for Nature Protection (BfN) actively promote the concept of an integrated approach to the management and development of floodplains (BMU and BfN 2009). Such an approach seeks to harness multiple benefits for flood protection, water resource management, nature and biodiversity conservation and climate change mitigation. The strategy is based on the three principles of strict protection of remaining natural floodplain habitats, restoration of modified floodplains in agricultural use that are still subject to regular flooding and increased efforts to restore historic floodplains by dike realignments where feasible.

During the 1990s an analysis of potential dike relocation sites for large scale conservation programmes was conducted by Neuschulz and Purps (2000 and 2003). They identified 52 possible sites with a total of 23249 ha including 11 summer polders. The public debate on dike realignments gained momentum in the aftermath of the major flood on the Elbe in 2002. The International Commission for the Protection of the Elbe commissioned an action plan with the purpose to develop a comprehensive flood risk management strategy for the river (IKSE 2004). The proposed measures include amongst others, reactivation of the retention capacity along the river floodplains and rehabilitation of dikes to the design safety standard. Several federal state governments have since commissioned detailed studies to evaluate potential sites for dike relocations and retention polders (cf. Figure 1).

However dike relocations continue to be discussed primarily as nature conservation measures. The major driver for floodplain restoration on the Elbe to date has been the large scale conservation projects programme funded largely by the Federal Ministry of the Environment (BMU) through its Federal Agency for Nature Conservation (BfN). To date two such dike realignment projects are under way: Lödderitzer Forst (ca. 590 ha) and Lenzen (ca. 425 ha). A third dike realignment has been carried out in Roßlau (ca. 140 ha) funded by the state government, that is responsible for maintenance of the flood protection infrastructure. In this case, the decisive factor was the lower costs of a dike realignment compared to a reinvestment for the maintenance of the old dike line.

Finally this study refers to nutrient management goals set out under the draft management plan for the Elbe (FGG Elbe 2009) that are deemed necessary to comply with target of achieving a good ecological status of the river and coastal seas. The current goal is to reduce both phosphorous and nitrogen load to the sea by 24 % by the year 2027. This is to be achieved in a stepwise approach, with a third of the commitment to be achieved over three reporting periods ending 2014, 2021, 2027. The Federal Agency for Nature Conservation (BfN) has commissioned studies to explore potential overlaps and synergies of conservation oriented floodplain management with water resource management as mandated by the European Union Water Framework Directive (cf. Korn et al. 2006). While river management plans have taken up the issue of restoration of river morphology including the restoration of the natural floodplains on minor rivers, major floodplain restorations on the main river trajectory as discussed in this paper have not featured in river basin management plans to date.

3.2 Management options

For this study, we analyse seven potential restoration programmes that are based on various scoping studies that have been conducted in recent years. People and property assets are not part of the trade-off in this set of projects. The sites have been deliberately chosen in the scoping studies to avoid such conflicts and basically only involve the conversion of agricultural and forestry land to natural floodplains.

The number, exact location, area and retention volume of potential sites is the subject of public debate and constant review. For our assessment, the proposed sites and dimensions from several data sources were included (Merkel et al. 2002; Ihringer et al. 2003; IKSE 2004; Förster et al. 2005, BfG 2006). In case of divergent information on dimensions for a site, the larger alternative was chosen for this study. A more detailed description of the sites can be found in Grossmann et al. 2010. A map of the potential sites is presented Figure 1. Table 1 provides a summary of key characteristics the sites included in this study. The total area (A_{tot}) of the 60 potential restoration sites is 20749 ha, with an average annual inundated area ($E(A_{in})$) of 1717 ha. This difference is due to the different topography of the sites that will lead to different inundation frequencies. The mean ratio of total area to average inundated area for the proposed project sites is 1:13.

Table 1: Summary of key characteristics of the potential restoration sites

	Restoration area			New dike line	
	Total area (A_{tot})	Average annual inundated area (A_{in})	Ratio ($A_{tot}:A_{in}$)		
	ha	ha	(-)	km	km/ha
Mean	482	40	13	4.2	0.013
Median	350	15	10	2.9	0.011
Total	20749	1717	0.08	181	0.009

N = 60 sites

We consider two types of measures for the restoration of floodplain sites: dike We consider two types of measures for the restoration of floodplain sites: dike relocations and retention polders. Dike relocations entail removing the old dike line and construction a shorter backward dike line. Dike relocations require a change in land use and constitute a restoration of natural floodplain functions. The principle of retention polders is that they enable controlled flooding of an area that is enclosed by a dike line and that the inflow is regulated by weirs. The advantage of regulated retention polders is that they can be more effective in reducing the peak water levels of a flood wave. Continued intensive agricultural land use is possible because these polders are only flooded during extreme flood events – however a change from arable to grassland production may be necessary. Retention polders may therefore be considered to provide a partial restoration or enhancement of natural floodplain functions. However flood polders may alternatively be operated with an "ecological" flooding regime, whereby the flood polders are generally open to regular flooding but are closed prior to major flood events. In this case natural floodplain habitat and functions are fully restored.

For analytical purposes this paper concentrates on seven programmes, consisting of various combinations of measures (Table 2) which were chosen to illustrate the magnitude of effects that can be achieved by various schemes of different magnitude and location. Each programme is compared to the baseline that describes the situation as outlined in the flood action plan for the year 2000. Recent improvements of the dikes since the flood are not included in the database.

Table 2: Characteristics of the floodplain management programmes

Programme		Polder operation	River stretch	Number of sites	Polder area	Relocation area
			Elbe km	n	ha	ha
DR L	Dike relocation (large scale)	-	117 – 536	60	0	34658
DR S	Dike relocation (small scale)	-	120.5 – 536	33	0	9432
P L	Controlled retention polders (large scale)	Flood	117 – 427	31	25 576	0
P S	Controlled retention polders (small scale)	Flood	180	5	3248	0
P(e) S	Controlled retention polders (small scale) with ecological flooding	Ecological	180	5	3248	0
P+DR	Combination of polders and dike relocation	Flood	117 – 536	17	4143	3402
P(e)+DR	Combination of polders with ecological flooding and dike relocation	Ecological	117 – 536	17	4143	3402

In detail, the programmes are defined as follows:

DR L: dike relocation (uncontrolled operation) of all 60 potential sites included in the database irrespective of their designation for dike relocation along the river stretch Elbe km 117-536. The total floodplain area is 34,658 ha with a storage capacity of 738 million m^3. The purpose is to examine the potential effects of a dike relocation program which is much larger than the 15,000 ha analyzed in Merkel et al. (2002) or otherwise currently under discussion.

DR S: dike relocation (uncontrolled operation) of the 33 potential sites identified in the IKSE action plan (IKSE 2004) in the river stretch Elbe km 120.5-536. The total area is 9,432 ha with a storage capacity of 251 million m^3. The purpose is an assessment of a dike relocation program of a more realistic dimension as is currently being discussed.

P L: controlled operation of 31 potential sites for retention polders identified in IKSE (2004) along the river stretch Elbe km 117-427 with a total area of 25,576 ha and a total storage capacity of 494 million m^3. The polders in Sachsen-Anhalt are dimensioned according to Ihringer et al. (2003) and the polders on the Havel are included and dimensioned according to Förster et al. (2005). The purpose is an assessment of the hypothetical maximum attainable damage reduction through the retention effect.

P S: controlled operation of only the largest 5 potential sites for retention polders identified in Ihringer et al. (2003) near Elbe km 180 with a total area of 3,248 ha and a storage capacity of 138 million m^3. The purpose is to assess the contribution of the largest upstream sites to the maximum attainable damage reduction of alternative P L.

P(e)S: like P S, but the retention polders are operated with ecological flooding.

P+DR: This programme is a multifunctional scheme based on the results of more detailed scoping studies by the Federal States as described in BfG (2006). This includes controlled operation of 6 retention polders upstream near Elbe km 117-180 with a total area of 4,143 ha and a storage capacity of 92 million m^3. In addition 11 dike relocations along the trajectory of 3,402 ha. Polders are operated without ecological flooding.

P(e)+DR: like P+DR, but the flood polders are operated with ecological flooding.

4 Valuation methods and data

4.3 Cost benefit model of floodplain restoration

The components of the cost-benefit analysis are summarised in equation (1) and follow the standard with and without procedure, which in this case sets the net discounted benefits and costs of each management strategy against the reference or hold-the-line management strategy.

The project costs include three major components: the cost of establishing the management option in terms of investment and operation and maintenance costs. The costs for the rehabilitation and maintenance of the existing dike line are equal in the with and without scenarios apart from two situations: the construction and maintenance costs for necessary new dike sections are included in the project costs, while the saved rehabilitation and maintenance cost for those sections of the existing dike line that are realigned and removed are counted as a benefit. Further costs are the opportunity cost, which represents the loss in economic rent from the initial use of the floodplain land resource, in our case the loss of benefits from agricultural and forestry.

We consider four benefit elements. First, the saved costs resulting from a reduction of the necessary rehabilitation and maintenance efforts from shortening the dike line are considered as a benefit. In addition we consider three dimensions of ecosystem service benefits: flood risk mitigation, nutrient retention and habitat and biodiversity conservation.

The net benefit or net present value of implementing a floodplain restoration programme then is:

$$NPV_t = \sum_{t=0}^{T} \frac{1}{(1+r)^t} \left[(SC_t + FD_t + NR_t + BD_t - PC_t \right] \quad (1)$$

where NPV is the net present value of a floodplain management option in comparison to the "hold the line" baseline, PC are the project costs, SC are the saved rehabilitation and maintenance costs for the realigned dike line, FD is the reduction in flood damages, NR are the nutrient retention benefits and BD are the benefits from habitat and biodiversity conservation.

Throughout we use a social discount rate of 3 % and a project life time of 100 years, which is the lifetime of dikes. We conduct sensitivity analysis for a lower discount rate of 1 % and a shorter project life time of 30 years. We present the net present value (NPV) and the benefit-cost-ratio (BCR) for two assessment perspectives: a single benefit perspective focusing only on benefits from flood risk reduction, as is typically employed in cost benefit analysis of flood risk management options in Germany (frm), and an integrated floodplain management (ifm) perspective, that takes full account of multiple benefits provided by floodplains. The indicators are calculated as follows:

NPV_frm = SC +FD-PC (2)

NPV_ifm = SC + FD + NR + BD - PC (3)

BCR_frm = (SC + FD) / PC (4)

BCR_ifm = (SC + FD + NR + BD) / PC (5)

4.4 Benefits from reduction of flood risk

The benefits in terms of flood risk are measured in terms of avoided average annual flood damages. The change in expected average annual damage is the correct way to estimate the monetary effect of a mitigation measure in a cost-benefit analysis (Penning-Rowsell et al. 2003, NRC 2000). In the context of the risk based approach, flood risk is understood to be the product of the flood hazard (i.e. extreme events and associated probability) and the resulting damage. Ideally, a flood risk analysis should take into account all relevant flooding scenarios, their associated probabilities and possible damage. From these both a risk curve, i.e. the full distribution function of the flood damage, and the annual expectation value of the flood damage can be derived.

The downstream effects of flood risk mitigation measures require a flood risk assessment methodology that can be applied at the scale of a large river trajectory. For this study we apply a rapid flood risk assessment methodology developed for the River Elbe. The details of the methodology are described in de Kok and Grossmann (2010). A one dimensional hydraulic routing model is used to model the effect of planned (regulated and unregulated) and unintended retention (dike breaches) on the peak water levels. The model is complemented by an inundation model for dike breaches due to dike overtopping and a macro-scale economic approach to assess the resulting flood damage as function of inundation depth and land use classes. The method allows for the comparison of the flood risk at the scale of the main river trajectory, which has not been possible for the River Elbe to date (Figure 2). However, the method has some limitations regarding the additional local water level reductions by dike relocations.

The model system was applied to calculate the expected average annual damage for each set of measures. The flood risk was calculated by repeating the damage assessment for a series of flood events with recurrence intervals of 2, 10, 20, 50, 100, 200, 500 and 1000 years at the gauge station of Dresden (Elbe km 56):

$$E(AD_{tot}) = [(1-P_1)D_1 + \cdots + (P_{n-1} - P_n)D_n + \cdots + P_N D_{\max}] \quad (6)$$

where $E(AD_{tot})$ is the expected average annual value of the flood damage in €, P_1 is the exceedance probability of the lowest peak discharge causing flood damage with a recurrence interval of 2 years, P_n is the exceedance probability of flood event with a recurrence interval of *n* years, D_n is the corresponding total flood damage in €, and D_{max} is the maximum flood damage for event *N* (a 1000-year event).

Table 3: Avoided average annual flood damages per unit of restored floodplain area for seven restoration programmes (from de Kok and Grossmann 2010).

Programme	Restored floodplain area		Avoided average annual damage
	Total area	Controlled polder	
	ha	%	€ ha^{-1}
DR L	34 659	0	165
DR S	9 432	0	68
P L	25 576	100	1015
P S / P(e) S	3 248	100	4120
P+DR / P(e)+DR	7 545	55	1825

The avoided average annual damage is then calculated as the difference between the flood risk estimates for the measure and for the reference “hold the line” scenario. In terms of overall performance (Table 3), the maximum reduction of the expected annual flood damage is achieved by the controlled operation of the maximum potential of retention polders (P L). Singling out the effect of the major upstream polder groups included in P L, option P S shows that approximately 50 % of the avoided damage of the

P L measures can be traced back to the effect of the upstream polder group P S alone. The results indicate that there are decreasing returns to scale, because additional retention capacity downstream does not proportionally reduce the flood damage further. The additional effect of additional polders is also dependent on the location of sites in relation to the areas at risk. This is well illustrated by the distribution of the avoided average annual damages along the river trajectory for the two scenarios (Figure 2). The avoided annual average damage is lower for the two dike relocation programs with uncontrolled retention (DR L and DR S).

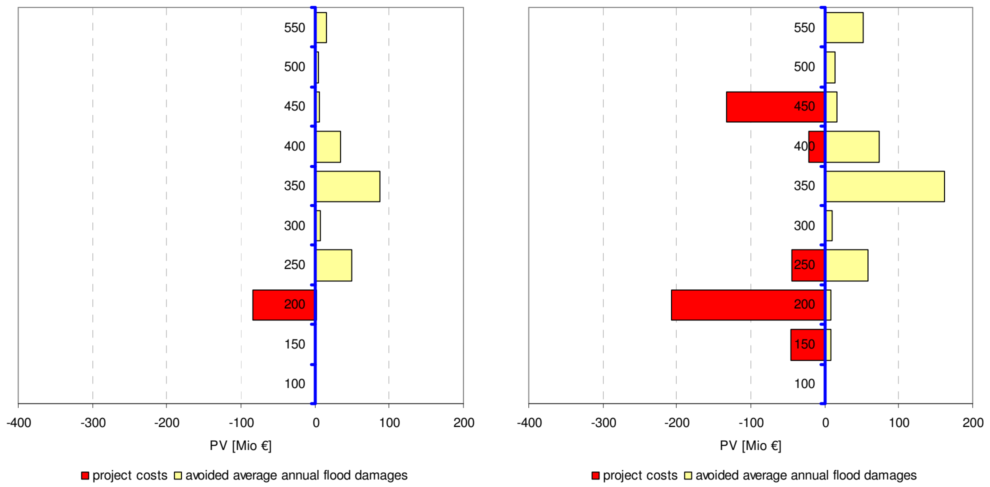

Figure 2: Distribution of present value of the project costs and avoided average annual damages along the river trajectory for two restoration programmes with retention polders, P S (left) and P L (right).

4.5 Benefits from nutrient retention

We use an indirect method of benefit estimation that is based on the replacement or alternative cost method to value benefits from nutrient retention. Replacement cost values do not constitute a direct estimate of the benefit from the ecosystem service to society (i.e. the value of clean water); they represent the value of having the ability to provide the benefit in demand through an ecosystem service rather than through an alternative method.

However, this method can be used for an indirect valuation of ecological services if the following conditions are met (NRC 2005, Turner 2008): (1) the alternative considered can provide similar services as the natural wetland (2) the alternative used for cost comparison should be the least-cost alternative (3) there should be substantial evidence that the service would be demanded by society if it were provided by that least-cost alternative.

Nutrition retention by restored floodplains can in principle substitute for other measures to reduce nutrient loads in a river basin. The cost savings compared to the least-cost combination of alternative measures (such as waste water treatment or land management) required to reach a reduction target are then the alternative or replacement costs. The validity of the alternative cost approach in this example hinges on the implicit assumption that the water quality targets set out under public environmental policies reflect the underlying water quality preferences (aggregated demand) of the general populace. In this case the marginal benefit from a policy to achieve the targets will be equal to the marginal costs of the required measures. By taking advantage of this assumed equality, the replacement cost approach allows one to value the marginal benefit from nutrient retention using the marginal costs of nutrient abatement.

For the current analysis, we use results from the application of a cost-minimization model that selects least-cost combinations of measures to reduce nutrient loads in the Elbe Basin. The shadow price reflects the savings in total nutrient abatement costs if one additional unit of "average annual inundated area" is made available.

We estimate the total retention of restored floodplain sites as the product of the average annual inundated area of a site (see below) and the specific nutrient retention rate per inundation day. Based on an evaluation of the literature, the model uses a retention rate of 0.8 kg TP ha^{-1} d^{-1} for phosphorus that we consider to be a lower reliable estimate (0.33 percentile). We use the recommended value of 1.5 kg TN ha^{-1} d^{-1} for nitrogen. The details of the model and its application are reported in Grossmann (accepted).

The resultant shadow prices are dependent on the abatement target. The study is based on the nutrient load reduction targets set out under the draft management plan for the Elbe (FGG Elbe 2009) that are deemed necessary to comply with target of achieving a good ecological status of the river and coastal seas. The current political goal is to reduce both phosphorous and nitrogen load to the sea by 24 % in a stepwise approach. In Table 4 we present the model estimates of the shadow price of floodplain restoration. The shadow price increases with increasing abatement requirements - from a load reduction requirement of 5, 15 and 25 %. It decreases with an increase in the available restored floodplain area. The shadow price for a marginal increase in inundated floodplain area (A_{in}) from the current level for a load reduction target of 25 % is 23,416 € ha^{-1}. The shadow price for a further marginal increase after the activation of the 1,718 ha floodplain decreases to 19,809 € ha^{-1}. We interpolate the appropriate shadow price depending on the total floodplain restoration area of the programs.

The annual nutrient retention benefit is then estimated as the product of the average annual inundated floodplain area and the shadow price. The average annual inundated area is estimated for each of the project sites as follows. Discharge - exceedance and discharge - stage functions for each river section are available from a study of the morpho-dynamics of the Elbe River (cf. Nestman and Büchele 2002). We use the discharge exceedance curve for each site along the river trajectory and divide these into ten classes of equal duration. We then interpolate the corresponding stage for each of the eleven breakpoints (Qmin, Q10,..., Q90, Qmax) from the discharge-stage function. The corresponding inundated area (Ain) for each stage is estimated by extrapolating the water level over a digital elevation model of the floodplain using GIS analysis. The expectation value or average annual inundated area E(Ain) for each project site is then calculated by sum of the product of inundated area and the corresponding occurrence probability to ensure normalization as follows:

$$E(A_{in}) = 0.05*A_{in}(Q_{min}) + 0.1* A_{in}(Q_{10}) + .. + 0.1* A_{in}(Q_{90}) + 0.05*A_{in}(Q_{max}) \quad (7)$$

Table 4: Shadow price of additional floodplain areas (in € ha^{-1} average annual inundated area) for increasing nutrient load reduction targets and restored floodplain area for the River Elbe (from Grossmann 2010b).*

Additional inundated area (Ain) in ha	TN and TP load reduction requirement (%)		
	5	15	25
1	1716	12218	23416
1500	1531	11849	19809

* based on a retention rate of 0,8 kg P $ha^{-1}d^{-1}$ and 1,5 kg N $ha^{-1}d^{-1}$ of inundated floodplain area E(Ain).

In Table 4 we present the estimates of the shadow price of floodplain restoration. The shadow price reflects the savings in total nutrient abatement costs if one additional unit of "average annual inundated area" is made available. The shadow price increases with increasing abatement requirements. It decreases with increasing restored floodplain area. The shadow price for a marginal increase in inundated floodplain area from the current level rises from 1716 € ha A_{in}^{-1} for a load reduction target of 5 % to 23416 € ha A_{in}^{-1} for a target of 25 %. The shadow price for a further marginal increase after the activation of 1718 ha rises from 1531€ ha A_{in}^{-1} for an abatement target of 5 % to 19809 € ha A_{in}^{-1} for a target of 25 %. We interpolate the appropriate shadow price depending on the scale of the restoration project using a 25 % load reduction target.

4.6 Benefits from habitat and biodiversity conservation

We use results from stated preference studies to estimate the benefits of non-consumptive uses associated with the restoration of natural floodplain habitats and biodiversity. Non-consumptive uses are generated from maintaining rather than harvesting organisms and are based on amenity and recreational activities (such as enjoying the scenery) and the "non-use" values for instance in preserving natural heritage for future generations independent of any personal use of a site for example for recreation (Turner et al. 2008). The economic values deriving from non-consumptive use of biodiversity and habitats are potentially considerable; however they are also extremely difficult to measure. Revealed preference methods, such as the travel cost method or the hedonic pricing method can be used to estimate effects of changes in biodiversity levels on the recreational or amenity use-value component. However, these methods can not measure the "non-use" values. Stated preference methods, such as contingent valuation or choice experiments, are the only techniques regarded as suitable to derive estimates of biodiversity values that include non-use value components.

For this analysis we combine the results from two studies. First of all, we use the result of a primary study eliciting willingness to pay for restoration of the Elbe floodplains using the contingent valuation method. Details are reported in Meyerhoff (2003 and 2006). The study estimates the annual willingness to pay of the German population for a proposed 55, 000 ha program of floodplain restoration that includes 40,000 ha of habitat restoration on current floodplains and an additional 15,000 ha of floodplains to be gained by dike relocation. A mean adjusted annual willingness to pay per household of 5.3 € was estimated. This value includes protest bids as true zero bids and is adjusted for outliers and embedding.

Provided that wetland conservation is a normal good, economic theory would require the value estimates to be sensitive to the scope of proposed measures. We therefore combine the above point estimate with an estimate of the elasticity of demand derived from a meta-analysis of wetland valuation studies to scale the willingness to pay estimates to restoration projects of varying dimension. The details of this meta-analysis are presented in Grossmann (in review). The results indicate that the willingness to pay estimates are sensitive to the scope (area) of the proposed wetland measure and that on average willingness to pay is higher for larger wetland measures, however at a decreasing rate. The meta-analysis uses a log-linear functional form specification, whereby both the dependant variable (willingness to pay) and the wetland area are held in logarithmic form. In this case, the coefficients of the variables expressed as logarithms can be interpreted as the elasticity. The elasticity of demand for the area of conservation measures is estimated to be 0.3. We combine the point estimate from the primary study with this elasticity to derive willingness to pay estimates as function of restored area (Table 5).

Table 5: Estimated general public's willingness to pay for floodplain habitat and biodiversity conservation along the Elbe River trajectory.

	Unit	Restored area (ha)					
		5 000	15 000	25 000	35 000	45 000	55 000
willingness to pay per household	€/HH [a]	3.1	3.8	4.3	4.7	5.0	5.3
Aggregated willingness to pay per unit area	€/ha [b]	5142	2142	1461	1134	936	810

[a] based on an average WTP per HH of 5,3 €$_{2005}$ for 55 000 ha floodplain programme and a price elasticity of 0.3

[b] based on a population of 18,5 Mio in the Elbe Basin and an average 2.2 persons per HH.

4.7 Costs

We consider two sets of cost elements: the costs related to construction, operation and maintenance of the flood protection infrastructure and the opportunity costs of land use change. The assumptions for each cost element are summarized in Table 6. More details of the procedure can be found in Grossmann et al. (2010). The costs for flood protection infrastructure include the investment costs for dike construction, dike reinforcement, construction of weirs for polder operation and initial works for landscaping. Operation and maintenance costs for the weirs, dikes and nature conservation land management are considered.

The opportunity costs of agricultural and forestry land use are estimated for two cases: the permanent loss of land to agriculture for restoration and the permanent conversion of arable to grassland land in the case of controlled flood polders. The opportunity costs of agriculture or forestry land restored to natural wetlands is equal to a perpetual land rent. This is roughly equivalent to the purchase price for land. We therefore use market values of forest, arable and grassland land to estimate the opportunity costs. In the case of a conversion to controlled retention polders, that are only flooded during major flood events, agricultural land use can be retained. However we assume that arable land needs to be converted to grassland. The annual costs are estimated on the basis of loss in net margin. This is approximated by the compensation payments offered under agri-environmental schemes. Additional opportunity costs arise from temporary one-off losses from flooding of the controlled polder. This loss is valued on the basis of the gross margin. A probability of flooding during the damage prone growth season of once in 10 years is assumed.

The total cost for each of the 60 sites is the calculated as the product of the cost estimates and site specific data on the total area of a site, the initial share of grassland, forest and arable land use, the length of required new dikes, the number of required weirs, the length of dikes that can be removed and their status (rehabilitation required: yes/no). All site specific data is generated using GIS analysis based on CORINE land cover data, a digital elevation model, information on size and location of potential restoration sites and data on dike infrastructure from IKSE (2001).

Table 6: Costs of floodplain management measures.

Cost category	Cost element	Value	Unit
Land use: opportunity costs of permanent conversion of land use to restored floodplain	Land purchase costs: arable land	5500	€ ha^{-1}
	Land purchase costs: grassland	2500	€ ha^{-1}
	Land purchase costs: forestry land	2000	€ ha^{-1}
Land use: opportunity costs of conversion from arable to grassland for flood polder operation	Annual costs	250	€ ha^{-1} a^{-1}
Land use: flood damages under flood polder operation [a]	Expected average annual damage to grassland	25	€ ha^{-1} a^{-1}
Land use: landscaping of restored floodplain	Investment	300	€ ha^{-1}
	O&M	10	€ ha^{-1}
Weir [b]	Investment	650 000	€
	O&M	4500	€ a^{-1}
Dike [c]	Investment: new construction	525	€ m^{-1} m^{-1}
	Investment: rehabilitation of old dike line (as percentage of construction costs)	40	%
	Investment: opening old dike line	6000	€ km^{-1}
	O&M	0.1	€ m^{-2} a^{-1}
Planning overhead	% of total investment costs	5 %	%

[a] assuming a loss of gross margin of 250 € ha $^{-1}$ and flooding every 10 years

[b] assuming two weirs per polder and an economic lifetime of 30 years

[c] assuming a dike height of 4 m and an economic lifetime of 90 years.

5 Results of the cost benefit analysis

The central results of the cost-benefit analysis are presented in Table 7 and Figure 3. We present the net present value (NPV) and the benefit-cost ratio (BCR) for two perspectives of assessment: a flood risk management and an integrated floodplain management perspective. Projects with a positive NPV or a BCR larger than one are economically advantageous. The programs with the largest overall welfare effect / economic benefit are those with the highest net present value. Where there is a budget constraint that forecloses the realization of the largest programs the benefit cost ratio can provide an indication or ranking in terms of the returns to resource use.

We find that, addressed solely from a perspective focused only on flood risk management related benefits (frm), the BCR is highest for large retention polders at upstream locality (P S and P(e) S). The BCR for the option with ecological flooding is higher because the once-off payment for permanent conversion of land to nature is less costly than a continuous annual compensation for the conversion of arable land to grassland. The combination of upstream retention polders with dike relocation (P+DR / P(e)+DR) reduces the BCR, because from the flood risk management perspective the dike relocation programs taken alone (DR L / DR S) have a BCR lower than one. The program with a large number of additional retention polders along the trajectory (P L) also has a lower BCR because the additional flood risk damage reduction compared to the major upstream retention polder location is low.

Table 7: Net present value and benefit cost ratios for the floodplain management programmes

Programme		Area	NPV frm	NPV ifm	BCR frm	BCR ifm	NPV frm	NPV ifm
		ha	Mio. €	Mio. €	-	-	€/ha	€/ha
DR L	Dike relocation (large scale)	34659	-128	2520	0.8	5.8	-3,706	72,707
DR S	Dike relocation (small scale)	9432	-69	1465	0.7	7.6	-7,364	155,337
P L	Controlled retention polders (large scale)	25577	354	354	1.8	1.8	13,836	13,836
P S	Controlled retention polders (small scale)	3248	331	331	5.0	5.0	101,990	101,990
P(e) S	Controlled retention polders (small scale) with ecological flooding	3248	352	1396	6.6	23.1	108,261	429,746
P+DR	Combination of polders and dike relocation	7545	300	1375	2.8	9.0	39,769	182,198
P(e)+DR	Combination of polders with ecological flooding and dike relocation	7545	326	1481	3.2	11.2	43,227	196,337

* Discount rate of 3 % over a project lifetime of 100 years.

** frm = only flood risk management perspective, ifm = integrated floodplain management perspective.

The picture is different if viewed from an integrated floodplain management (ifm) perspective. First of all the BCR of all projects that generate additional benefits associated with floodplain habitat restoration are significantly higher. Second the ranking according to the BCR changes. The large scale upstream retention polders with habitat restoration (P(e) S), that provide both major reduction in flood damages and other ecosystem services, continues to rank highest. Programs with dike relocations (DR L / DR S / P +DR) now also rank highly with a large BCR. Programs that do not provide restoration benefits (P L / P S) rank lowest.

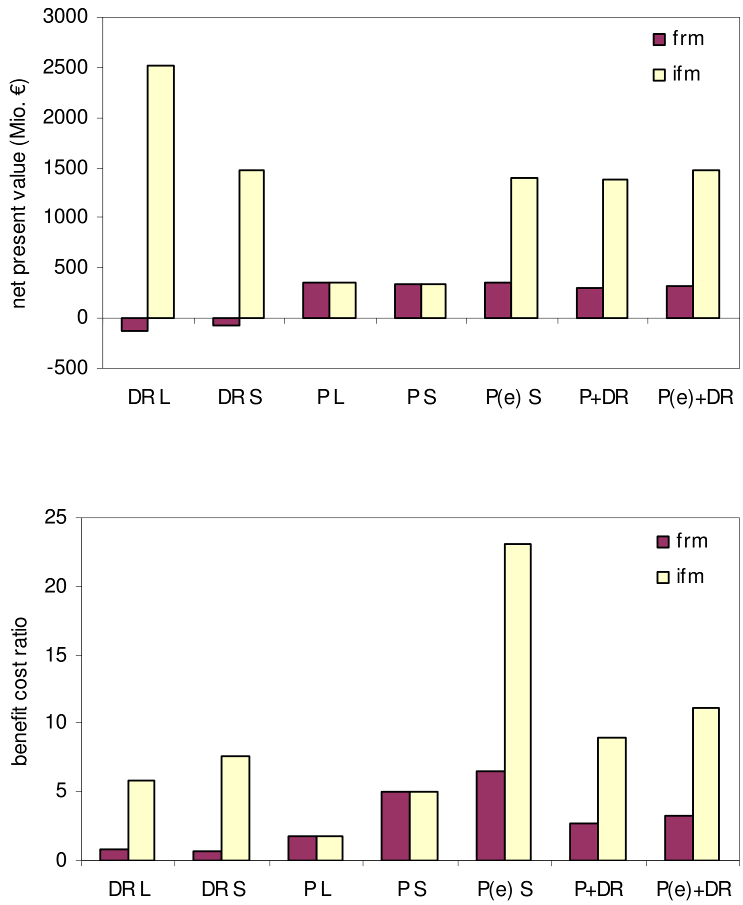

Figure 3: Net present value (upper) and benefit cost ratio (lower) for the floodplain management programmes

The total net present value gives an indication about the possible absolute increase in economic welfare that can be realized with the various programs. From an integrated flood plain management perspective, it is highest for the largest dike relocation (DR L) program of ca. 35,000 ha. The other programs with a dike relocation or habitat restoration component (DR S / P(e) S / P +DR / P(e)+DR) covering 3,200 -9,400 ha also generate high net present values. The NPV of programs that do not contribute to habitat restoration (P L / P S) is significantly lower, despite comparable areas of 3,250 -25,000 ha.

We also report the NPV normalized per project floodplain area. It describes the net present value generated per unit of restored floodplain land resource. This also facilitates comparison with other studies. The ranking of measures according to this criterion is related and follows the same pattern as described for the BCR.

Next we explore the share of the various benefit components of the total present value of benefits (Table 8). The share is dependent on the type of project. Of the projects that also address habitat restoration, benefits from reduced infrastructure maintenance account for 6 -14 %, reduced flood risk 1-28%, habitat conservation 37-66 % and nutrient retention for 6-44 % of the total present value of benefits.

Table 8: Share of benefit components of total benefits for the floodplain management programmes

Programme		Share of benefit component (in %) [a]			
		SC	FD	BD	NR
DR I	Dike relocation (large scale)	14	5	37	44
DR II	Dike relocation (small scale)	15	1	55	29
P L	Controlled retention polders (large scale)	0	100	0	0
P S	Controlled retention polders (small scale)	0	100	0	0
P(e) S	Controlled retention polders (small scale) with ecological flooding	0	28	66	6
P+DR	Combination of polders and dike relocation	6	27	61	7
P(e)+DR	Combination of polders with ecological flooding and dike relocation	6	25	59	10

a SC = saved costs, FD = flood risk reduction, BD = habitat and biodiversity conservation, NR = nutrient retention

Finally, we explore the sensitivity of results to changes in key parameters: (a) discount rate (b) appraisal period (c) assumptions regarding costs and the value of ecosystem services. Results are presented in Table 9. An interesting result is that all projects have a positive net present value even if for an appraisal period of 30 years, that is roughly one third of the economic lifetime of the dike infrastructure. A lower discount rate further increases the net present value of all projects. With regard to the change in the assumed specific costs or values of ecosystem services, the results show that the effects are in most cases under proportional and, with one exception do not reduce the NPV below zero. The size of effects vary dependant on the type of project and the share of the different types of benefits it generates. For projects that involve habitat restoration, a 50 % decrease in biodiversity benefit estimate reduces results by 24 - 35 %, a 50 % reduction of nutrient retention benefits by 4 -19 % and a flood damage benefits by 1-16 %. An increase in costs by 50 % only reduces the NPV by 3 - 6 %. Finally, a combined conservative combination of 50 % higher costs and 50 % lower values for all ecosystem services reduces the NPV between 55 and 62 %. The sensitivity analysis indicates that the key results of this analysis regarding the positive economic effects associated with floodplain restoration are stable over a very large range of assumptions regarding both cost and benefits.

Table 9: Sensitivity analysis

Programme		NPV in Mio. €	Sensitivity for components [b]						
			C	NR	BD	FD	MIN. [a]	R = 1,5	T = 30
			Change of NPV in Mio. €						
DR L	Dike relocation (large scale)	2520	-6	-29	-24	-4	-62	69	-42
DR S	Dike relocation (small scale)	1465	-3	-18	-34	-1	-56	65	-40
P L	Controlled retention polders (large scale)	354	-64	0	0	-114	-177	86	-55
P S	Controlled retention polders (small scale)	331	-13	0	0	-63	-75	65	-41
P(e) S	Controlled retention polders (small scale) with ecological flooding	1396	-3	-3	-34	-15	-55	63	-38
P+DR	Combination of polders and dike relocation	1375	-5	-4	-35	-16	-59	64	-39
Pe+DR	Combination of polders with ecological flooding and dike relocation	1481	-3	-5	-34	-14	-57	64	-39

[a] MIN = conservative estimate based on a combination using 50 % higher estimates for costs and 50 % lower value estimates for each of the benefits.

[b] C = costs, NR = nutrient retention, BD = biodiversity conservation, FD = flood damages, r = discount rate and T = appraisal time frame

6 Policy implications and conclusion

Both an integrated and strategic approach are required to provide a framework for developing, appraising and implementing major public works on the system of dikes and floodplains in a coherent manner. Extended cost benefit analysis can contribute to the development of such an approach by providing an economic efficiency oriented perspective. The key empirical result of this study then is that large scale restoration of floodplains of the River Elbe provides an economic efficiency gain – largely independent of the type of measures and the appraisal perspective. However, the results also illustrate the sources of controversies around floodplain restoration among sectoral planning agencies and their stakeholders. When assessed purely from a flood risk management perspective, dike relocations may seem to be less favorable then from an integrated floodplain management perspective.

The results of this study support the general policy shift in floodplain management from a “hold-the-line” to a “space for the river” strategy. The largest scenario presented in this study proposes a restoration of 350 km², which is way beyond the dimension of programs currently debated in the political realm. However, this scenario only constitutes 10 % of the loss of active floodplains in the last centuries that is estimated at 3,285 km² (IKSE, 2005). Given the complicated political economy of floodplain management, a sequential approach to the selection and appraisal process for the implementation of restoration sites would be appropriate. Initially all sites in which the opportunity costs do not involve complex trade-offs need to be identified. In such cases the opportunity cost will largely involve the loss of lower value agricultural land and an efficiency oriented analysis based on a CBA as presented in this case study could provide decisive information. In other cases, where people and built property assets are part of the opportunity cost calculation, CBA will not be as decisive and will need supplementation.

The development of an integrated approach to floodplain management will make multi-functional projects more advantageous. The aim of promoting multifunctional projects would be to provide a range of services (and address a range of policy targets) at a lower cost than if each where provided separately. This paper has addressed the question of efficiency of such programmes from an economic perspective. However, the analysis of cost and benefits presented here can also contribute to questions of how the costs of such programmes can be equitably shared. It provides information on the spatial and sectoral distribution of costs and benefits that can be used to negotiate cost sharing keys. This pertains for example to the sharing of costs for flood risk mitigating measure based on the distribution of benefits from reduced flood risks along the downstream trajectory (cf. Figure 2). For projects that address multiple policy targets, there may be situations where it will be more equitable to consider dividing costs in the ratio of the major benefits (cf. Table 8). It is reasonable to assume that for such projects, no sectoral agency or group of beneficiaries would be willing to pay a contribution which is larger than the whole life cost of meeting their specific requirement on a standalone basis.

We conclude that for an integrated assessment of floodplain management options, the standard cost benefit analysis applied for flood risk management needs to be extended to systematically incorporate the economic benefits derived from ecosystem services from floodplains. This paper has presented an example of the currently available approaches for such an extended assessment in the context of a major river in Germany. To improve the potential for the application in regular decision making processes, still more attention will need to be devoted to develop more readily available methods and data for the quantitative description of ecosystem service provision levels as well as for estimates of their value.

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Water Resources Management (accepted)

PAPER VII

INTEGRATED ECONOMIC-HYDROLOGIC ASSESSMENT OF WATER MANAGEMENT OPTIONS FOR REGULATED WETLANDS UNDER CONDITIONS OF CLIMATE CHANGE: A CASE STUDY FROM THE SPREEWALD (GERMANY).

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This paper presents an integrated economic-hydrologic approach for the assessment of water management options for wetlands. It is based on a water resources modeling framework for long-term basin planning, that is augmented to model ecosystem service provision levels of wetlands as a function of water availability and water management. The approach is applied to a case study of the Spreewald wetland that is major fen wetland in the mid-reaches of the River Spree (Germany). Different management options at the wetland and basin scale are assessed in a cost-benefit analytical framework regarding their performance under projections of future climatic conditions. The cost-benefit analysis is based on the valuation of important wetland ecosystem services: grassland fodder production, recreational boating, habitat and biodiversity conservation and regulation of greenhouse gas emissions.

It is found that under future climatic conditions, regulated and drained wetlands such as the Spreewald will require an increasing amount of water to maintain the current levels of benefits derived from the wetlands ecosystem services. Additional inter-basin water transfer could compensate some of the negative effects of increased water demand. However, the assessed transfer option is not economically efficient. Water management approaches that increase the intra-annual water storage in the wetland soils by higher groundwater level regulation targets are found to generate net gains in benefits compared to the current water management without any increase of the water supply.

Keywords: wetlands, water resources management, cost-benefit analysis, valuation of ecosystem services, integrated hydrologic-economic assessment model, climate change

1 Introduction

This paper presents an integrated economic-hydrological assessment methodology developed to assess the effects of water management options on the economic value of ecosystem services provided by wetlands. It provides a case study from the Spree River in Germany that highlights the potential contribution of economic analysis to improved water management in the basin. Almost all lowland fen wetland areas in Germany have been drained in the last centuries and as a result their water table is today regulated by weirs. Because of the negative climatic water balance additional water from the river systems is required to maintain water levels during summer months. The system of regulation and drainage has been constructed in a manner that allows water transfers and sub-irrigation of the wetlands. Regulated wetlands thus constitute one of the major water users within the lowland river systems. The wetlands provide many ecosystem services that are directly regulated by water availability such as fodder production, recreational opportunities, habitat and biodiversity conservation or regulation of nutrient and greenhouse gas fluxes. From both a hydrological and an economic point of view, wetlands such as the Spreewald must be understood as multi-functional water users competing with other water users upstream and downstream for sufficient water supplies (cf. Turner et al. 2008).

The need for economic analysis for the design and implementation of efficient water resources management policies is well documented in the economics literature. This need is also emphasised in the European Union's recent Water Framework Directive (cf. WFD-CIS 2003a). The need to include the multifunctional nature of wetland water use and the ecosystem service benefits that are generated by wetlands in the assessment of policy options have long been recognised in principle, for example in guidance documents of the Convention on Wetlands of International Importance (cf. Ramsar Convention Secretariat 2007), the EU Water Framework Directive (cf. WFD-CIS 2003b) or the International Union for Conservation of Nature (Emerton and Bos 2004). However, in the practice of economic assessment of water management at a basin or sub-basin scale, the economic value of ecosystem benefits provided by wetlands other than the private goods are still generally omitted. One of the reasons for this neglect lies in the difficulty and lack of experience in determining the value of the benefits from public goods. Recreational uses of waterways, the conservation of water dependent habitats in wetlands or regulation of nutrient and greenhouse gas fluxes are typical examples of such public goods. Even though there is no rivalry in the use of the services, the production of these public good services often is in competition with other water uses. Although there is some overlap, the valuation methods appropriate for public environmental goods differ from those for private goods. An increasing literature is available on the valuation of the diverse benefits provided by wetlands (cf. Brander at al. 2006, Woodward and Wui 2001). However, only few studies explicitly address the valuation of benefits as a function of water availability or water allocation towards wetlands. Such an approach is a prerequisite for the assessment of management options that affect the water availability for wetland sites in any water resources modelling framework (cf. Young 2005, Heinz et al. 2007). Examples for integrated approaches are mainly to be found in studies that attempt to assess the opportunity costs of diverting water for agriculture from wetlands (cf. Barbier 2003, Ringler and Cai 2006, Dadaser-Celik et al. 2009, Veijaleinen et al. 2010).

This study takes previous research further by providing a methodology for the systematic integration of multifunctional wetland water uses into a water resources modelling and assessment framework for large river basins. It uses the water resources modelling system WBalMo that is also used by the state water management authorities in Germany for long term water resource planning for example for the Spree River (Koch et al. 2005 and 2006). This model has been complemented by a more complex water management module for wetlands that is described in Dietrich et al. (2007a, 2007b, 2007c). This paper presents the integrated economic assessment methodology. It utilizes the ecosystem services approach to explore the economic effects of wetland water management options (cf. Turner et al. 2008, Fischer et al. 2009, NRC 2005). Different valuation approaches for different ecosystem services provided by wetlands are combined in an integrated, cost-benefit analytical framework. The use of cost-benefit analysis in decision making contexts where public or environmental benefits are expected to be significant has been extensively debated in the literature (Brouwer and Pearce 2005, Hanley and Barbier 2009).

The paper presents an application of the methodology to assess water management options for the Spree River and the major wetland in its river course, the Spreewald, under conditions of future climate change. The next section elaborates the key land and water use issues in the Spreewald. It is followed by a description of the assessment method, with detailed description of the valuation approaches for different ecosystem services. Results for management options both within the wetland and in the upstream river basin are presented. Finally these options are assessed in a cost-benefit analytical framework, comparing alternative water management options to the current practice, in order to identify economically efficient strategies.

2 Case study Spreewald wetland

The Spreewald wetland (Fig. 1) is located southeast of Berlin and is an inland delta wetland area within the middle reaches of the Spree River, which splits up into several branches that meander through a wide floodplain. The size of the wetland is approx. 320 km². The long-term mean precipitation is about 530 mm year^{-1} and the FAO grass reference evapotranspiration about 610 mm year^{-1} (HAD, 1998). Major current land and water uses are for agriculture, especially fodder production, forestry, nature protection, fisheries and recreation. During the last century, large areas of the floodplain have been drained and embanked in order to intensify agricultural production. Due to the low levels of precipitation, the drainage systems were augmented with weirs in the 1970s and 80s to regulate water levels and to enable sub-irrigation. This was a prerequisite for intensive agricultural production. As a result this region today has a complex water regulation system that is an integral part of the regulation system for the whole river basin. The system of streams, canals and ditches with a total length of more than 1600 km distributes the inflow from the rivers Spree (catchment size 2535 km^2), Malxe (345 km^2) and further smaller tributaries (1160 km^2) within the floodplain. At the lower end, the outflow is united again in the Spree River. All water levels are regulated by more than 600 weirs of different size. In the last 20 years the inflow from the upstream catchment has decreased dramatically because the pumping rate of drainage water from open cast lignite mines in the headwaters of the Spree River was reduced from approx. 30 $m^3 s^{-1}$ in 1989 to 17 $m^3 s^{-1}$ in 2010 (Grünewald 2001). This has resulted in conflicts over the allocation of available water resources among the water requirements for the restoration of open-cast mines in the headwaters, other upstream water uses such as thermal power stations and fish ponds, the environmental water demand for maintaining the Spreewald wetland and the minimum flow requirements for the downstream metropolitan area of Berlin (Koch et al. 2006).

Since 1991 the Spreewald wetland has the status of an UNESCO-Biosphere Reserve (Hiekel et al. 2001). Agriculture is the main land use in the Spreewald, covering 67% of the total area, with the rest of the area mainly covered by forest. 70 % of the agricultural land is grassland. Arable land is mainly found at the fringes of the wetland. Almost all grassland is currently managed as low-intensity grassland, even though there are some intensive milk production enterprises in the area. In the central wetland areas, a large share of the grassland is additionally managed to conform to nature conservation targets on a contract basis. The main rivers in the Spreewald have the status of navigable state waterways. To ensure navigability the large number of weirs on these river and canal sections are fitted with locks. The waterways are mainly used by traditional punting boats that offer trips for tourists. Tourism has a long tradition going back to the 18th century (MUNR 1998) and is of importance to the regional economy (Lienhoop and Messner 2009). In 1930 the region drew almost 200,000 visitors a year; in 1960 500,000 were recorded. Currently, about 2 to 2.2 million visitors visit the area each year, of which an estimated 1 million visitors participate in punt trips.

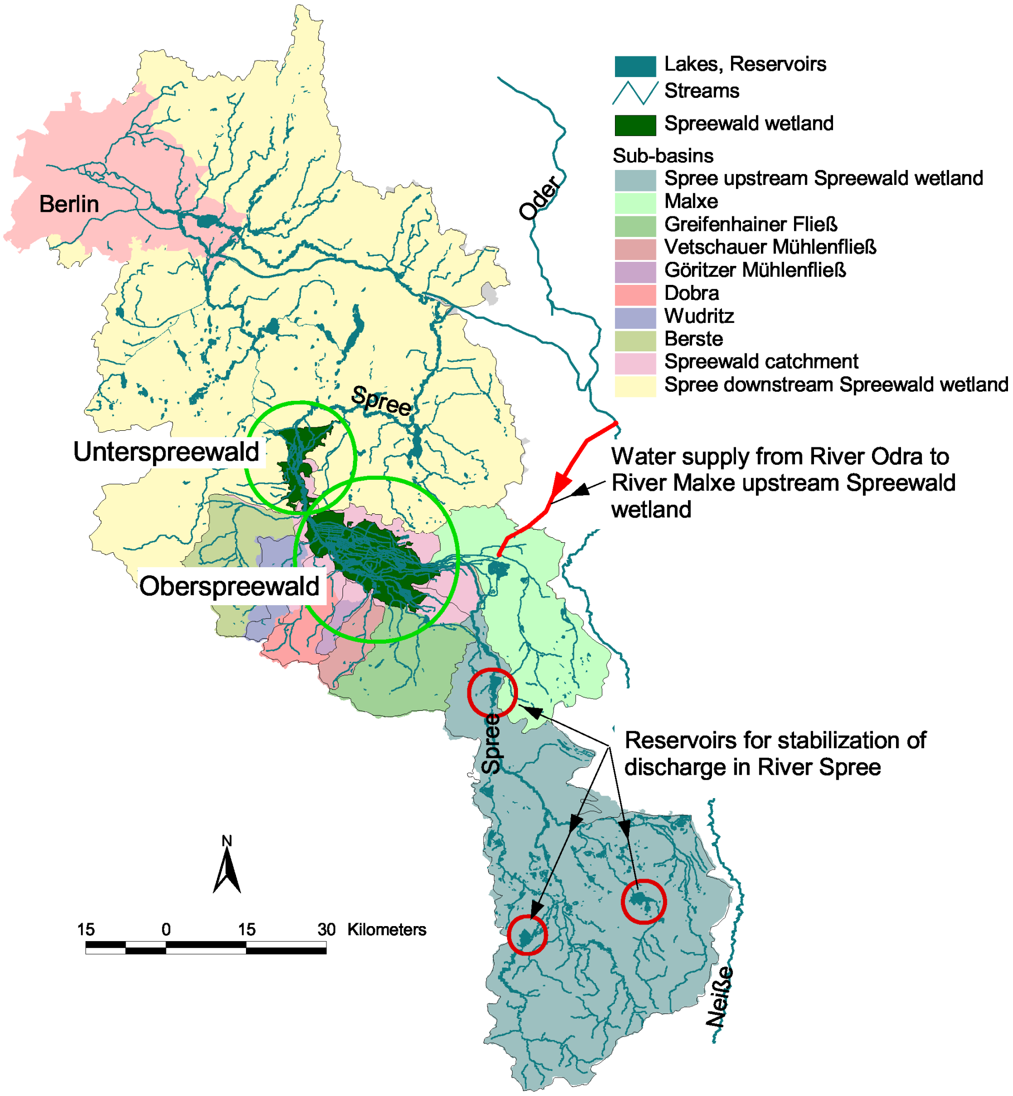

Figure 1: Location of the Spreewald within the Spree RiverBasin and of major infrastructure to augment water inflows to the wetland.

3 Methods

3.1 Conceptual approach: wetland functions, benefits and values.

In contrast to other water users, the water use of a wetland water regulation entity is multifunctional in that it regulates various ecosystem functions or processes at the same time. The concept of ecosystem services has become an important model to link functions of ecosystems to human welfare. The basis for an integrated ecological-economic evaluation is given by an analytical distinction between ecosystem functions, uses or benefits and values (Turner et al. 2008). Ecosystem functions in themselves have no economic value: the value is derived from the existence of a demand for the benefits (or uses) they give rise to. The wetland ecosystem functions give rise to ecosystem services that can be broadly categorized into hydrological, biogeochemical and ecological services. These services may give rise to benefits, either directly as final services or indirectly as intermediate services. The concept of total economic value is one of the most widely used approaches to systematically identify the various benefits that arise from wetland ecosystem functions. According to this approach, the total economic value is comprised of direct and indirect use values and of values that are independent of use.

Table 1: Characterisation of ecosystem service benefits from wetlands taken into consideration for the Spreewald case study.

Policy target	Indicators for hydrological-ecological function	Ecosystem service benefit	Public / private goods	Use / non use value	Valuation approach
Farm income generation	biomass energy yield	fodder production	private	direct use	change in net income
Safeguarding of recreational opportunities	navigability of canals	recreational benefit	public	direct use	travel cost method
Climate protection	greenhouse gas emissions	mitigation of climate change	public	indirect use	marginal abatement costs
Conservation of important wetland habitats	biotic development potential: area with high groundwater floor levels	non consumptive recreational benefits and non use benefits	public	non-use	stated preferences / benefit transfer

The total economic value framework is utilized to identify key benefits from ecosystem services that are relevant for water management decisions in the Spreewald basin. The selection of services to be included in an assessment of river basin management should reflect the relevant policy targets pertaining to the various wetland ecosystem services. For example, targets of rural development, agricultural and forestry policies, water resources and flood risk management, climate change and nature conservation policies are all affected by changes in water management of wetlands. Table 1 provides an overview of the ecosystem services and the underlying ecosystem functions and uses that were taken into consideration for this study. This study considers four ecosystem services: production of grassland biomass, navigability of waterways for recreational boating, regulation of greenhouse gas fluxes from peat and conservation of typical fen wetland habitats and biodiversity.

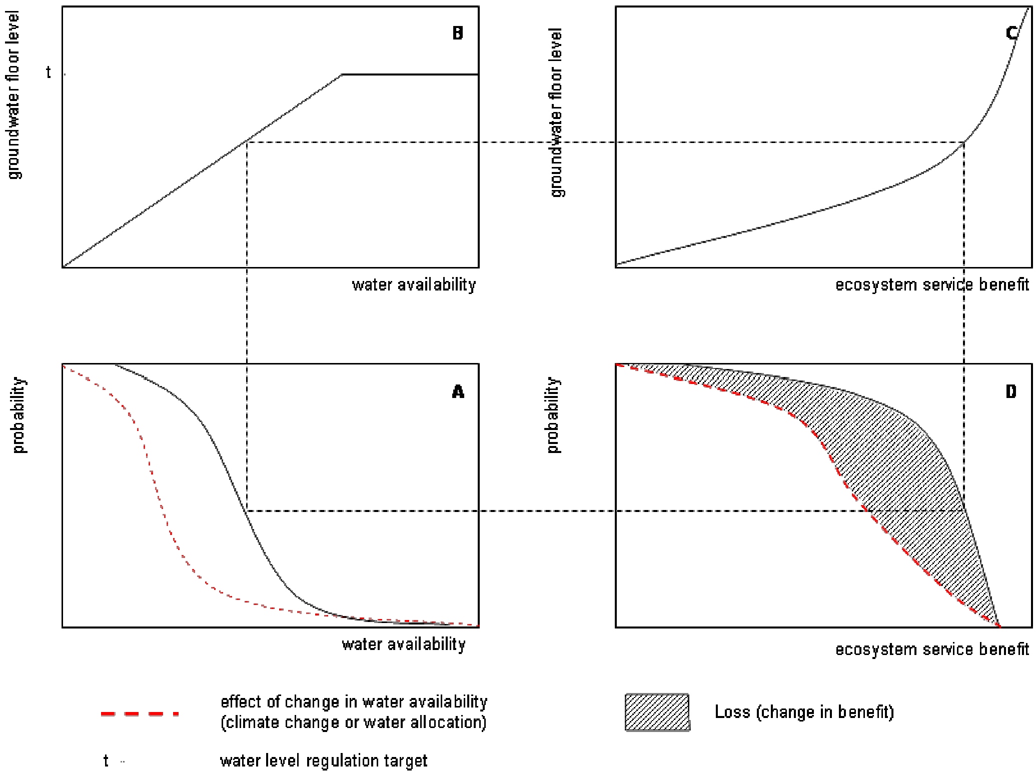

Figure 2: Conceptual approach to calculating the expectation value of wetland ecosystem service benefits for variable conditions of water availability.

Most of the ecosystem services provided by wetlands are joint products, in the sense that the services are produced jointly from the water provided to a wetland water regulation entity or hydrological response unit. The optimal water levels for different ecosystem services may be different, so that a trade off has to be made between different services when water regulation targets are defined. In order to systematically assess effects of water level regulation and water availability, groundwater-level dependent production functions for ecosystem services are combined with a spatially explicit hydrological model that describes wetland water levels. Figure 2 provides a conceptual illustration of the approach that is used to determine the benefits from ecosystem services as a function of water availability. The lower left quadrant (A) describes the cumulative distribution of the wetland water balance as a function of climate and basin water availability. This information is generated from climate and hydrological models for the river basin. The water balance is translated to groundwater levels below surface (quadrant B) by the wetlands hydrology sub-model integrated into the basin model. The water levels in turn are the key determinant for the ecosystem service production functions (quadrant C) that are integrated into the wetlands model. Following this procedure, the cumulative frequency distribution of water availability can be transformed to a cumulative probability function for the provision of ecosystem service benefits (quadrant D).

3.2 Water management model

The water resource modelling framework WBalMo for the simulation of water management for the Spree River Basin (WASY 2005, Kaltofen et al. 2004, Koch and Grünewald 2009). The model balances water abstraction by water users and water availability. These water users are characterised by their position in the river network, their monthly abstraction and return flow quantities and their priority ranking in relation to other users. Ecological minimum flows are also considered. Dietrich et al. (2007a, 2007b) improved the existing Spree River model by the implementation of a detailed sub-model for the Spreewald wetland in order to describe the complex water use process in this major wetland (). This model, the WBalMo Spreewald model, reduces the Spreewald's complex river network to the main water courses involved in distributing flows, as well as dividing the wetland into a total of 197 sub-areas whose groundwater levels can be regulated by means of weirs. A sub-area is defined as the smallest area in which the ground water levels can be regulated separately. An important assumption of the model is a horizontal ground water level in each sub-area. In the WBalMo Spreewald model, each sub-area is designated as a water user for which monthly water balances are calculated. The water balances of the sub-areas are modelled based on the underlying concept of hydrologic response units (HRU). The main soil types (peat, sand, and loam), a digital elevation model, land use and regulation sub-areas are blended by means of GIS to define the HRU's. The sum of hydrological responses from all the HRU's of a sub-area gives the reaction of the sub-area being investigated, and the sum of these gives the reaction of the whole wetland sub-basin. The information about the elevation distribution of the HRU's, a water storage curve and monthly target water levels of all sub-areas were prepared in a pre-processing process. The definition of target water levels for each HRU opens the possibility to simulate different management scenarios. Input data for precipitation, potential evapotranspiration and inflow from the sub-basins to the wetland are provided by the basin model WBalMo Spree (Kaltofen et al. 2004).

The model was calibrated and validated by the outflow downstream the Spreewald wetland (Nash and Sutcliffe 0.9), the water use of the wetland (Nash and Sutcliffe 0.7) and selected groundwater levels (difference between the median of observed and calculated levels lower 10 cm). The model calibration and validation is described in detail in Dietrich et al. (2007a).

At the same time it is possible to couple ecosystem processes with hydrological processes on the basis of the HRU concept. Mean depths to groundwater are derived for all HRU's for the every month of the simulation period. The provision of ecosystems services is therefore modelled by coupling the various production functions to the simulated groundwater levels below surface of each HRU.

The model uses a stochastic approach to calculate the respective probabilities for different water levels and levels of ecosystem service provision for every sub-area and aggregated wetland sites. This is done by simulating 100 statistical realisations of climate projections over a time period of 50 years.

The stochastic simulation framework was used to estimate the expected (or average annual) level of wetland ecosystem service provision E(ES) in year t under varying conditions of water availability as follows:

$$E(ES_t) = \sum_i \left[P_i \cdot (\sum_{wl} ES_{wl} \cdot A_{wl,i,t}) \right] \tag{1}$$

where P is the occurrence probability of a realisation i and where $\sum_{i=1}^{n} P_i = 1$ to ensure normalisation, wl is the water level and A_{wl} is the area of the spatial aggregation unit with an average annual water level of wl in realisation i, and ES_{wl} is the ecosystem service provision level as a function of water level wl.

3.3 Costs-benefit assessment model

This study considers management options that (a) affect water availability by either increasing the water supply by inter-basin water transfer or changing the allocation of water and that (b) involve a change in the water level regulation targets and land use for sub-areas of the wetlands. Changes in water level regulation targets are generally interdependent with changes in land use. Therefore a cost benefit assessment model was developed, that can accommodate for the long-term changes in ecosystem service benefits when moving from one water regulation target to a second and the short-term effects of changes in inter-annual water availability (flooding and drought). This approach is based on the basic framework for evaluating the benefits from changes in land drainage and water regulation for the enhancement of agricultural production (cf. Penning-Rowsell et al. 1986). However, the cost benefit model is extended to also include the positive and negative externalities of the land use system related to public ecosystem service benefits such as greenhouse gas regulation.

Following the standard with and without procedure which in sets the net discounted costs and benefits of each management option against the baseline management option, this view of the change of benefits from long-term or fundamental changes in wetland land use and water level regulation and changes in short-term or inter-annual availability of water can be summarised as follows:

$$\Delta B_t = \Delta tarB_t - \Delta E(L_t) \tag{2}$$

with

$$\Delta tarB_t = (tarB_t^m - tarB_t^b) \tag{3}$$

$$\Delta E(L_t) = E(L_t^m) - E(L_t^b) = (tarB_t^m - (E(aB_t^m)) - (tarB_t^b - E(aB_t^b)) \tag{4}$$

where B are the agricultural and other ecosystem services benefits, tarB is the target benefit at water level regulation target, E(aB) is the expectation value of the actual benefit under actual conditions of water availability for the measure m and baseline b. E(L) is then the expectation value of the average annual loss compared to the target water level. This formulation has advantages for the chosen modelling approach, because the integrated economic-hydrological model can be used for a direct dynamic estimation of ΔE(L). ΔtarB needs to be estimated on the basis of a static comparison of land use change scenarios.

The economic feasibility criterion for evaluating changes in wetland and basin water management then can be written as:

$$PVNB = \sum_{t=0}^{n}\left[\frac{\Delta B_t^{AGR}}{(1+r)^t}\right] + \sum_{t=0}^{n}\left[\frac{\Delta B_t^{oESB}}{(1+r)^t}\right] - \sum_{t=0}^{n}\left[\frac{\Delta C_t}{(1+r)^t}\right] - \sum_{t=0}^{n}\left[\frac{\Delta D_t}{(1+r)^t}\right] \tag{5}$$

Where PVNB is the present value of net benefits, t is a year during the schemes life, n is the expected life of the scheme, r is the discount rate, B^{agr} and B^{oESB} are the incremental benefit from agricultural and other ecosystem service benefits induced by changes in wetland or basin water and land management, C is the change in capital and operating costs for wetland and basin water management and D is the incremental disbenefit (forgone benefits or external costs) to other water using sectors in the basin. In this study, three other ecosystem service benefit dimensions are taken into consideration, namely recreation, greenhouse gas regulation and biodiversity and habitat conservation.

3.4 Valuation of wetland ecosystem services

Grassland biomass production

The change in net income method is used to value changes in benefits from agricultural land use. Three valuation cases can be differentiated. The first two cases are related to permanent changes of the land use as result of changes to the water level regulation targets (ΔtarB) within the wetland. The third valuation case relates to the inter-annual variation of water availability and the associated one-off losses (ΔL) of agricultural output compared to the expected output at target water levels.

As a consequence of wetland restoration, land may either be permanently be lost to agricultural production or may require a change to the agricultural production system. In the first case land ceases to have any value for current agricultural use and the long term loss in agricultural benefit (land rent) is valued using the market price difference between land of the current quality and rewetted wetland land. In the second case, permanent adjustments are required to the agricultural production system. These are valued at the change in net margin. The change in net margin is determined as the change in gross margin (residual of gross income less direct costs for variable inputs) less any changes in fixed costs (labour, land, building and machinery costs). For this study, the compensation payments offered under agri-environmental schemes for changes in land management are used as proxies (EPLR BB 2007). The payments are generally calculated to compensate the change in net margin of typical farm enterprises in the region and include an incentive component to cover transaction and risk costs.

In the third case of one-off losses of agricultural output due to water shortages relative to the regulation target, the loss in gross margin from a production activity is used to reflect the loss to the farm enterprise. The loss in gross margin is calculated as the value of the lost output less any savings of variable costs from reduced harvest and storage costs. As the energy yield of biomass produced for fodder is not directly tradable in the market, a substitute price is used. It is assumed that a deficit in energy yield of the fodder grown in the wetland during dry years is compensated by maize from the arable land outside the wetland that would alternatively have been used for biogas production. The loss in energy yield is then valued at the price of maize biomass for biogas production as follows:

$$L_t^{BIOMASS} = \sum_{crop} ((tarYE_{crop} - aYE_{crop,t}) \cdot SED \cdot P_t) - ((tarY_{crop} - aY_{crop,t}) \cdot \Delta VC) \tag{6}$$

Where YE is the annual metabolic energy yield in MJ ME, Y is the annual biomass yield in dry matter (DM) in dt at target (tar) and actual (a) water levels, SED is the energy density of the substitute crop's dry matter in MJ ME $dtDM^{-1}$, P is the price of maize biomass for biogas production in € $dtDM^{-1}$ and ΔVC is the saving in variable harvest costs in € dt DM^{-1}.

Whilst the energy density and maximum potential biomass yield is an input to the model and is defined for each crop production system, the actual biomass yield is determined as a function of average annual water levels during the vegetation period. The annual energy yield is then calculated as follows:

$$YE = \sum_{soil} \sum_{wl} \sum_{crop} Y \max_{crop,soil} \cdot \left[\frac{ETa}{ETp} \right]_{wl,crop,soil} \cdot R_{wl,soil} \cdot ED_{crop} \cdot A_{crop,soil,wl} \tag{7}$$

where YE is the annual energy yield in MJ, Ymax is maximum attainable biomass yield under current management with no water stress in dt ha^{-1}, ETa / ETp is the water stress factor (actual to potential evapotranspiration of a crop), ED is the energy density of a crop in MJ ME dt^{-1}, R is a water logging factor and A is the area of crop in ha. Indexes denote soil, crop and water level (wl).

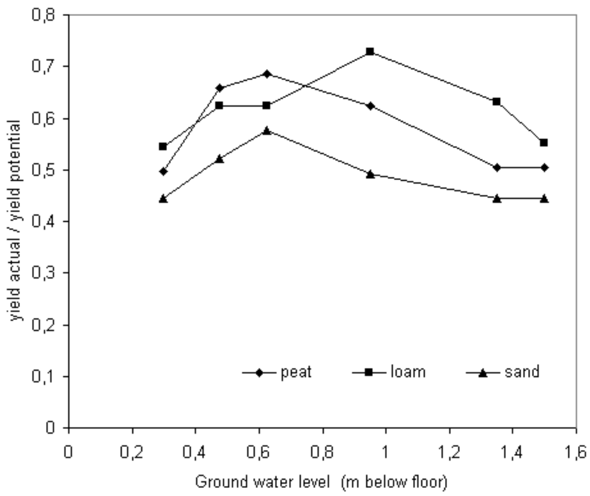

Figure 3: Biomass yield from agricultural crops as a function of mean annual water levels for different soils (based on data from Lorenz et al. 2008)

For the current application, the water stress factors and water logging factors as a function of water levels are taken from specific values developed by Lorenz et al. (2008) for the Spreewald based on a modelling approach developed by Wessolek et al. (1987). The resultant trajectory of relative yields as a function of water levels is shown in Figure 3.

Table 2: Parameters of the cropping systems

Land use	Crop	Target WL*	Maximum yield per soil type			Energy density	N / P fertilizer
		m	dtDM ha^{-1}			MJ ME $kgDM^{-1}$	kg N / P2O2 ha^{-1}
			peat	loam	sandy loam		
Arable	Maize	>0.45	130	140	120	10,8	150 / 70
Grassland	High intensity	>0.45	80	90	70	10	120 / 40
Grassland	Low intensity	>0.45	70	80	60	8	0 / 20
Grassland	Low intensity wet	0.45 – 0.20	50	60	40	6	0 / 0
Grassland	Conservation / reeds	<0.20	35	45	25	n.a.	0 / 0

* average annual water level below ground

Five cropping systems are differentiated on the basis of a combination of land use data and water table levels. Arable land is assumed to be planted with the dominant crop, which is maize. Grassland is classified into four subtypes based on target water levels and additional information on the land use intensity: high intensity grassland (groundwater levels > 0.45 m), low intensity grassland (groundwater levels > 0.45 m), low intensity wet grassland (groundwater levels < 0.45 and > 0.2 m) and conservation grassland / reeds (groundwater levels < 0.2 m). The production activities and the target water levels are interdependent, so that a change in target water levels requires changes to the production system. Table 2 summarises the parameters of the cropping systems.

Recreational boating

The results of an application of the travel cost method are used to estimate the recreational benefit of visitors participating in punting boat trips in the Spreewald. The basic assumption of the travel cost method is that the costs incurred by individuals travelling to participate in recreational boating can be used to derive an estimate of the consumer surplus derived from a visit to the site (cf. Eiswerth et al. 2000, Haab and McConell 2002). To calculate the effect of a change in water availability on the recreational value of a site, three basic types of information are required: the number of visits, the recreational value of a visit, and the change in both variables under a change in navigability.

The effect of reduced flows on recreational boating in the Spreewald is modelled as a disruption of the longer boating routes that require passing certain locks, because the water level in the locks is not sufficient to allow boats to pass. Monthly water levels in the locks are calculated on the basis of the wetland groundwater levels. A lock is considered impassable when the water level is lower than the required minimal draught of approximately 0.3 m. Boating is considered to be disrupted, if the number of affected locks is higher than the threshold value (set at 20 %). The share of the season (sS) that boating in the Spreewald is disrupted by low flows in a simulation year t is then calculated as follows:

$$sS = \left(\sum_m if\left(\sum_l if(nD_l - dWL_{m,l} <= mD;1;0) > TH;1;0 \right) \right) \Big/ S \tag{8}$$

where subscripts l denote locks, nD is the nominal depth of water in the lock under target conditions in meters, dWL is the difference of actual water level in a month m from target water level in meters, mD is the minimum required depth in meters, TH is an evaluative threshold of the number of locks above which the whole system is considered to be disrupted, m is a month of the season, S is the duration of the season in month.

The annual recreational value B^{REC} of punting trips in the Spreewald is then calculated as a function of the share of the season with low flows (sS) in year t as follows:

$$B_t^{rec} = (sS_t \cdot X_t^* \cdot sv \cdot CS) + ((1 - sS_t) \cdot X_t^* \cdot CS) \tag{9}$$

where CS is the consumer surplus per person per trip in €, sS is the share of the season with trip limitations, sv is the share of visitors that would not have come to the Spreewald at low flows and X* is the estimated total number of participants in punt trips per year.

The estimated total number of participants in punt trips is approx. 1 million per year. The effect of various scenarios of a change in navigability on the percentage of visitors that would no longer come to the Spreewald and the recreational value per trip was determined based on the results of an interview survey. A survey of 750 tourists that took part in punting boat trips was carried out. Details of the study are presented in Grossmann (2011). The share of visitors that would not have come to the Spreewald at low flows is estimated at 45 %. The consumer surplus (welfare measure) per person and punting trip was estimated to be 19 €.

Regulation of greenhouse gas flux

Estimates of the shadow price of carbon are used to value the greenhouse gas emission externalities associated with the wetland water management. Peat carbon sequestration is a result of low biomass decay rates under anaerobic conditions in water logged wetland sites. When wetlands are drained the peat is no longer conserved but decomposed, because lowering the water table stimulates aerobic decomposition of the peat. The most important effect of rewetting degraded peatlands is therefore not a reactivation of carbon sequestration but the avoidance of carbon emissions from peat oxidation.

A growing number of governments have started to use a shadow price of carbon to value the externality from greenhouse gas emissions (or the benefits of abatement) which needs to be incorporated into cost-benefit and policy appraisal (cf. DECC 2009, UBA 2007). The rational is to make policy and investment decisions across sectors comparable. There are basically two approaches to defining a shadow price of carbon, either based on the marginal damage costs of carbon or the marginal abatement costs. Due to uncertainties regarding the damage costs (Tol 2009), a climate policy target consistent approach that is based on estimates of the abatement costs of meeting a specific reduction target for a political jurisdiction is increasingly considered to be more appropriate for project appraisals (Tol and Lyons 2008, DECC 2009). This paper uses the price projections for the EU Emissions Trading Scheme (ETS) traded sector provided in the DECC (2009) guidelines. The price of carbon is projected to rise from an initial 26 € $tCO2^{-1}$ in 2008 to 30 € $tCO2^{-1}$ in 2020.

The main drivers controlling greenhouse gas fluxes of fen wetlands are largely related to aspects of hydrology. In order to estimate the greenhouse gas flux from wetlands a method presented in Grossmann and Dietrich (accepted) is used. This method combines water level dependent emission functions for the major greenhouse gases to estimate the global warming potential (GWP) measured in carbon dioxide equivalents. The GWP balance of the greenhouse gas exchange for wetlands is calculated as:

$$GWP_{CO2e} = (NEE_{CO2-C} \cdot GWP_{CO2-C} + F_{CH4-C} \cdot GWP_{CH4-C} + F_{N2O-N} \cdot GWP_{N2O-N}) \cdot 44/12 \tag{10}$$

where GWP is measured in CO2e, NEE_{CO2-C} is the net ecosystem exchange of carbon measured in CO2-C, F_{CH4-C} is the mean annual flux of methane measured in CH4-C, F_{N2O-N} is the mean annual flux of nitrous oxide measured in N2O-N and GWP are the corresponding elementary global warming potentials. The factor 44/12 converts from Ce to CO2e which is the usual accounting unit. The GWP balance is an atmospheric balance, so that emissions from wetlands have a positive and sinks a negative sign.

Carbon sequestration by carbon accumulation occurs at high water levels under anaerobic conditions. The difference between the carbon fixation by photosynthesis and respiration from the ecosystem is the net ecosystem exchange (NEE). The carbon surplus gained in the process is available for long term carbon accumulation. However a share is emitted again by anaerobic respiration as methane (CH4). Under aerobic conditions carbon is no longer accumulated in peat but emitted as a result of peat degradation. Simplifying, the NEE flux into the atmosphere is estimated as follows:

$$NEE_{CO2-C} = \Delta C - F_{CH4-C} \tag{11}$$

and

$$\Delta C = LORCA_{CO2-C} \text{ for WL < 10 cm below ground} \tag{11a}$$

$$\Delta C = F_{CO2-C} \text{ for WL > 10 cm below ground} \tag{11b}$$

where ΔC is the net change in soil carbon storage[1], F_{CH4-C} is the methane carbon flux, $LORCA_{CO2-C}$ is the long term rate of carbon accumulation and F_{CO2-C} are the soil carbon

[1] Note that NEE and ΔC are viewed from the atmospheric carbon balance, so that LORCA has a negative sign and F_{CH4-C} and F_{CO2-C} have a positive sign.

emissions from peat degradation. Water level dependent emission functions for each component from the literature are used and these are summarized in Table 3.

The annual benefit from the greenhouse gas regulation function of fen soils in year t is then calculated as:

$$B_t^{GHG} = -SPC_t \cdot \sum_{wl} GWP_{wl} \cdot A_{wl} \tag{12}$$

where SPC is the shadow price of carbon in € $tCO2e^{-1}$, GWP is the annual global warming potential in tCO2e ha^{-1} at water level wl and and A is the area in ha with water level wl.

Table 3: Water level dependant functions used to estimate greenhouse gas balance of fen peat soils.

GHG balance component		Valid for water level range	Unit	Equations [a]	Source
Longterm rate of carbon accumulation	$LORCA_{CO2\text{-}C}$	-10 - 10 cm	CO_2–C kg^{-1} ha^{-1}	(0.143*(WL-10))*CD*10*CC [b]	Blankenburg et al. 2001
Soil carbon emissions	$F_{CO2\text{-}C}$	10 – 150 cm	CO_2-C kg^{-1} ha^{-1}	(WL*121)-(0.482*WL^2) -121	Renger et al. 2002
Methane emissions [a]	$F_{CH4\text{-}C}$	-10 – 150 cm	CH_4-C kg^{-1} ha^{-1}	EXP(3.57-0.08*WL)*10 * CF	Van den Pol - van Dasselaar et al. 1999
Nitrous oxide emissions	$F_{N20\text{-}N}$	-10 -150 cm	N_2O-N kg^{-1} ha^{-1}	IF(WL>20;8;0)	Höper 2007

[a] WL: water level below ground in cm

[b] CD: peat density in peat of 80 g l^{-1} and CC: carbon content fraction of 0.43

[c] for the high impact trajectory we apply a CH4 emissions estimate reduced by 50 % compared to reference function

[d] CF: we use a conversion factor of 12/16 to convert from CH4 to CH4-C.

* We use following GWP: GWPCO2-C =1, GWPCH4-C = 7.6 and GWPN2O-N = 133.

Habitat and biodiversity conservation

The values attached by the general public to the conservation of wetland biodiversity and habitats are potentially considerable; however they are also extremely difficult to measure. Stated preference methods, such as contingent valuation or choice experiments, are the only techniques suited to derive estimates of the general public's preferences for habitat and biodiversity conservation that include non-use value components (cf. Bateman et al. 2002, Haab and McConnell 2002). Non- use values derive from preserving natural heritage for future generations independent of any personal use of a site for example for recreation. However, in practice it is difficult to separate non-use values from non consumptive use values, because improvements of the wetland habitat and biodiversity quality may also improve the recreational use value of that part of the population that uses the wetland areas for recreational purposes.

All of the stated preference methods require interview surveys to elicit primary value estimates for a proposed policy or measure. This paper uses benefit-transfer from stated preference studies to estimate the general public's willingness-to-pay for the conservation of natural wetland habitats and biodiversity. The transfer of benefit estimates either by direct value transfer or functional transfer can be used as an alternative when no primary studies for the site of interest can be carried out (cf. Bergstrom and Taylor, 2006 and Nelson and Kennedy, 2009 for theoretical and methodological overviews). This study uses a meta-functional transfer method that is based on the systematic quantitative summary of evidence across empirical studies. It is based on a meta-analysis of European valuation studies for protection and restoration of terrestrial wetlands from the European Union (Grossmann in review). The transfer function is used to estimate per household willingness to pay, using policy site specific parameters for the assumed market area (population) and the area of the wetland conservation programme. The spatial extent of the population that is considered to hold a value for the site is an important modifying factor, because distance decay effects may be important, whereby marginal willingness to pay decreases as distance from the wetland increases. It is further expected that marginal willingness-to-pay decreases with the size of the restoration programme. The Spreewald is considered to be at least of regional importance, so for a lower bound estimate the population from the Federal States of Brandenburg and Berlin are defined to constitute the market area. The conservation efforts in the Spreewald are considered to be a part of regional wetland conservation effort that targets all of the approx. 150 000 ha of fen wetland sites in the State of Brandenburg equally. These assumptions yield an annual household WTP of 8.5 $€_{2005}$. Assuming a population of 5.95 Mio inhabitants and an average household size of 2.2, the aggregated total annual WTP is 23 Mio € or of 1535 € ha^{-1} for the reference wetland area.

Effects of water management on habitat quality are estimated using the area of wetland sites with an average annual water level less than 20 cm below surface as an indicator. This water level corresponds to the minimum requirement for wetland conservation as set out in the regional wetland management strategy (LUA 1997). Fen wetlands with lower water levels may be considered to be degenerating and will ultimately loose their wetland status. The annual benefit from wetland habitat and biodiversity conservation B^{HBC} in year t is then calculated as follows:

$$B_t^{HBC} = Awet_t \cdot WTP \qquad (11)$$

where Awet is the area of fen wetland with an average annual groundwater level < 20 cm below surface in ha and WTP is the aggregate willingness to pay for wetland habitat conservation in € per ha.

4 Water management options

For this paper six management options are analysed, that result from a combination of two management options in the upstream headwaters of the Spree River with three management options within the Spreewald wetland (Table 4).

Table 4: Definition of management options by combination of different water resources management options for the basin and the wetland

Name	Basin management option	Wetland management option
Baseline	Current long term management strategy	Current management
Fen protection	Current long term management strategy	Fen protection
Redistribution.	Current long term management strategy	Redistribution
Transfer	Current long term management strategy with water transfer from Odra River	Current management
Fen protection + Transfer	Current long term management strategy with water transfer from Odra River	Fen protection
Redistribution +Transfer	Current long term management strategy with water transfer from Odra River	Redistribution

All management options are analysed for a projection of future climatic conditions and water availability. This baseline projection and methods used for downscaling are described in detail in Wechsung et al. (2008). The scenario is based on a mean temperature increases of 1.4 K by 2050 compared to 1960-1990. Key effects for the Spreewald region are a shift in intra-annual precipitation distribution. While there is a reduction of precipitation in summer, winter precipitation increases slightly. In combination with an increase in potential evapotranspiration this leads to an increased deficit in the climatic water balance in the summer half of the year.

Two basin water management options are considered. The baseline option represents the current management practice, in which the augmentation of river flows in dry periods from reservoirs at Bautzen, Quitzdorf and Spremberg is the main approach. This baseline assumes a phasing out of open cast lignite mining in the headwaters of the Spree River by 2030. The pumping of mine water into the river will stop and additional water will be needed to refill the remaining pits. A transfer of water from the Odra River to the Malxe

River upstream of the Spreewald (Fig. 1) is an additional alternative management option to increase water supply to the Spreewald. The water could be taken from the Odra River at Ratzdorf and transported by the means of a 30 km canal to Peitz on the Malxe River. The maximum transfer rate is projected to be 2 $m^3 s^{-1}$. The total costs of this measure are projected to entail investment costs of roughly 30.4 Mio € with a lifetime of 50 years and annual maintenance costs of 1 %. The cost for pumping are a function of the transferred water volume with an energy requirement of 0.45 kWh m^{-3} at electricity costs of 0.0225 € kwh^{-1}.

Water level regulation and distribution of flows within the wetland are the key elements of water management options within the wetland. Water demand can be lowered by lowering the target water tables, but this would be detrimental to wetland ecosystem functions. Higher water levels in winter combined with a later lowering in spring can help to store water in wetland soils and contribute to reduce summer water deficits. This would be associated with benefits for fen wetland restoration. The distribution of flows within the wetland can be regulated to prioritise the water supply of certain sub-areas, for example of high production or conservation value.

Three wetland water management options are considered. The first option replicates the current management approach in terms of water level regulation targets and distribution of inflows between the sub-areas (cf. LUA 2002). The second option entails a redistribution of inflows from the rivers Spree and Malxe. The flow of the Malxe River is expected to decrease significantly by 2050 because it is to a large extent fed by groundwater pumped from active open cast mining. This will lead to a reduction of water tables in the northern section of the central Upper Spreewald, which is currently exclusively fed by water from the Malxe River. This option proposes to divert part of the Spree inflows into the northern flood canal and to use this water to augment water supply of this area. There are no additional water management costs associated with this option.

The third option builds on the targets of fen conservation programmes and the Landscape Framework Plan for the Biosphere Reserve (MUNR 1998). In this management option two types of conservation targets according to the guidelines for fen conservation in Brandenburg (LUA 1997) were identified: (a) conservation/restoration of fen habitats with target groundwater levels of 10 cm above floor in winter and 20 cm below in summer, and (b) stabilisation of fen habitats with target groundwater levels at floor level in winter and 30 cm below floor in summer (cf. Fig. 4). The target area for conservation and restoration covers an area of 4,000 ha in the scenario. The required changes in water levels would exclude any further agricultural land use. In total it would be necessary to convert 1,700 ha of grassland currently used for agricultural production. The fenland stabilisation development goal covers a further 3,800 ha, 2,000 ha of which are currently under agricultural land use and that would require compensation payments to offset losses of net margin from changes in land management. A purchase price for low quality grassland of 2,800 € ha^{-1} and compensation payments for maintaining near-surface groundwater levels up to 30th May of € 100 ha^{-1} and up to 30th June of € 200 ha^{-1} are assumed (EPLR 2007). Besides the loss in agricultural benefits, the fen restoration measures entail costs for changes in water management. The investment costs for restructuring the water infrastructure in the restored areas is estimated to be roughly 1,500 € ha^{-1} and the annual regulation costs are reduced by a third from ca. 10 to 7 € $ha^{-1}a^{-1}$. There are no changes in regulation costs for the stabilisation target..

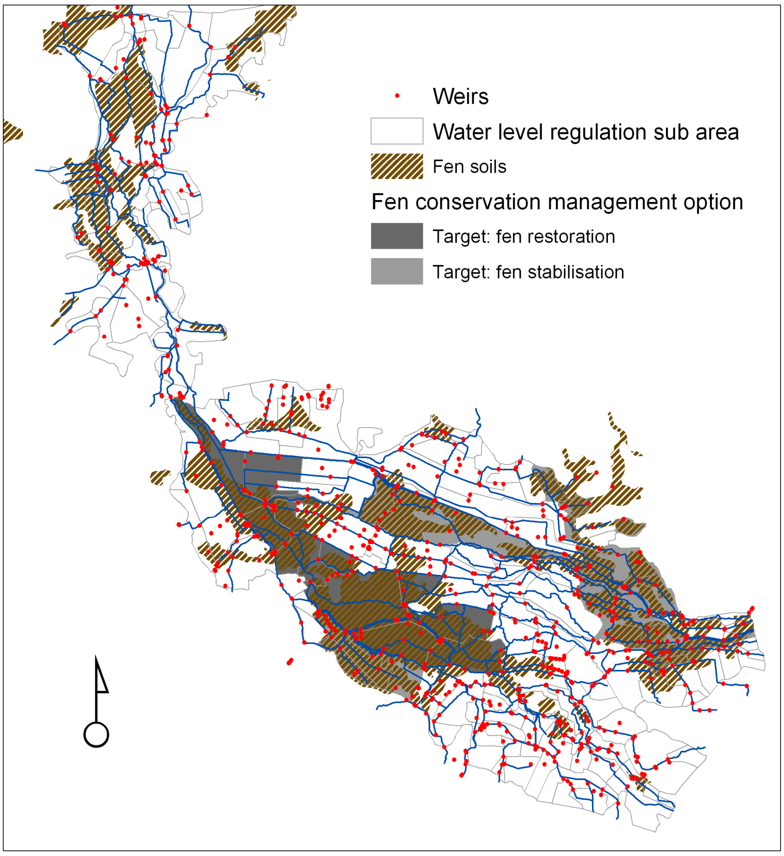

Figure 4: Map of the Spreewald showing fen soil areas and target areas for the fen conservation management option

The economic valuation is based on a Monte-Carlo simulation using a +/- 50 % variation around central estimate of the value estimates for different ecosystem services and a +/- 25 % range for costs. Table 5 provides a summary of both the range and the criteria according to which the range is specified. Throughout we use a social discount rate of 3 % and an appraisal period of 50 years. The price base is 2008.

Table 5: Uncertainty range for key value estimates for benefits from ecosystem services.

Variable		Unit	Value range			Source
			Min	Central	Max	
Shadow price of carbon	SPC	€ $tCO2e^{-1}$	15	30	45	DECC 2009
Willingness to pay for habitat conservation	WTP	€ HH^{-1} a^{-1} / € ha^{-1} a^{-1}	4.3 / 767	8.5 / 1535	12.8 / 2302	Grossmann 2010 in review
Consumer surplus from recreational boating	CS	€ $visit^{-1}$	9.9	19.8	29.7	Grossmann 2010
Loss in gross revenue from biogas maize substitute / saved variable costs for grass harvest	ΔGR / ΔVC	€$dtDM^{-1}$ / €$dtDM^{-1}$	2.7 / 1.5	5.4 / 2	8.1 / 2.5	KTBL 2006, Neubert 2006

5 Results

5.1 Effects of water management on wetland groundwater levels

Under the climate change scenario that is used for the projections in this study, the average inflow from the basin in the summer months will decrease in 2050 but the variability of the inflow will increase (Fig. 5). These effects are illustrated using the monthly median value and the 20^{th} and 80^{th} percentile values representing dry and wet years. The percentiles are based on 500 simulated years in each of the 5 year-periods. The reasons for this decrease are the changes in climatic conditions as well as the projected changes in the open cast mining activities in the basin. In particular, the inflow from the Malxe River will decrease in the future because there are two large opencast mines in this sub-basin that are to stop operations by 2030. The water transfer from the Odra River into the Malxe River can improve the water supply situation. However the volume under this scenario is not sufficient to compensate for the long term decrease in inflows. Frequent high and low extremes of water availability were already observed in the last decade in many northeast German river basins and the projection of future climatic conditions indicate that the frequency of extremes in the water balance will increase.

In all scenarios there is an increase in water withdrawal from July to September because of the increased water demand. However the withdrawal is limited by the availability of water inflow, technical limitations in the existing water regulation system and flow requirements below the wetland. Under the climate change scenario the water deficit will increase. The consequences are decreasing groundwater levels in the wetland and decreasing discharge in the Spree River below the wetland. Especially in dry years the increasing water demand cannot be fulfilled because of the limited water inflow. This is the main reason for the slightly decreasing range of the water withdrawal in 2050.

The analysis presented in Figure 5 characterises the variability of water balance that is based on the simulation of 100 realisations of the climatic projection. The following hydrological analysis focuses on the median conditions (50th percentile). The economic analysis is based on the expectation value, which integrates over the whole range of conditions.

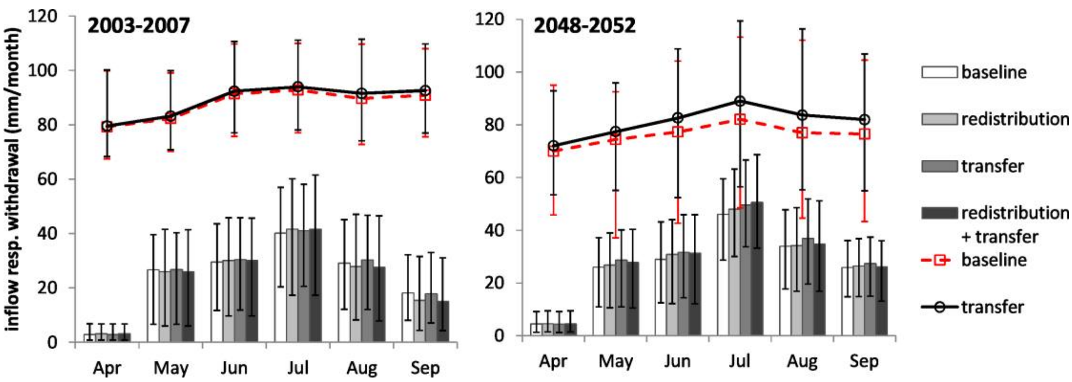

Figure 5 Simulated median values of water inflow (line) and water withdrawal of wetland sub-areas from inflows (bar) in the periods 2003-2007 and 2048-2052 for baseline, water transfer and redistribution management option. The values are medians referred to the total wetland area with ranges representing 20th and 80th percentile.

While groundwater levels are very good indicators of the water balance, groundwater levels below surface are key determinants of ecosystem functions. The key effects of the management options on groundwater levels are shown in Figure 6. There are little changes in mean water levels from the first to the last simulation period to be expected in the central areas of the Spreewald (Fig. 6-A). The central areas are predominantly supplied with water from the main Spree River. However, the areas of the Upper Spreewald that are predominantly supplied with water from the Malxe River, have decreasing groundwater levels. In general, the water levels along the margins of the wetland will fall because these areas receive water inflow from small stream catchments. It is difficult or even impossible to transfer water from the main inflow of the Spree River to these areas.

The higher target water levels for fen soils under the fen conservation target lead to an improvement of the water levels in the targeted areas (Fig. 6-B). All other areas are not affected. In principle this option undertakes an intra-annual temporal transfer of water. In times of water surplus, water is retained in the soil. In the summer higher water levels lead to increased evapo-transpiration but this can be compensated by the additional stored water.

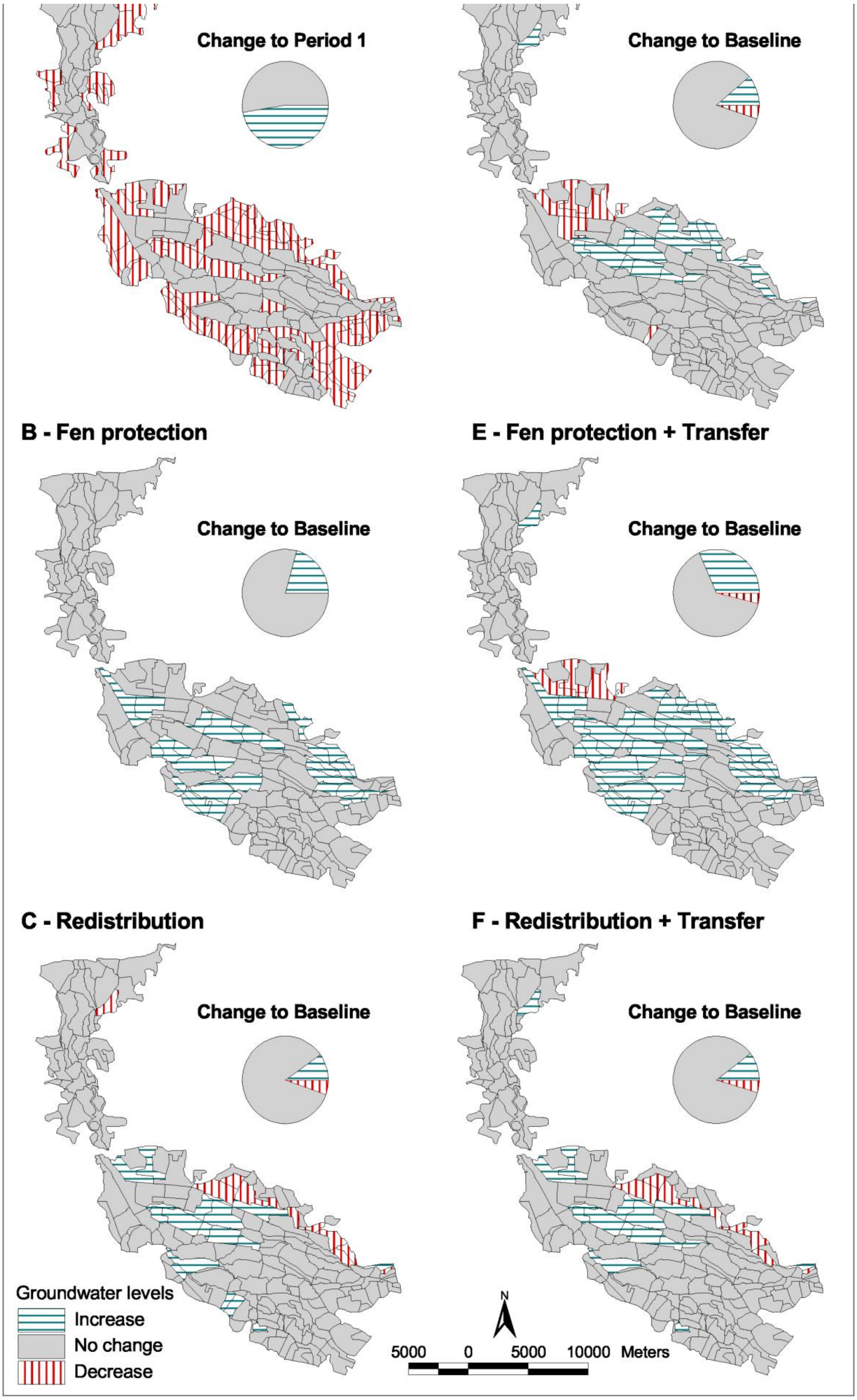
Change to Period 1
Change to Baseline
B - Fen protection
Change to Baseline
E - Fen protection + Transfer
Change to Baseline
C - Redistribution
Change to Baseline
F - Redistribution + Transfer
Change to Baseline
Groundwater levels
Increase
No change
Decrease
N
5000 0 5000 10000 Meters

Figure 6: Water table regulated sub-areas of the Spreewald with higher, lower or unchanged ground water levels in July for average years (median). The simulation period 2048-2052 for the baseline is compared (A) to the baseline in 2003-2007 and (D-F) to the five management options in 2048-2052. The pie charts show the corresponding percentages.

The redistribution of water from the main Spree River to the northern area of the central Upper Spreewald, which is the most important area for nature conservation, increases the water levels in the targeted areas. However the improvements come at the cost of increasingly lower water tables in marginal areas of the Upper Spreewald (Fig. 6-C).

The three options with transfer of additional water to the Malxe River increase the share of area with higher groundwater levels (Fig. 6-D to 6-F). Because the total transfer volume is limited the additional water in first line benefits those areas that are upstream on the Malxe inflow. In some cases downstream areas may receive less water than without the transfer, because the higher water levels of the well supplied areas greatly increases the evapotranspiration. In the combination with fen protection, the water transfer induces additional increases in water levels over a large area. In contrast there are no improvements in combination with the redistribution option, because the additional water supply is directed towards the central Upper Spreewald that is already advantaged by the redistribution. The additional water therefore is passed on downstream.

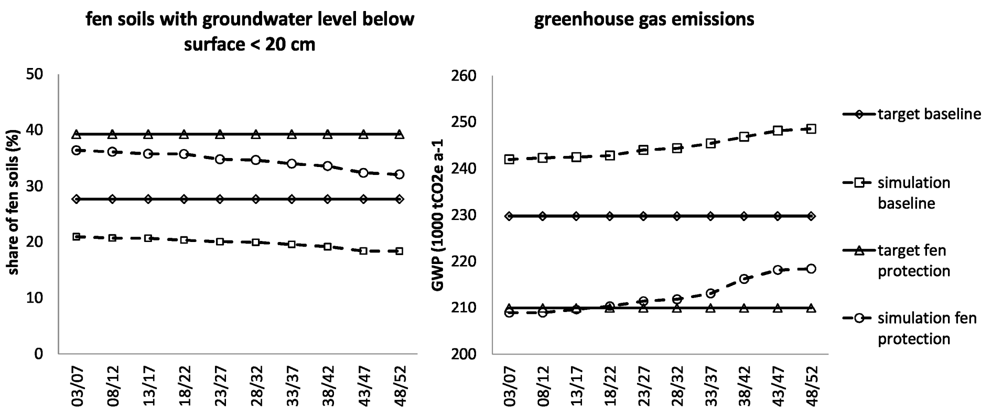

Figure 7: Effects of changes of regulation target and land use on the average share of fen soils with groundwater levels below surface < 20 cm (left) and the annual greenhouse gas emissions (right) for the baseline and the fen protection management option over the simulation period.

5.2 Effects of water management on wetland ecosystem functions

The change of water level regulation targets and land use in the restoration option is associated with benefits from an increase of the area of typical wetland habitats and reduced greenhouse gas emissions (Fig. 7). At the current water level regulation targets, 28 % of the total fen soil area is targeted to have ground water levels higher than the minimum water level requirement to safeguard for fen wetland soils. Under the fen protection management option this would increase to 39 %. The change of regulation target could potentially reduce the targeted negative externality from carbon emissions by approx. 0.02 Mio $tCO2e\ a^{-1}$. The design of the fen protection management option therefore only targets a 10 % reduction of the total emissions. The remaining 60 % of fen soils with lower water targets continue to be subject to subsidence.

5.3 Effects of water management on economic losses

The effects of the water management options on the average annual loss (cf. eq. 2 and 4) from water deficits compared to the water level regulation targets for the baseline and the fen protection management options are shown in Figure 8. The results illustrate that the water availability does not suffice to maintain the targeted water levels under current conditions already at the beginning of the simulation period. This leads to higher social costs from greenhouse gas emissions and reduced wetland habitat conservation benefits then targeted. Lower water tables may however have positive effects on agricultural benefits in areas where the target water levels are higher than optimum levels for agricultural biomass production (cf. Fig. 3). The balance of areas that benefit from lower water levels and those that experience losses from water levels that fall below the optimal levels for production determines the aggregate effect shown in Figure 8.

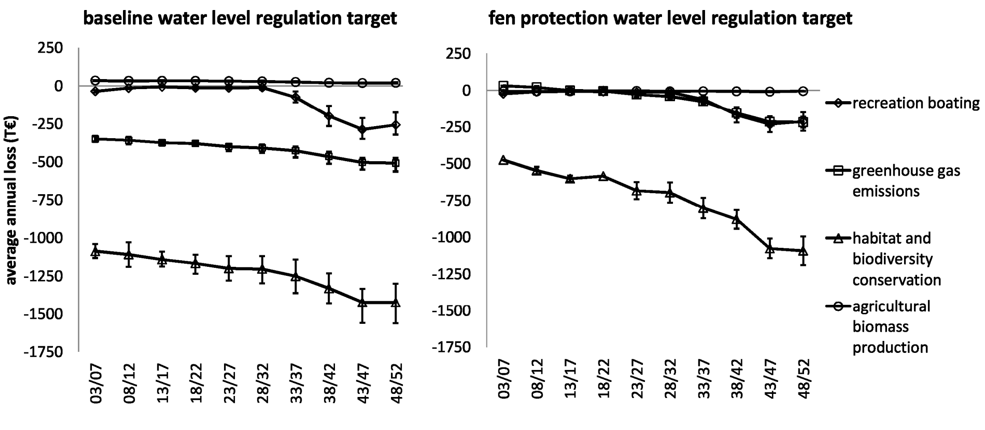

Figure 8 Mean and range of average annual loss compared to the water level regulation targets for the baseline (left: baseline, redistribution, transfer, redistribution + transfer) and fen protection (right: fen protection, fen protection + transfer) over the simulation periods 2003/52 for recreational boating, greenhouse gas emissions, habitat and biodiversity conservation and agricultural biomass production.

The emerging general picture is that under current water level regulation targets, the insufficient water supply has opportunity costs regarding public environmental goods and benefits regarding agricultural production. The projected completion of additional upstream water storage infrastructure will lead to an initial improvement in water availability compared to the current situation. However, from the middle of the simulation period the negative effects of climatic change and reduced mining water on water availability will lead to increased loss of ecosystem service benefits. This is particularly clear for the recreational boating, but the trend is apparent for all considered ecosystem services. The magnitude of the aggregated average annual loss increases from the first to the last simulation period by ca. 0.5 - 0.9 Mio € a^{-1}, which is equivalent to ca. 16 - 28 € ha^{-1} a^{-1} for the total wetland area. A change of the regulation target as proposed under the fen protection option leads to a reduction of the average annual losses from recreation, greenhouse gas emissions and habitat conservation but not from agriculture. However the reduction of average annual losses comes at the opportunity costs of changes to the agricultural land use.

5.4 Cost-benefit analysis of water management options

Finally, the net present value (NPV) of the water management options is calculated according to equation (2-5), taking both the change in average annual loss (ΔL), the change in benefits from a change in regulation target ($\Delta tarB$) and the change in water management costs (ΔC) into account. Table 6 summarises the present value for each ecosystem service benefit and water management costs separately.

The NPV estimates presented in Figure 9 are generated using a Monte-Carlo simulation over the range of price assumptions. The NPV estimates therefore consider both the risk of low flows in the expectation value of ecosystem service production and the uncertainty regarding the economic value. The aggregate results show that economically the most efficient option is the fen protection management option. The costs of mobilizing additional water by transfer from the Odra River, if fully apportioned to the Spreewald, are not justified by the generated benefits except possibly in combination with the fen protection target. The water transfer option must therefore be considered economically disadvantageous. There may however be some additional benefits downstream of the Spreewald that are not considered here. The internal redistribution option does not generate any significant net benefits. The standard deviation and range of the NPV estimates indicates that the results in terms of ranking are stable over the range of assumptions for the estimates of the value of different ecosystem services.

Table 6 Present value of the ecosystem service benefits and management cost and net present value for five water management options compared to the baseline in Mio €.

Benefit and cost components		Management option				
		fen protection	redistribution	transfer	fen protection + transfer	redistribution + transfer
Ecosystem service benefits						
Greenhouse gas regulation	ΔtarB	15.50	-	-	15.50	-
	ΔL	9.88	0.98	0.49	10.33	1.19
Recreational boating	ΔtarB	-	-	-	-	-
	ΔL	0.66	0.03	1.95	2.27	1.16
Habitat conservation	ΔtarB	19.20	-	-	19.20	-
	ΔL	15.03	2.70	1.23	16.20	3.87
Agriculture	ΔtarB	-14.86	-	-	-14.86	-
	ΔL	-1.77	-0.12	0.02	-1.73	-0.11
Water management costs						
Transfer	ΔC	-	-	68.61	68.61	68.61
Restoration	ΔC	6.11	-	-	6.11	-
Net present value						
NPV* (mean and range)		68 (24 - 115)	4 (2 - 5)	-65 (-47- -83)	4 (-54 – 62)	-62 (-44 – -82)

The appraisal period is 50 years and the discount rate is 3%. ΔtarB, ΔL, ΔC are the change in targeted benefits, average annual loss and costs compared to the baseline. The area for conversion to per unit area values is 31,089 ha.

* mean, min, max of NPV are the results of a Monte-Carlo simulation across the uncertainty range of the value estimates with 1000 realisations.

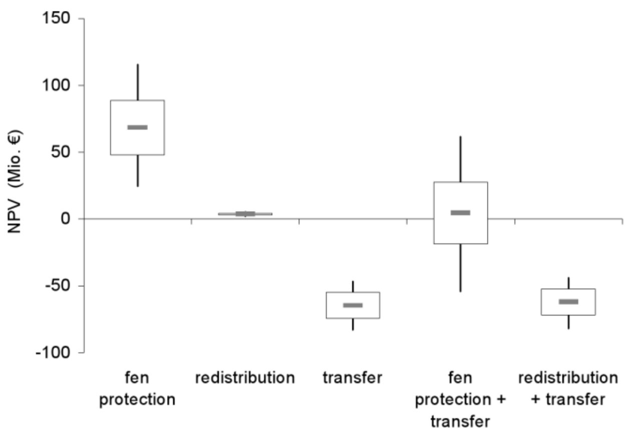

Figure 9 Net present value (NPV) of management options showing mean, standard deviation (box) and range of estimate (bar).

6 Conclusion

The assessment of the impact of water resources management decisions on ecosystem service benefits is a new challenge to integrated water resources management. This paper demonstrates a method to quantify in economic terms the multiple benefits from ecosystem services that are dependent on environmental flows and water diverted from the river. The results show that changes in water management can generate benefits from wetland ecosystem services that are of substantial economic value. These values need to be taken into consideration in the economic analysis of water management options both at the scale of wetland sub-basins and at the scale of complete river basins. Economic impact assessment methods have the advantage that a single, monetary criterion can be used to compare benefits across all types of water uses. However, the value of public goods such as the ecosystem services provided by wetlands have generally not been included in the economic assessment of water resources planning in Germany to date, partly because of lacking experience with appropriate valuation methods. The presented approach can be used for the assessments of river basin management plans and the appraisal of trade-offs in the formulation of long term water resources allocation strategies.

The results further show that the current land and water management regime of regulated and drained fen wetlands such as the Spreewald is associated with considerable negative external environmental effects, for example from greenhouse gas emissions and the loss of natural wetland habitats. Under future climatic conditions and without adaptation, wetlands in the study area will require an increasing amount of water to compensate an increasing summer water deficit and to maintain the current levels of benefits derived from the wetlands ecosystem services. The management options analysed in this paper indicate that additional water transfer could compensate some of the negative effects of increased water demand. However, this option comes at high investment and operation costs that are not offset by the increase in benefits. Instead, it is shown that water management approaches that prioritise the restoration of wetlands and the reactivation of the inter-annual water storage capacity of the fen wetland soils are economically efficient. This type of measures can substantially improve the net economic benefits compared to the current wetland water management without requiring an increase in water supply.

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Manuscript

PAPER VIII

ECONOMIC RISKS ASSOCIATED WITH LOW FLOWS IN THE ELBE RIVER BASIN (GERMANY): AN INTEGRATED ECONOMIC-HYDROLOGIC APPROACH TO ASSESS VULNERABILITY TO CLIMATE CHANGE

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This paper presents a novel methodological approach to integrate dynamic demand and economic valuation functions for a large set of different types of water uses into a stochastic simulation framework for long-term water resources planning for river basins. It is the first integrated economic-hydrologic river basin model to be presented for Germany that addresses issues of water scarcity. It presents a climate impact assessment method using the economic risk associated with variability of river water availability as the central indicator. Risk is estimated from a stochastic evaluation of short term scarcity costs.

This paper also presents the application of this method to asses the impact of regional climatic and socio-economic change on the economic risk from low flows for the main surface water using sectors within the German section of the Elbe River Basin. The analysis considers the effects on water demand, reliability of supply and economic risk for six off - stream uses (thermal power plants, industry, municipal water utilities, pond fisheries, sprinkler irrigation, water level regulated and sub irrigated wetlands) and three in - stream uses (hydropower, recreational boating and transport shipping).

The effects of climate change generally increase the water scarcity related risks for those water uses that are either affected by an increasing water demand induced by increased evapo-transpiration demands (irrigation, wetland landscapes) or that are directly susceptible to variations in in-stream flows (hydropower, shipping). Climate risks for industrial, thermal power generation and municipal uses are alleviated by reduction in demand associated with the projected population economic and technological developments.

Keywords: integrated economic-hydrologic model, water resources management, climate change, economic assessment

1 Introduction

Adapting to climate variability has always been a central concern of water management, yet has gained additional importance with prospects of climatic change. Economic assessment approaches provide a consistent and understandable framework to help compare across different water users the costs and benefits of complex mixes of water management options. Under variable and changing climatic conditions, such assessments require information on the economic costs associated with a failure of the water resources system to maintain the expected water supply. The economic assessment of complex water management interventions in large river basins that have simultaneous impact on water demand and supply at different locations along a rivers trajectory, are best addressed using a combination of hydrologic and economic models (Young 1995).

Basically three major management challenges have motivated the development of basin wide integrated hydrologic-economic water resources models: (a) to develop strategies to address drought and periodic water shortages (cf. Booker and Young 1994, Booker and Colby 1995, Ward et al. 2006), (b) to assess basin wide efficiency of water use and to assess instruments to improve efficiency for example by inter-sectoral reallocations in water scarce basins (cf. Cai et al. 2003, Rosegrant et al. 2000, Ringler and Cai 2006) and (c) to assess infrastructure investments in terms of benefits and costs in the context of long term water systems planning (Tanaka et al. 2006, Jenkins et al. 2003, Draper et al. 2002). Whereas projections of changes in demand have always been important to long-term planning, it is one of the implications of climate change for water resources management that long-term planning can no longer be based on static assumptions regarding climatic conditions and resultant water availability (Aerts and Droogers 2004, Veraart and Bakker 2009). Recent applications of integrated hydrological-economic models have been extended to assess the impacts of climate change on the long term performance of water resource system (cf Tanaka et al. 2006).

The two principal approaches to integrated modelling are simulation approaches where the behaviour of water resource users is simulated based on a set of rules governing water allocation and infrastructure operation, and optimisation approaches that optimize water allocations based on objective functions and accompanying constraints (McKinney et al. 1999). The strength of the optimisation models lies in the ability to identify economically efficient water allocations and to analyse different institutional mechanisms of water allocation. Simulation models allow a more detailed analysis of the hydrological processes. This enhances the assessment of the feasibility of management options with regard to infrastructure operations and to identify systems components that have a high risk of failure under extreme conditions.

Water scarcity has not been a major water management issue in most German river basins with the exception of the Elbe River Basin. A water resources simulation framework (named WBalMo, cf. Kaden et al. 2008) has been developed and used to address long term water resource planning in water scarce sub-basins of the Elbe River Basin. This modelling system is also used by the state water management authorities for long term water resource planning for example for the Spree River Basin which is a major sub-basin of the Elbe River Basin (cf. Koch et al. 2005 and 2006, Kaltofen et al. 2004, Dietrich et al. 2007b and Finke et al. 2004). It allows to model very large river basins larger than 100.000 km^2 and has been adapted to the conditions of the Elbe River Basin as the WBalMo Elbe Model (Kaltofen et al. 2010).

The methodological innovation reported in this paper is the development of dynamic demand functions and economic valuation functions to value the losses associated with short term water scarcity for different types of water uses and their integration into the WBalMo simulation framework. Initial efforts to include economic valuation functions were made for the Spree sub-basin of the Elbe River Basin (cf. Messner et al. 2007, Grossmann and Dietrich 2010). This paper presents both a major expansion in the types of the water uses and of the spatial scale that is covered. We present a climate impact assessment method that uses the change of scarcity costs associated with changes in availability of river water as the central indicator. The impact is analysed in terms of risk based on a stochastic evaluation of the scarcity costs that is enabled by the integration of loss functions into the water resource simulation model.

This paper also presents the application of this method to asses the impact of regional climatic and socio-economic change on the economic risk from low flows for the main surface water using sectors within the German section of the Elbe River Basin. The Elbe River Basin, which has a catchment area of ca. 148,268 km^2, is one of the European basins that regularly experiences water scarcity and recurrent low flows during summer months. The main reason for this is the low yearly precipitation total of ca. 616 mm in the period from 1951-2000. The observed climate change over this period is characterised by two trends: increasing mean annual temperatures and a decrease of total precipitation including a shift towards increased winter precipitation. A continuation of this trend is therefore mostly likely to aggravate the existing water scarcity in the central part of the Elbe River Basin (Gerstengarbe et al. 2011).

Climate change is only one of the drivers of water scarcity. Economic and technological developments as well as population and land use change have a profound impact on the hydrological cycle that may both relieve or increase the pressure on the water resource system. The largest share of the German basin is located in the eastern German federal states, so the economy of the basin is strongly influenced by post-socialist transformation processes. Reduced population growth and economic and technological catch up processes have lead to significant reductions in industrial and domestic water demand that may be expected to reduce the pressure on the water resource system well into the future. The decline of open cast lignite mining activities, on the other side, has accentuated water shortages in several regions of the basin, because mine water discharges has ceased and additional water is now required for restoration of the mining pits. At the same time recreational and environmental demands for water are becoming more important.

The remainder of this paper is organised as follows: in the next section a general description of key concepts of the assessment approach is provided. The detailed description of the modelling approach for each of the nine water using sectors addressed in this paper has only been published in parts and is therefore presented as supplementary online material to this paper. The methods section is followed by a short overview of the main characteristics of the major water uses selected for inclusion in the Elbe River Basin model. We then present the two baseline scenarios of key drivers of future water availability and demand. Finally we present and discuss the impact of these projections on the economic risk associated with river low flows.

2 Methods

2.1 Stochastic simulation of long-term water availability and management

In the WBalMo model framework, the simulation of the natural discharge and climate parameters follows a stochastic approach. These input parameters are provided by a stochastic simulation of climate data and the resultant runoff conditions. Water use processes are considered to be deterministic and dependant on changes with time and meteorological conditions. The model describes the flow of the river system as a node link network. Key model elements are the balance profiles located along the watercourses, the catchments, water users or demand sites, reservoirs and wetlands. The model operates on a monthly time step and balances water demand and water supply according to the physical capacities of the river and water management infrastructure and the water management rules in place. Water is allocated not only on a first come first serve basis, but according to the rank or priority accorded to a water use within the system of water use rights. Within the WBalMo Elbe setup, the simulation of the water balance is carried out with 100 stochastically generated realisations of the climatic and discharge conditions. This procedure enables the estimation of water supply reliabilities by means of statistical analysis after completion of the simulation.

2.2 Key concepts: risk, loss functions, coping range and adaptive capacity

We utilise the stochastic simulation framework to calculate the risk or expectation value of economic losses from low flows. The concept of risk in the assessment of environmental hazards such as flooding or drought is based on a distinction of two components that taken together determine the risk to a particular system (NCR 2000). The first component describes potentially damaging physical events (or hazards) that are characterised by their location, intensity and frequency. The second component describes the vulnerability of the elements at risk. Vulnerability is a function of exposition to the physical hazard (location) and the sensitivity or susceptibility. Sensitivity denotes the relationship between the intensity of hazard and the degree of damage caused. The sensitivity or susceptibility relationship is also referred to as a damage or loss function. A loss function therefore is a function that maps physical events onto the economic cost associated with the event.

In order to derive loss functions, we use an approach proposed by Jenkins et al. (2003). At first maximum water demand is defined as the amount current users would require if water were priced at its current level and had unrestricted availability. In any period in which water availability is less than the maximum demanded by users, economic losses represent the economic value or benefits that users would gain from additional water if availability were increased to the maximum quantity demanded. Losses therefore reflect the value of the forgone water use or the scarcity costs of water and conceptually are equal to the willingness to pay to have water supply increased to the demanded level.

As the loss function is expressed as a function of randomly distributed water availability as modelled in the stochastic simulation of the water resources system, we can establish a cumulative distribution function and an expectation value. The expectation value of loss or the average annual loss is also known as risk and is defined as:

$$E(L) = \int_{-\infty}^{\infty} \lambda(w) f(w) \delta w \qquad (1)$$

where $\lambda(w)$ is the loss function, w is a continuous random variable describing water availability, f(w) is the probability density function. We utilise the stochastic simulation framework to estimate the expected (or average annual) loss E(L) in year t under varying conditions of water availability from a discrete number of realisations as follows:

$$E(L_{t,ds}) = \sum_{r} \left[P_r \cdot L(W_{r,t,ds}) \right] \qquad (2)$$

where P is the occurrence probability of a realisation r and where :

$$\sum_{r=1}^{n} P_r = 1$$

to ensure normalisation. W is the appropriate measure of water availability and L(W) is the loss function. This basic approach is implemented for every water user or demand site. The average annual loss can be aggregated, depending on analytical interest, to economic sectors or river basin districts.

What we define as vulnerability and risk, not only reflects the exposure and sensitivity of the system to water scarcity conditions but also the ability or capacity to cope with and to adapt to the conditions of water scarce periods (cf. Aerts and Droogers 2004). Adaptive capacity describes ways of reducing vulnerability and thus risk (Smit and Wandel 2006). Adaptive capacity can be further differentiated based on the concept of coping ranges (Smit and Pilifosova 2003). Most water uses can cope with normal climatic conditions and moderate deviations from the norm, but exposures involving extreme events may lie outside the coping range. We use the term coping range to denote the shorter term capacity to deal with droughts and employ the term adaptive capacity to denote longer term adjustments in the water using processes or the water resource management system. This distinction is relevant for the type of analysis that can be carried out with the WBalMo model, because many short term coping mechanisms, even when associated with additional costs, are included in the definition of the loss functions and formulation of the water management rules within the model. The coping range can increase or decrease with time. External socio-economic factors may lead to a narrower or wider coping range, for example because the water use intensity of water demanding production processes decreases. Furthermore, the cumulative effects of increased frequency of events near the limit of the coping range may decrease the coping range or lead to the abandonment of a certain water use altogether. Such feedback effects are not considered in the model setup. Adaptive adjustments on the other hand are not modelled endogenously, but are analysed in a comparative static fashion and can be implemented as different management or adaptation scenarios. These types of measures include changes to the water allocation rules and minimum flow requirements, investments in the water supply infrastructure or changes in the water demanding use processes, such as water saving technologies.

2.3 Basic approach to demand - and loss functions

The demand and economic assessment sub-models that were developed and integrated into the WBalMo model for each type of water user consist of two interrelated subroutines that calculate water demand and return flows and subsequently the economic loss associated with water deficits. In the first step the water demand for each simulation month in a year is calculated. The general approach is to use long-term trend projections generated with models or scenarios exogenous to the WBalMo model that are then dynamically adjusted within the model to the relevant climatic parameters for the simulation month. In many cases this requires a simplified model of the production process that considers surface water and other climate variables such as temperature as inputs. The water demand for irrigation for example is a function of the long term trend in the irrigated area and the short term variation of crop water requirements that are a function of the climatic water balance in a simulation year. Similarly, water demand for thermal power plants in the long term is conditional on the installed capacity and cooling system. In the short run, cooling water demand of a cooling system is a function of air humidity and water temperature. This first component of the model therefore determines a dynamic water demand based on long term trend projections that are modified by short term climatic variability within the simulations. It also estimates the return flows that remain after any consumptive use. Consumed water is no longer available because it has been evaporated, incorporated into products or otherwise removed from the hydrological system of the river. Return flows are therefore the amount of water that is returned to the river system and is available for further use downstream.

For every month, the water demand for every water user or demand site is balanced with the available water, taking management rules, water allocations and minimum flow requirements within the river system into account. In this process, the actual allocation or supply of water to a water user in a month is determined. The resulting water deficit (difference between demand and supply) is then translated into an economic loss. This also requires simplified models of the production or use process, which take water as an input. In a first stage, the production effects are quantified in terms of physical indicators such as the available aquaculture pond area, the generated electricity or the crop yield. In the final step, the economic loss associated with these production effects is valued in monetary terms. Principally a large range of methods are available to quantify the economic loss from water deficits (cf. Young 2005). The choice of appropriate methods depends on the characteristics of the benefit or economic good generated by the water use. For example these can have characteristics of public or private goods. Another option is to differentiate between producer goods (water is used to produce other goods) or consumer goods (water is used for direct benefits to consumers). Most off-stream uses of water – such as agriculture, industry or households are private goods. Recreational uses of lakes and waterways, or the protection of wetland habitats are typical examples of public goods. Even though there is no rivalry in the use of the services, the production of these public goods often is in competition with other water uses. Although there is some overlap, the valuation methods appropriate for private goods differ from those for public goods. Table 1 provides an overview of the characteristics and valuation methods used for the water uses included in this study.

A further important distinction for our analysis is between short- and long run values (cf. Young 2005). This distinction relates to the degree of fixity of inputs, particularly where water is a producer good, as in crop irrigation, industry or power generation. In the short run, the capacity and necessary inputs are fixed, so that the sunk costs of the fixed resources can be ignored when estimating the short term losses. For purposes of water resources planning, a short run formulation is appropriate for modelling temporary variations in water supply. However, for the analysis of adaptation options that may require long-lived capital investments, all costs must be considered.

2.4 Cost-benefit framework for analysis of adaptation options

The method we present is primarily designed to be used in the assessment of water management options in a cost-benefit analytical framework. In this framework, we use a short run approach for the formulation of the loss functions to describe scarcity costs and a long run approach for the comparison of adaptation measures. Following the standard with and without procedure which in sets the net discounted costs and benefits of each management or adaptation option against the baseline management option, this view of the costs and benefits can conceptually be summarised as follows:

$$\Delta NB_{t,ds} = \Delta tNB_{t,ds} - \Delta E(L_{t,ds}) \tag{3}$$

with

$$\Delta tNB_{t,ds} = (tNB^{m}_{t,ds} - tNB^{b}_{t,ds}) \tag{4}$$

$$\begin{aligned}\Delta E(L_{t,ds}) &= E(L^{m}_{t,ds}) - E(L^{b}_{t,ds}) \\ &= (tNB^{m}_{t,ds} - (E(aNB^{m}_{t,ds})) - (tNB^{b}_{,dst} - E(aNB^{b}_{t,ds}))\end{aligned} \tag{5}$$

where NB are the net benefits of a management option for a water user at demand site ds, tNB is the targeted net benefit at full satisfaction of water demand. E(aNB) is the expectation value of the net benefit under actual conditions of water availability for the measure m and baseline b. E(L) is then the expectation value of the average annual loss compared to the water demand if water where priced at the level of tNB and had unrestricted availability. The loss in this case is equivalent to scarcity or opportunity costs of water at a demand site and a reduction of these opportunity costs would constitute an economic benefit from changes in water management. In this paper, we estimates E(L) directly and dynamically using the stochastic simulation with the integrated economic-hydrological model. In contrast ΔtNB needs to be estimated using the appropriate methods on the basis of static comparisons of scenarios.

The economic feasibility criterion for evaluating basin water management and adaptation options then can be written as:

$$PVNB = \sum_{ds}\sum_{t}\left[\frac{\Delta NB_{t,ds}}{(1+r)^{t}}\right] - \sum_{wi}\sum_{t}\left[\frac{\Delta C_{t,wi}}{(1+r)^{t}}\right] \tag{6}$$

where PVNB is the present value of net benefits from the river basin management scheme, t is a year during the schemes life, n is the expected life of the scheme, r is the discount rate, ΔNB are the incremental net benefits for water users, and ΔC is the change in capital and operating costs for water management infrastructure wi. Implementing this approach requires the application of appropriate methods to estimate the marginal or incremental benefits and benefits forgone from changes in water management. This paper does not present an analysis of adaptation options but presents the necessary methods to estimate E(L) and an analysis of the trajectory of E(L) across different baseline scenarios.

2.5 Detailed description of the demand - and loss functions

A detailed description of the demand and loss functions can be found in the online supplement to this paper. It describes the specific approaches to estimate water demand and loss for six off - stream uses (thermal power plants, industry, municipal water utilities, pond fisheries, sprinkler irrigation, water level regulated and sub-irrigated wetlands) and three in - stream uses (hydropower, recreational boating and transport shipping). These are also summarised in Table 1 below. In addition an attempt is made to clarify the key input parameters that can be varied in the context of scenario analysis and the main coping and adaptation options that have been implemented in the model to date. More detailed descriptions of some of the sectoral approaches to loss functions have already been published (cf. Koch and Vögele 2009 for thermal power plants; Mutafoglu 2010 for industry, Messner et al. 2007 for pond fisheries, Möhring and Grossmann 2011 for transport shipping, Grossmann and Dietrich 2010 and 2011 for lowland wetlands; Koch and Grossmann 2011 in prep. for hydropower), but the methodology as a whole is presented here for the first time.

3 Water users included in the Elbe model

The WBalMo Elbe model was developed on the basis of different data sources (cf. Kaltofen et al. 2010). Water users to be included in the WBalMo Elbe model were determined on the basis of two criteria: (a) they are major water users with regard to their total water demand or (b) their water use was assumed to be associated with a high economic value. The general intention was to cover all major in-stream and off-stream water uses. The water uses that were included are summarised in Table 1. We limit the economic assessment of low flows presented in this study to the German part of the basin. Data describing the locality and estimates of current water use or current allocated water rights were provided by the different state and local water management authorities and include those water users identified as significant under the reporting requirements of the EU Water Framework Directive (FGG 2004). All other user specific parameters that are required for the modelling approach, such as irrigated area, pond area or installed power plant capacity were subsequently compiled by the authors from various sources and surveys specifically for this study. In the following section the water using sectors and demand sites included in the model setup are characterised in more detail.

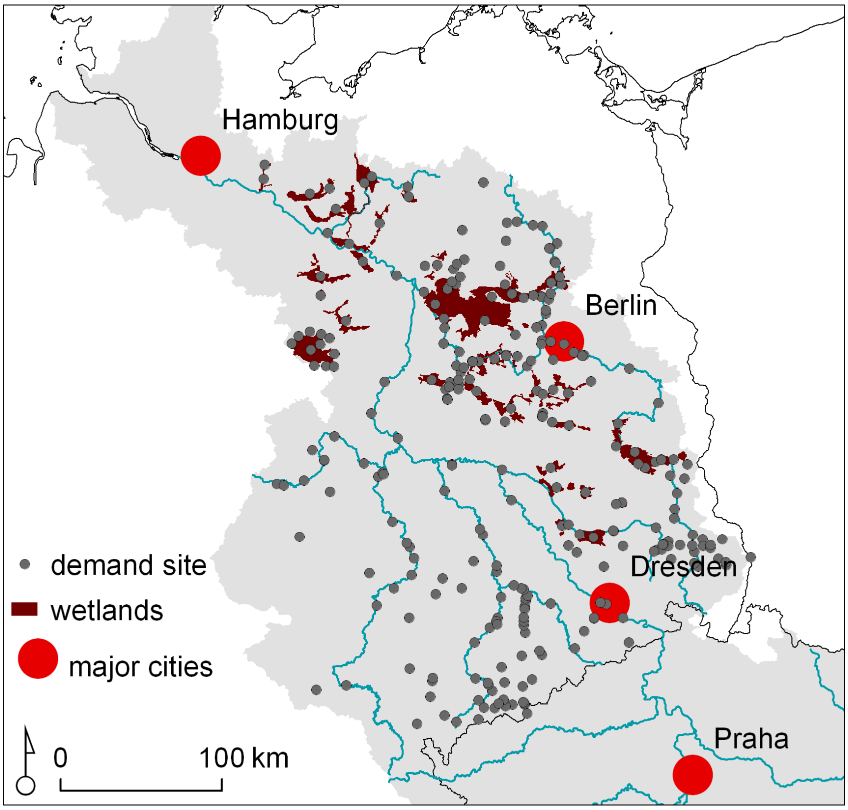


Figure 1: Overview map of the German part of the Elbe Basin showing water demand sites on the main tributaries of the Elbe River and the location of major wetland sites.

Thermal power plants

There are several different types of thermal power plants operated in the Elbe River Basin, with capacities ranging from a few kW to several GW (Vögele and Markewitz 2011). Different fuels are used that include nuclear, gas and lignite coal. Whilst some power plants are operated to produce base load electricity throughout the year, others are mainly operated to generate heating energy during the winter. Of the 25 power plants in the German section with a capacity larger than 50 MW, 17 are included as water users with a loss function in the model. This includes all major power plants upstream of the nuclear power plants on the lower Elbe (upstream of Krümmel). Of the power plants included in the model, 9 have a once-through cooling, whilst the rest have a cooling tower system.

Industry

Freshwater intake for industrial uses is either supplied by direct withdrawal from surface or groundwater resources or from piped sources from municipal water utilities (Mutafoglu 2010). All major industrial abstractors of surface water are modelled within the WBalMo framework, including both individual firms as well as operators of industrial parks who supply industrial water users located at their site. A total of 86 industrial users are taken into consideration with respect to water demand and return flow into surface waters in the German part of the Elbe River Basin. As the focus here is placed on the supply side of water resources, only those 47 firms which actually are surface water abstractors are included in the model. About half of these belong either to the chemical industry or are firms from the pulp and paper industry while the remaining are distributed over various manufacturing sectors as e.g. food and beverages, metals or non-energy mining and quarrying.

Municipal water supply utilities

Most water abstracted by municipal water supply utilities in the German part of the Elbe River Basin is taken from groundwater (Ansmann 2010). Especially in the mountainous areas, surface water is taken from reservoirs specifically designated for drinking water supply purposes and transported to consumption sites via a system of large water pipelines. In the lowland regions bank filtration and groundwater sources predominate. In total there are 57 demand sites included in the model that use surface water directly or from bank filtration. Of these 9 are solely dependent on surface water taken from reservoirs and 14 use both reservoir- and bank filtration water sources. Water abstracted by water supply utility returns to the river via the sewage system and is discharged at waste water treatment plants that may or may not be in the vicinity of the abstraction point. All 281 major waste water treatment plants are included however with constant discharges in the model..

Table 1: Overview of key characteristics of water users and the deployed methods for the parameterisation of loss functions included in the WBalMo model

Water user type	Demand sites in model	Public / private good	Main short term effect of reduced water availability	Method for parameterisation of loss function	Detailed description of method
Off-stream, consumptive uses					
Thermal power plants	17	private	Reduced electricity generation	Change in net rents	Koch and Vögele 2009
Municipal water supply utilities	57	private	Substitution	Change in net rents	-
Industry	47	private	Reduction of production capacity	Change in net rents	Mutafoglu 2010
Pond fisheries	62	private	Reduced pond area and yield	Change in net rents	Messner et al. 2007
Irrigation for agriculture	91	private	Reduced crop yield	Change in net rents	-
Sub irrigation of wetland sites:	35				
agricultural production		private	Reduced crop yield	Change in net rents	Grossmann and Dietrich 2010
habitat conservation		public	Reduction of area with high water levels	stated preferences / benefit transfer	Grossmann 2010
greenhouse gas emissions		public	Increase in GHG emission from peat oxidation	Shadow price of carbon	Grossmann and Dietrich 2011
In-stream, non consumptive uses					
Hydropower	53	private	Reduced electricity generation	Change in net rents	Koch and Grossmann 2011 in prep
Recreational boating of minor waterways	36	public	Reduced locking frequency with increased waiting time	stated preferences / choice model	Meyerhoff and Grossmann 2007
Transportation by ship on inland waterways (transport relations)	12	private	Reduced effective carrying capacity of ships	Change in transport costs	Möhring and Grossmann 2011

Pond fisheries

The German section of the Elbe River Basin has approximately 62 pond fisheries, which are mainly concentrated in the region of Lusatia in Saxony and Brandenburg, where pond fisheries date back to the 12th century (SLL 2005). A few pond fisheries can also be found along the Havel River and in Thuringia. The commercially dominant species within the pond fishery sector is the carp, which is raised over a 3 to 4 year long time period. During the past years pond fisheries have increasingly been affected by water scarcity. Trout farms are preferably located in the vicinity of freshwater springs in the mountains and are not included in this study. All 62 carp pond fisheries are included in the model.

Sprinkler irrigation of agricultural crops

In the east German states Thuringia, Saxony, Saxony-Palatinate and Brandenburg, which cover a major share of the Elbe River Basin, it is estimated that some 75,000 – 90,000 ha or 1.4 – 1.9 % of the agricultural crop are irrigated (Roth et al. 1995). Roughly three quarters of the irrigation water is taken from surface water (reservoirs and rivers) and the remaining quarter from ground water. While irrigation of fodder crops was wide spread during the socialist era, sprinkler irrigation is now applied mainly to high value crops such as potatoes and field vegetables. In the western German state of Lower Saxony, only parts of which are in the Elbe River Basin, some 8 %, regionally up to 80 %, of the crops are irrigated. This corresponds to a total irrigated area of 235,000 ha. Three quarter of the water in this area is taken from groundwater sources. The model covers 53,000 ha of irrigated crop at 91 demand sites, with an average size of 486 ha per demand site. The data is based on the irrigation water rights and partially of the corresponding irrigated area provided by state water authorities for the model setup. From the data it is estimated that on average an abstraction right of 0.0013 (SD +/- 0.0007) hm^3/ha is granted.

Sub-irrigated and water table regulated lowland (fen) wetlands

The lowland region of the Elbe River Basin is characterised by large wetland areas, in many sub basins with a share of around 20 % of the total area (cf. Dietrich et al. 2010 submitted). Almost all of these wetland areas have been drained in the last centuries and as a result their water table is today regulated by weirs. Because of the negative climatic water balance additional water from the river systems is required to maintain water levels during summer month. The system of regulation and drainage has been constructed in a manner that allows water transfers and sub-irrigation of the wetlands. Regulated wetlands thus constitute one of the major water users within the lowland river systems. In total 35 wetlands that are larger than 1,000 ha were included as water users in the WBalMo Elbe model. They have a total area of 3,840 km^2. Around 50 % of these wetland sites are groundwater influenced sandy soils, while fen soils constitute around

35 % of the area. 54 % of the wetland areas are grassland, 34 % are arable and the rest is forest and open water. Irrigation water supplied to maintain water levels delivers multifunctional benefits that are contingent on the ecological functions of the wetland ecosystem, such as agricultural biomass production, greenhouse gas sequestration, landscape amenity and wildlife habitat provision (cf. Turner et al. 2008).

Hydropower

There are approximately 484 hydropower plants in the German part of the Elbe River Basin (cf. Koch and Grossmann 2011 in prep). More than 400 of these have a nominal capacity which is less than 500 kW. 56 plants have a capacity ranging between 500 kW and 5 MW. 9 hydropower plants have a nominal capacity above 5 MW, three of which have a capacity above 150 MW. Plants with a capacity above 5 MW are with one exemption, pumped hydropower plants, which use surplus energy from the power grid to pump water from a lower reservoir to a higher reservoir for a subsequent release at peak electricity demand. These plants were not included in model because water is used in a more or less closed cycle. Plants with a capacity below 500 kW were also not included, because these are located on small tributaries for which no site specific flows are modelled. In total 53 hydropower plants are included in the model.

Transport shipping on inland waterways

The Elbe River and its most important tributaries have the status of inland waterways. These are the largely free flowing stretches of the Elbe River in Germany and the water level regulated stretches of the Elbe River in the Czech Republic, the Moldau River, Saale River and Havel River. Further connections are enabled by canals such as the Mittelland-Canal, Elbe-Havel-Canal, Elbe-Side-Canal and Elbe-Lübeck-Canal. The channel depth on regulated river stretches and canals is more or less constant and independent of discharges but varies with discharge for the main River. Channel depth is one of the key determinants of the degree of capacity utilisation of vessels and the lowest channel depth of a transport relation determines the maximum load capacity. We include all transport relations that use parts of the Elbe River in the model (cf. Möhring and Grossmann 2011). These are aggregated to 12 major transport relations, for example Hamburg Port – Magdeburg Port, or ARA Ports (Amsterdam Rotterdam Antwerpen) to Czech Republic, that vary greatly with regard to the demand for transportation services. Nearly 50 % of the total goods are transported on the Hamburg – Magdeburg relation, which can be operated independently of the Elbe River via the Elbe-Side-Canal.

Recreational boating on inland waterways

The Lakes Region in the north-eastern German States of Mecklenburg Vorpommerania and Brandenburg is characterised by a multitude of lakes and natural rivers which have been interconnected by artificial canals to create a network of waterways. The central

Müritz Lake is connected onward to the Elbe River by the Müritz Elde Waterway to the west and to the Havel River by the Müritz-Havel-Waterway. The Müritz Lake itself is an important reservoir, whose discharge to the waterways is regulated by weirs and influenced also by the frequency of lock use. The frequency of locking also determines the minimum flows required for maintaining navigability along the waterways. If water does not suffice, the frequency of locking has to be reduced, for example by shifting from on-demand to fixed locking schedules. The locks on the Müritz-Elde-Waterway registered ca. 4,000 boat movements and the Müritz-Havel-Waterway ca. 25,000 – 30,000 boat movements per year in 2000. The long term trend of boat movements through the locks show that activity levels on the waterways have more than doubled in the last ten years (cf. Meyerhoff and Grossmann 2007). An outcome of this increased recreational traffic is that at peak times in the summer holiday season (July – August), the capacity of the locks is exceeded and substantial waiting times of several hours have been registered. In total there are 18 locks on the Müritz-Elde Waterway and 27 locks on the Havel River and its tributaries upstream of Berlin. 36 of these locks are included in the WBalMo Elbe model.

4 Scenarios of regional climatic and socio-economic change

The economic risk from low flows is a function of water availability and water demand at a demand site and the susceptibility of the water user to water deficits. We evaluate the risk for two baseline scenarios that have been developed for the Elbe River Basin (Hartje et al. 2011, Gerstengarbe et al. 2011). These scenarios include both the effects of climate and land use changes on water availability and the changes in the water demand of major users that are determined by socio-economic change. This paper compares two of these baseline scenarios: MBasis A1° STAR T2 and MBasis B2+ STAR T2.

The STAR T2 climate projection corresponds to an average increase of temperature of 2°C for the Elbe estuary and 2.8°C for the middle and upper Elbe River until 2050. For the baseline scenarios it is assumed that water management corresponds to the current management practice. However, changes already planned by the water authorities are included. For both these scenarios additional climatic variance has been analysed, by selecting also a set of climate projections that correspond to 25 % dryer (STAR T2t) and 25 % wetter (STAR T2f) realisation from the central climate scenario (cf. Gerstengarbe et al 2011).

Table 2: Baseline scenario assumptions for key determinants of water demand

Water user	Summary of assumption	Source
Thermal power plants	The development of the power plant capacity and technology at different sites is the output of the power plant sector model. Higher water demand in A1o due to continuation of lignite mining and reduced water demand in B2+, due to fading out of lignite mining.	Vögele and Markewitz 2010
Industry	Demand changes according to sectoral rates of demand change that is a function of economic growth and increasing water use efficiency. Roughly 10 % higher demand in A1o and 30 % lower demand in B2+. Projection held constant after 2028/32	Mutafoglu 2010
Municipal water utilities	Demand changes according to rates of demand change for every utility that is dependant on changes the population growth, economic activity and water use efficiency. Generally reduction of demand by ca. 15-20 %, slightly higher reduction in B2+. Projection held constant after 2028/32	Ansmann 2010
Pond fisheries	Basically no change in pond area. Change in water demand based on evapotranspiration under the climatic conditions of the scenario.	n.a.
Irrigation	No additional irrigation perimeters and no change of crops, but water demand is based on optimal irrigation under the climatic conditions of the scenario.	n.a.
Wetlands	No change in water level regulation targets or land use assumed, but water demand is based on evapotranspiration water demand under climatic conditions of the scenario.	n.a.
Hydropower	Installed capacity is assumed to be unchanged.	n.a.
Transport shipping	Based on projection of transport and shift in fleet structure. In the A1o there is a much more pronounced growth of transported goods then in B2+ scenario due to higher economic growth rates and completion of the Saale waterway, which is not considered in B2+.	Möhring and Grossmann 2011
Recreational boating	Visitation rates increase according to projection. In the *A1* scenario further growth of tourism activity in the region is assumed as a consequence of enhanced economic growth, no such increases are assumed in the *B2* scenario. Projection held constant after 2028/32	PLANCO 2006.

Table 3: Baseline scenario assumptions for key price variables

Water use	Variable	Unit	*A1°*		*B2+*		Source for parameters
			2000	2050	2000	2050	
Thermal power plants, Hydropower	Market price of base load electricity.	€ kWh^{-1}	0.02	0.07	0.02	0.09	EWI/ PROGNOS 2005
	Renewable energy price for base load electricity	€ kWh^{-1}	0.065	0.07	0.065	0.09	EWI/ PROGNOS 2005
	Saved variable costs *	€ kWh^{-1}					
Industry	Gross revenue *	€ $employee^{-1}$					STABU 2005
	Share of variable costs of gross revenue*	[-]					STABU 2005
Water utilities	Additional variable costs for substituting surface water	€m^{-3}	0.035	0.035	0.035	0.035	this study.
Pond fishery	Produce price fish	€ kg^{-1}	2	2	4	4	SLL 2005
	Saved variable costs	€ ha^{-1}	2048	2048	2048	2048	SLL 2005
Irrigation	Produce price irrigated crop (potatoes)	€ dt^{-1}	8	9	8	12	Neubert et al. 2006
	Saved variable costs harvest	€ dt^{-1}	1.5	1.5	1.5	1.5	Neubert et al. 2006
	Saved variable costs irrigation	€ m^{-3}	0.5	0.5	0.5	0.5	Neubert et al. 2006
Wetlands	Produce price for substitute: biomass energy maize	€ dt FM^{-1}	1.8	1.8	2.4	2.4	Neubert et al. 2006
	Saved variable costs harvest	€ dt DM^{-1}	2	2	2	2	Neubert et al. 2006
	Shadow price of carbon	€ $tCO2^{-1}$	20	20	70	70	UBA 2007
	Willingness to pay for habitat conservation	€ ha^{-1} a^{-1}	55	55	350	350	Grossmann 2010
Shipping	Variable vessel operating costs*	€ h^{-1}					BMVBW 2005
Recreational boating	Willingness to pay to avoid waiting time	€ $boat^{-1}$ h^{-1}	1.2	1.2	1.2	1.2	Meyerhoff and Grossmann 2007

* too many sub categories and size classes to be reported here, refer to Mutafoglu 2010, Vögele and Markewitz 2009 and Möhring and Grossmann 2010

The A1 economic baseline assumes higher economic growth rates than the B2 baseline. A further key difference is the orientation of environmental policy, especially energy policy. In the B2+ baseline, more ambitious reduction targets for CO2 are set, that result in relative higher prices for CO2 emission certificates. As a result open cast lignite mining is faded out. In the A1° baseline, less ambitious reduction targets are assumed and as a result lignite coal mining is continued. The sources and characteristics of the key assumptions regarding determinants of demand used in this study are summarised in Table 2.

In addition, each baseline scenario requires assumptions regarding the trajectory of the key price variables that together with the functional form of the susceptibility or loss function determine the magnitude of economic loss associated with a water deficit. The key price variable assumptions used in this study are summarised in Table 3.

As a general summary to facilitate interpretation of the results, the water demand is generally higher in the A1° baseline compared to the B2+ baseline, while the price assumptions are higher in the B2+ baseline.

5 Results

5.1 General introduction to interpretation of results

In the following sections we present results that characterize the impact of regional climate and socio-economic change baseline scenarios on water demand, water deficit and risk (loss) and how are these distributed in space and across water using sectors. We begin with a sectoral perspective in order to provide some insight into the specific factors determining the results for each sector before presenting aggregate results for the basin.

We discuss results for each of the nine water using sectors along four indictors based on disaggregated data for each demand site. We present the mean water demand (Figure 3) and a measure of reliability of supply (Figure 4). These two indicators are evaluated for the water scarcest month of July. The focus on the water scarcest month provides insights into the distribution of water scarcity and risk because water scarcity is a phenomenon that occurs during the summer month. The central indicator of economic risk is the expectation value of loss or average annual loss (Figure 2). An average annual loss of zero implies that a water demand sites does not experience any water scarcities even in dry years. This corresponds to a reliability of supply of 100 %. However the July water deficits presented in Figure 4 do not translate directly into economic risk, because the risk indicator is based on annual losses that take the water availability over the course of a year and the available coping mechanisms into account. This difference becomes readily visible by comparing Figures 2 and 4. Finally we present maps to indicate the spatial distribution of demand and the expected change of risk across the simulation period (Figure 5). Aggregate data on water demand and risk by sector are presented in Table 4.

Generally we compare results for the simulation period 2008-2012 and 2048-2052. We present results for the two baseline scenarios to investigate the influence of assumptions regarding development of water demand and regarding key price variables. The range between the two baselines can therefore be interpreted as a sensitivity analysis regarding the impact of the demand and price assumptions.

In interpreting the economic risk indicator it has to be kept in mind that the average annual loss describes the opportunity costs of water scarcity and that this is conceptually not equivalent to the loss of rentability of existing enterprises because the rent expectations will already accommodate an (unknown) part of the current risk of water scarcity. However the change in risk over time or between management options is conceptually equivalent to changes in income.

In interpreting the results it has to be further kept in mind, that the scarcities modelled for specific demand sites also reflect the water allocation rules and minimum flow requirements that constitute the water resources management baseline. The results therefore indicate localities where these kind of restraints need to be investigated in more detail in order to identify measures to climate proof the long term water resource allocation plans. Such investigations need to consider potential changes in water allocation or technical adaptations to reduce the identified vulnerability. The increase in scarcity costs presented in this paper therefore reflect the costs of climate change without any adaptation beyond the coping mechanisms implemented in the modelling approach.

5.2 Water demand and risk from low flows by sector

Thermal power plants

The aggregate water demand of power plants in the initial period is ca. 26 m^3/s. The consumptive use is only ca. 20 % of the actual withdrawal. The summer water demand of the sector decreases significantly by 50 – 90 % from the period 2008-2012 to 2048-2052. This is a result of a reduction of the power plant stock and assumed changes to the cooling systems in the course of replacement investments. Roughly 75 % of the sites do not experience any losses at all and are therefore not at risk from low flows. The risks of the remaining sites are higher in the A1o baseline than in the B2+ baseline because the water demand is higher.

Industry

Industrial water demand during the month of July in the initial period totals ca. 15 m^3/s, ca. 70 % of which of which is consumed. Total industrial surface water demand decreases by 15 – 20 % over the period of analysis. Roughly 60 % of the sites do not experience any losses at all. The increases of risk are similar for both the A1o and B2+ baseline, despites differences in demand between the baselines.

Municipal water supply utilities

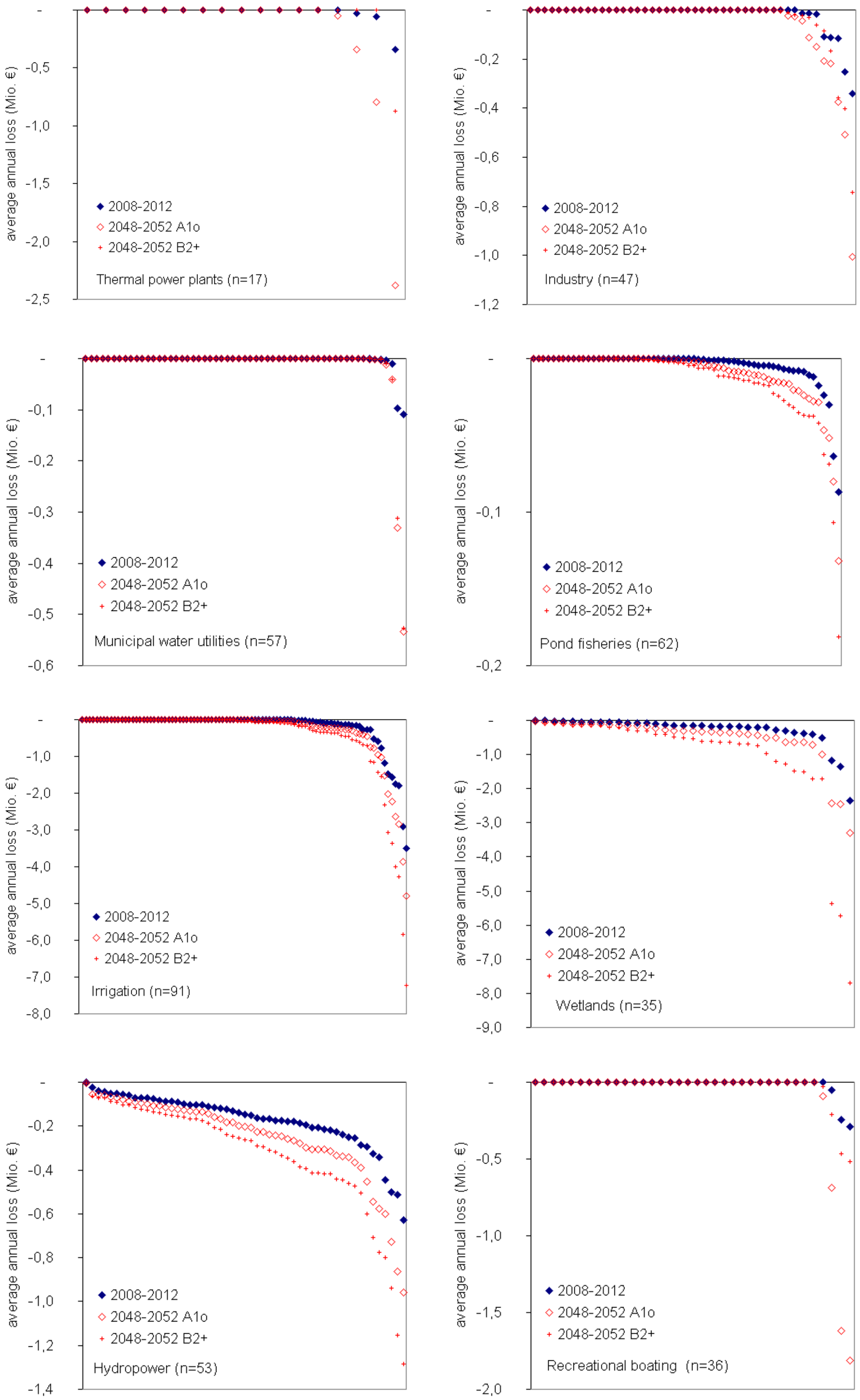
Municipal water supply utilities in total abstract ca. 10 m^3/s in July in the initial period. Approximately 4 m^3/s are abstracted from the drinking water reservoirs, the remaining water is abstracted mainly by the way of bank filtration. Return flows occur via the sewage system. The total water demand for water abstraction from the reservoirs decreases by ca. 10 – 15 % over the simulation period. Less than 10 % of demand sites are at risk from experiencing substantial losses. There are no marked differences in risk between the scenarios, despite differences in demand.

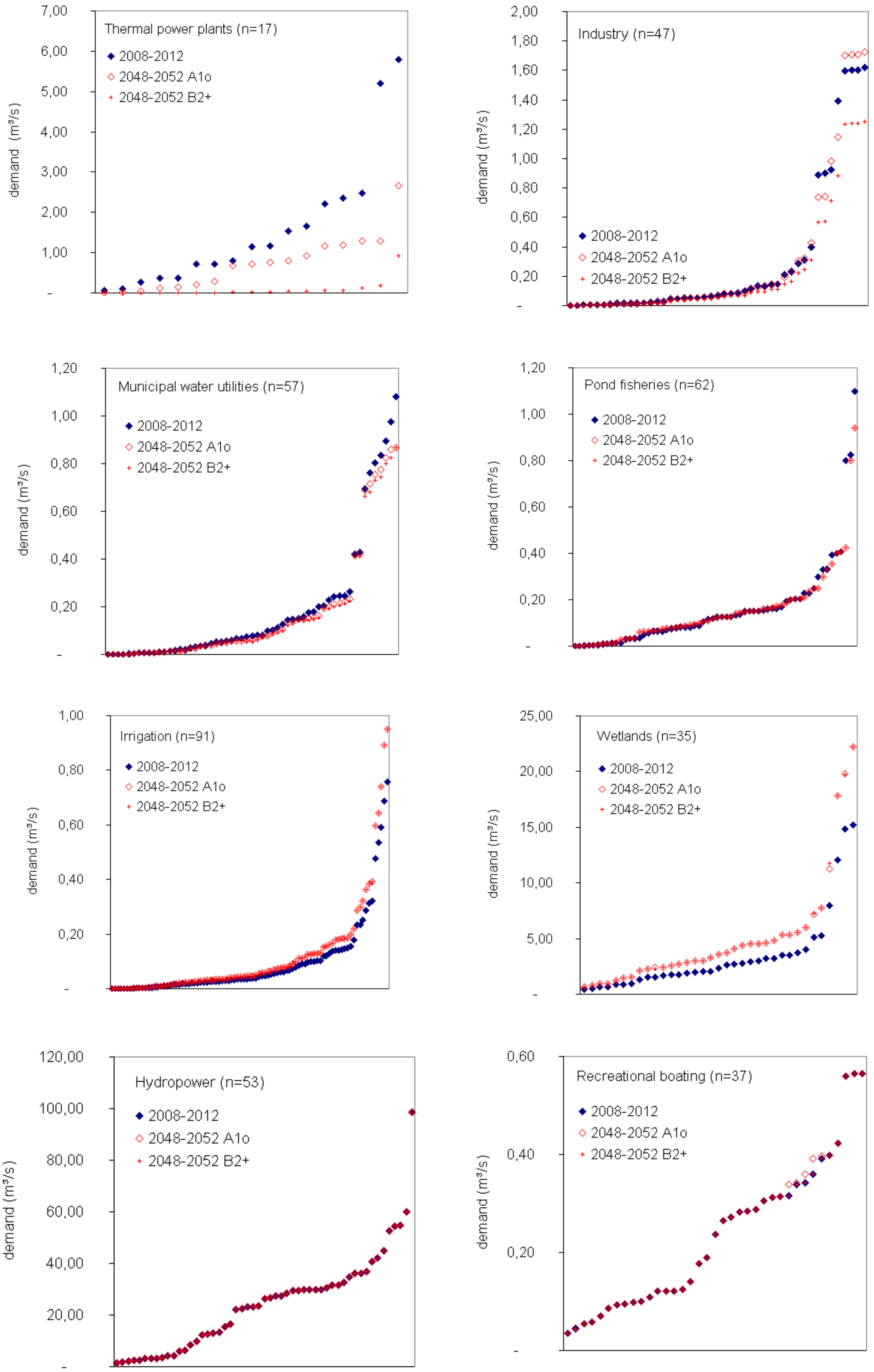
Figure 2 (next page): Cumulative distribution of average annual loss of every demand site for the period 2008-2012 and the period 2048-2052 for the <u>*MBasis*</u> ***A1° STAR T2*** and <u>*MBasis*</u> ***B2+ STAR T2*** baseline scenarios by water using sector.

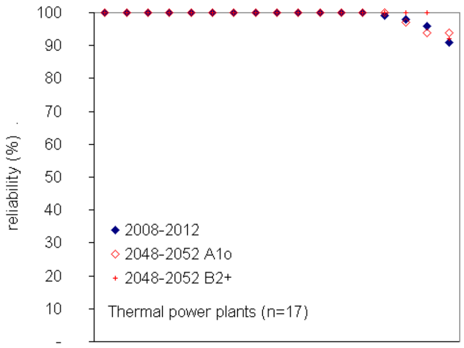
Figure 3 (second next page): Cumulative distribution of median July water demand (50th percentile) for every demand site for the period 2008-2012 and the period 2048-2052 for the <u>*MBasis*</u> ***A1° STAR T2*** and <u>*MBasis*</u> ***B2+ STAR T2*** baseline scenarios by water using sector.

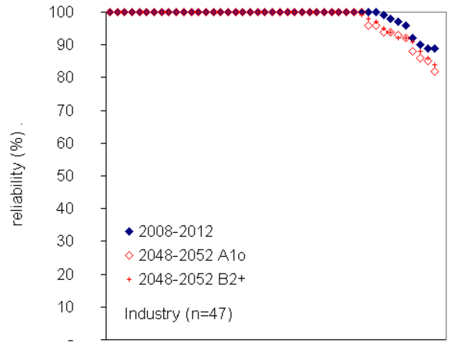
Figure 4 (third next page): Cumulative distribution of mean monthly reliability of supply for the month of July for every demand site for the period 2008-2012 and the period 2048-2052 for the <u>*MBasis*</u> ***A1° STAR T2*** and <u>*MBasis*</u> ***B2+ STAR T2*** baseline scenarios by water using sector.

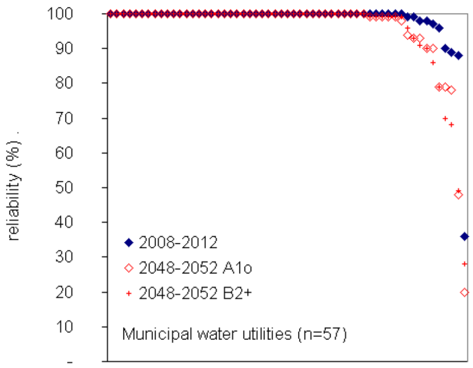
Figure 5 (fourth next page): Maps of the German section of the Elbe River Basin showing the distribution of median July water demand (50th percentile) for the period 2048 - 2052 and the average annual loss in the the period 2008-2012 (black symbol) and 2048-2052 (white symbol) for the baseline scenario <u>*MBasis*</u> ***A1° STAR T2*** by water using sector.

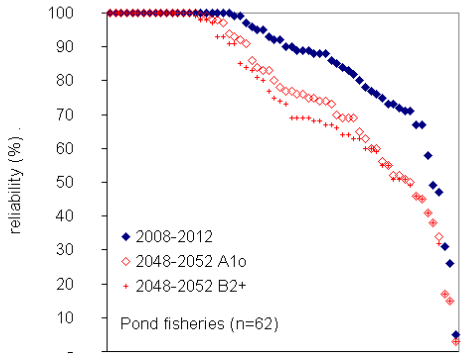

average annual loss (Mio. €)
2008-2012
2048-2052 A1o
2048-2052 B2+
Thermal power plants (n=17)
Industry (n=47)
Municipal water utilities (n=57)
Pond fisheries (n=62)
Irrigation (n=91)
Wetlands (n=35)
Hydropower (n=53)
Recreational boating (n=36)

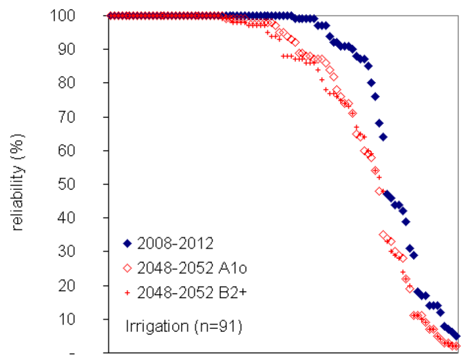

Thermal power plants (n=17)
2008-2012
2048-2052 A1o
2048-2052 B2+
demand (m³/s)
7,00
6,00
5,00
4,00
3,00
2,00
1,00
-
Industry (n=47)
2008-2012
2048-2052 A1o
2048-2052 B2+
demand (m³/s)
2,00
1,80
1,60
1,40
1,20
1,00
0,80
0,60
0,40
0,20
-
Municipal water utilities (n=57)
2008-2012
2048-2052 A1o
2048-2052 B2+
demand (m³/s)
1,20
1,00
0,80
0,60
0,40
0,20
-
Pond fisheries (n=62)
2008-2012
2048-2052 A1o
2048-2052 B2+
demand (m³/s)
1,20
1,00
0,80
0,60
0,40
0,20
-
Irrigation (n=91)
2008-2012
2048-2052 A1o
2048-2052 B2+
demand (m³/s)
1,00
0,80
0,60
0,40
0,20
-
Wetlands (n=35)
2008-2012
2048-2052 A1o
2048-2052 B2+
demand (m³/s)
25,00
20,00
15,00
10,00
5,00
-
Hydropower (n=53)
2008-2012
2048-2052 A1o
2048-2052 B2+
demand (m³/s)
120,00
100,00
80,00
60,00
40,00
20,00
-
Recreational boating (n=37)
2008-2012
2048-2052 A1o
2048-2052 B2+
demand (m³/s)
0,60
0,40
0,20
-

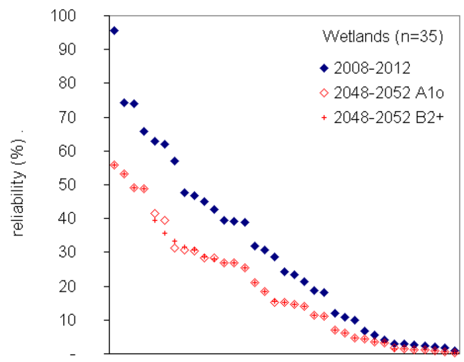

reliability (%)
2008-2012
2048-2052 A1o
2048-2052 B2+
Thermal power plants (n=17)

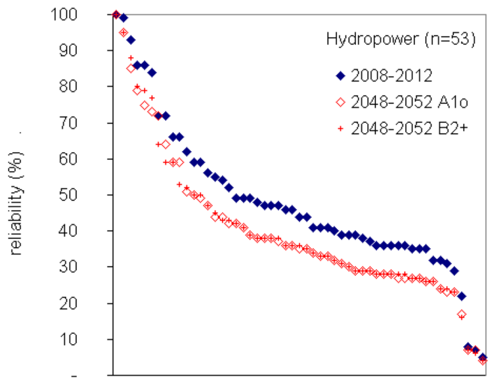

reliability (%)
2008-2012
2048-2052 A1o
2048-2052 B2+
Industry (n=47)

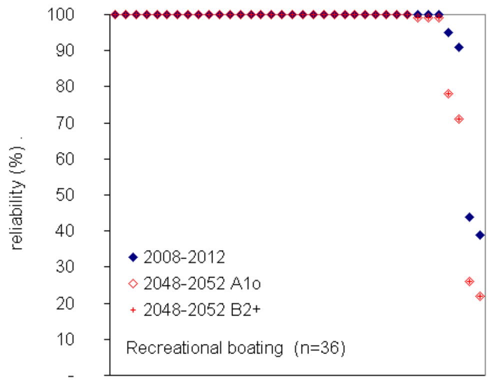

reliability (%)
2008-2012
2048-2052 A1o
2048-2052 B2+
Municipal water utilities (n=57)

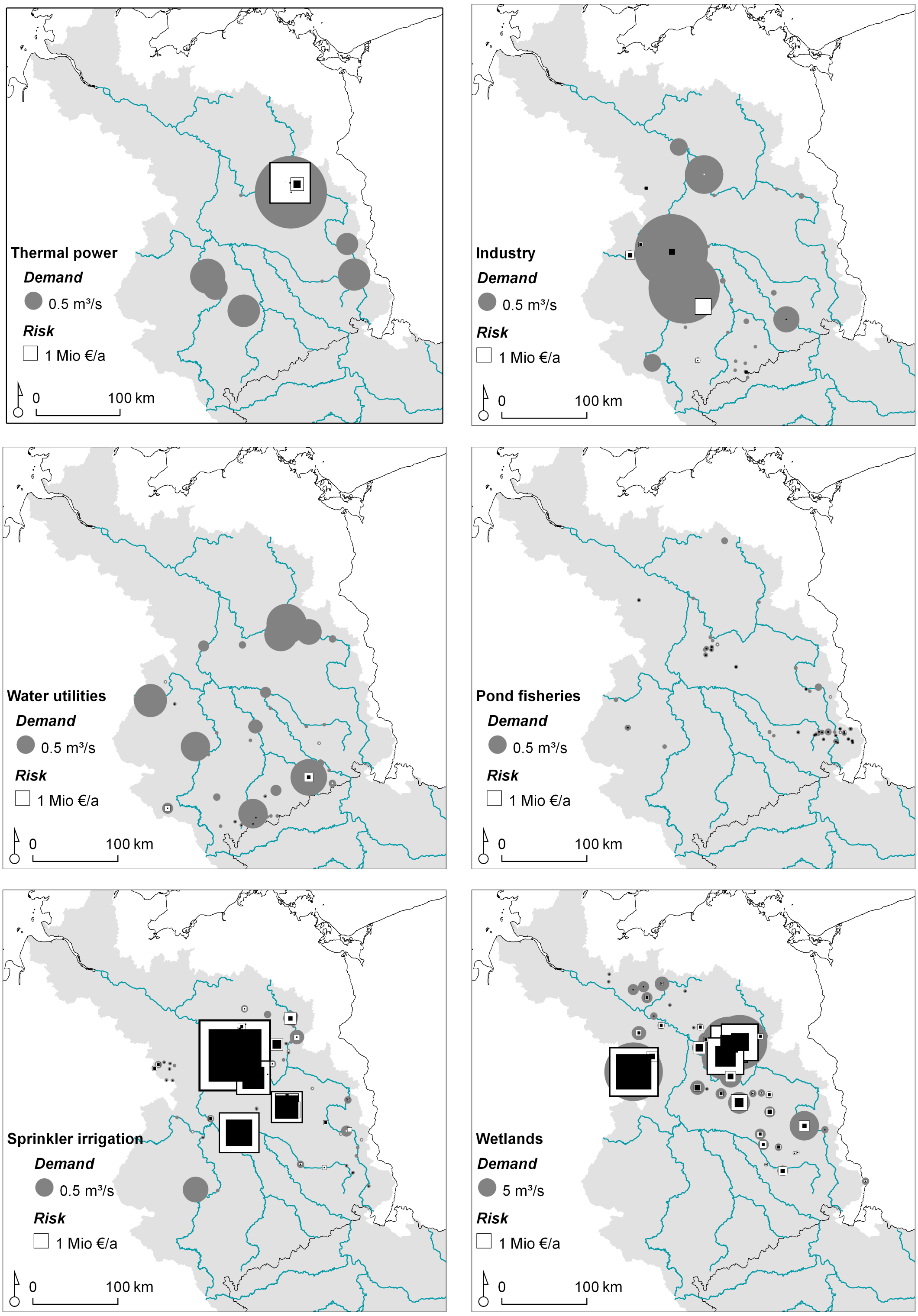

reliability (%)
2008-2012
2048-2052 A1o
2048-2052 B2+
Pond fisheries (n=62)

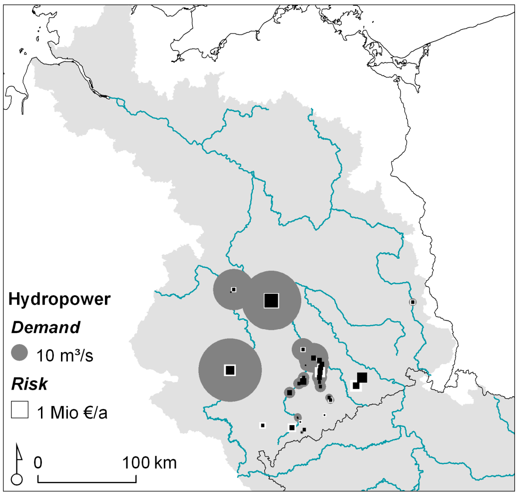

reliability (%)
2008-2012
2048-2052 A1o
2048-2052 B2+
Irrigation (n=91)

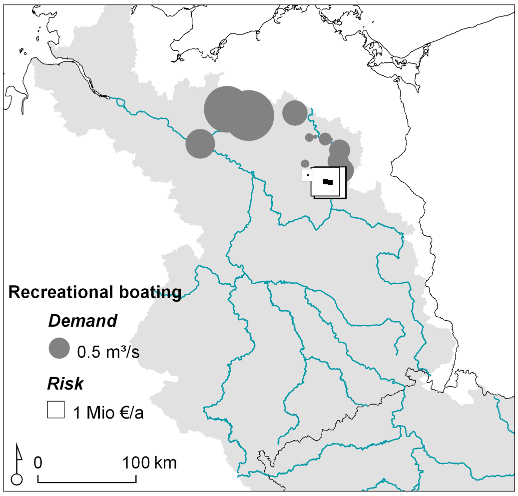

reliability (%)
Wetlands (n=35)
2008-2012
2048-2052 A1o
2048-2052 B2+


reliability (%)
Hydropower (n=53)
2008-2012
2048-2052 A1o
2048-2052 B2+


reliability (%)
2008-2012
2048-2052 A1o
2048-2052 B2+
Recreational boating (n=36)


Thermal power
Demand
0.5 m³/s
Risk
1 Mio €/a
0
100 km
Industry
Demand
0.5 m³/s
Risk
1 Mio €/a
0
100 km
Water utilities
Demand
0.5 m³/s
Risk
1 Mio €/a
0
100 km
Pond fisheries
Demand
0.5 m³/s
Risk
1 Mio €/a
0
100 km
Sprinkler irrigation
Demand
0.5 m³/s
Risk
1 Mio €/a
0
100 km
Wetlands
Demand
5 m³/s
Risk
1 Mio €/a
0
100 km





Pond fisheries

The water demand of pond fisheries in the initial period for July is ca. 10 m³/s. Approximately 45 % of the withdrawal is evaporated. Total surface water demand decreases slightly, because some ponds are projected to go out of production over the period of analysis, and the number of sites that are at risk from water scarcity increase from ca. 50 to 65 %. There is marked difference between the A1+ and B2+ scenario caused by differences in water availability in the basins with discharges from lignite mines and the differing price assumptions.

Irrigation

Agricultural sprinkler irrigation has initial July water demands of ca. 8 m³/s of which there are no return flows to the surface waters. Water demand increases by ca. 30 % over the simulation period. This is a direct effect of decreasing precipitation and increased water demand for evapo-transpiration of the crops, as the irrigated area is assumed to be constant. More than 50 % of the sites show water scarcity costs already in the base period, indicating that the potential demand for optimal irrigation is higher than the possible irrigation intensity. The number of demand sites at risk does not increase over the simulation period, but the risk for those at risk increases. The difference in risk between the baselines is largely a result of different price assumptions.

Wetlands

The demand for water for sub-irrigation of wetlands also rises with increased evapo-transpiration, in this case by ca 45 % over the simulation period. The demand for additional irrigation water to maintain water level regulation targets of the 32 major lowland wetland sites is estimated to be on average ca. 120 m³/s for the month of July in the initial period. Wetlands thus constitute the largest off-stream demand for water in the Elbe River Basin with a share of ca. 60 % of off-stream demands. All sites are at risk from water scarcity; however water scarcities are most pronounced in regions with higher climatic water balance deficit in summer and with small catchments providing additional inflows (cf. Dietrich et al. 2010 for a detailed analysis). The difference in risk between the baselines is largely a result of different price assumptions.

Hydropower

In-stream water users have high nominal demands for flows, but the water is returned to the river immediately. The hydropower plants included in the model have a potential demand of ca. 1,290 m^3/s. This demand remains constant because no new sites were included in the model. All sites show scarcity costs from low flows that increase over the simulation period. Hydropower plants are not designed to operate always at maximum capacity, so the loss in the initial period represents the difference to the maximum installed capacity. However the increase in risk over the simulation period represents the expected additional losses induced by reduced flows. The median reliability of supply decreases from approximately 48 % to 40 % over the simulation period. The difference in risk between the baselines is largely a result of different price assumptions.

Recreational boating

The total demand for minimum stream flows required for the operation of locks on the minor waterways is almost constant with slight increase at some tourism hotspots. The number of demand sites at risk increase from ca. 10 % to ca. 15 % over the simulation period. Differences in risk between the baseline scenarios are a result of differences in the utilisation frequencies of locks by recreational boaters induced by increasing demand.

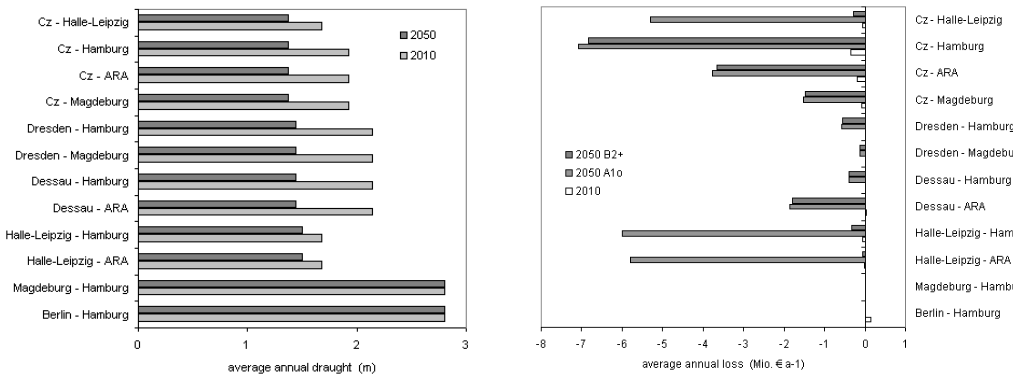


Figure 6: Average annual draught (left) and average annual loss (right) for twelve inland waterway transport relations for the period 2008-2012 and the period 2048-2052 for the <u>*MBasis*</u> ***A1° STAR T2*** and <u>*MBasis*</u> ***B2+ STAR T2*** baseline scenarios.

Transport shipping

Figure 6 shows average annual draught for the twelve transport relations. The figure also shows the loss from low flows over the simulation period for the twelve transport relations that potentially use the Elbe River. Loss is defined as the difference of aggregate transport costs compared to the waterway conditions as targeted in the Federal Transport Infrastructure Plan (cf. BMBV 2005). This target (GLW_{1995}) is defined as a minimum draught of 1.6 m on 345 days, and of 2.5 m more than 50 % of the year. Nearly all transport relations are affected by reduced flows in the Elbe River. However the losses are determined by possible alternative routes using water level regulated canals. For example the relation with the highest demand (Magdeburg – Hamburg) can be operated independent of the Elbe River and the resultant risk is very low. Risk is therefore highest for the relations to the Czech Republic that are limited by the lowest river channel depth (cf. Möhring and Grossmann 2011 for a more detailed analysis). The difference in risk between the baselines is mainly a result of different assumptions regarding the increase in demand for transportation.

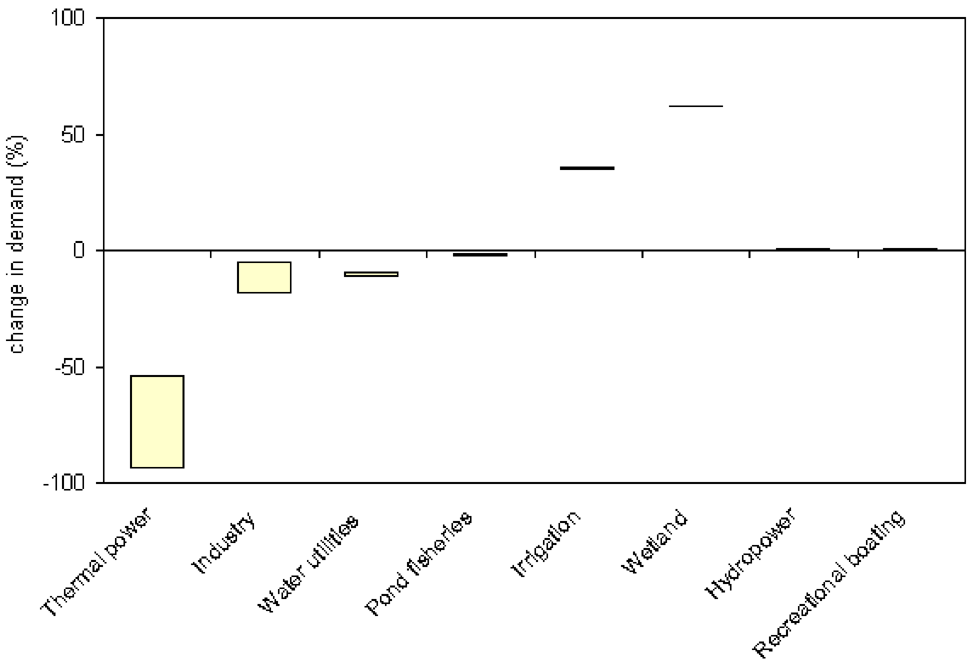


Figure 7: Change in aggregate median monthly demand for July from the simulation period 2008/12-2048/52 by water using sector. Range denotes difference between the A1o and B2+ baseline scenarios.

5.3 Aggregate regional impact of climatic- and socioeconomic change

Impact on water demand

Table 4 presents a comparative overview of the change in summer water demand from 2008-2012 to 2048-2052 aggregated for the water using sectors that summarise the trends identified above. It is useful to differentiate between those sectors for that we have considered both climatic and socio-economic drivers of demand (thermal power plants, industry, municipal water supply, recreational boating) and those that only take climatic modifiers of demand into account (irrigation, wetlands, pond fisheries) or that have constant demand (hydropower). The results in Figure 7 show that substantial demand reductions can be expected for thermal power plants (ca. 50 – 90 %), municipal water supply (ca. 10 %) and industry (ca. 0 – 15 %) that are largely driven by technological development and socio-economic developments. On the other hand, major increase in water demand can be expected for irrigation (ca. 25 %) and wetland (ca. 45 %) water demand that increase with increased potential evapo-transpiration of crops and wetland landscapes.

Impact on low flow risks

We also compare the magnitude in change of low flow associated risks over the simulation period (2008-2012 to 2048-2052) aggregated for sectors (Table 4 and Figure 8). This perspective gives an indication about the aggregate dimension of increase in low flow risks associated with regional climate and socio-economic change. The result of the aggregation is of course also contingent on the completeness of the sample of users included in the model which, whilst different for sectors, was designed to include all large or economically important water users in the basin.

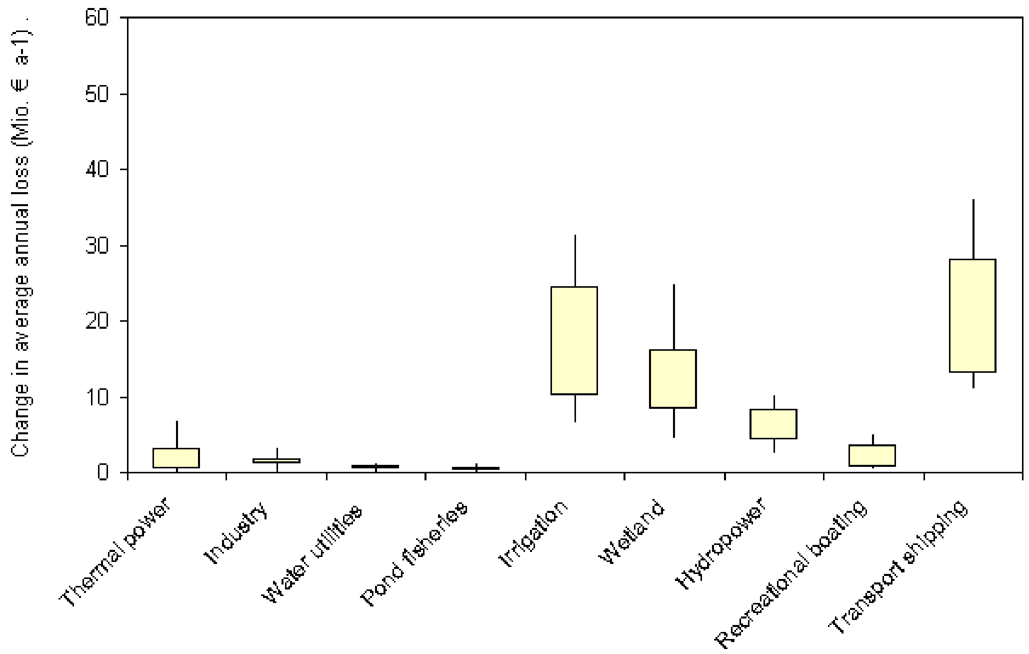


Figure 8: Change in aggregate average annual loss from the simulation period 2008/12-2048/52 by water using sector. The boxes denote the range between A1o and B2+ baseline scenarios for the central climate projection START T2 and the bars additional range for the drier and wetter climate realisations.

Table 4: Aggregate median July water demand (m³/s) and average annual loss (Mio. € a-1) by sector for the period 2008-2012 and 2048-2052 for the baseline scenarios A1o compared to base year.

	Median July water demand (m³/s)								Average annual loss (Mio. €. a-¹)							
	B2+				A1o				B2+				A1o			
	STAR T2 central		STAR T2 central		STAR T2 wet		STAR T2 central		STAR T2 dry		STAR T2 wet		STAR T2 central c		STAR T2 dry	
	2008/12	2048/52	2008/12	2048/52	2008/12	2048/52	2008/12	2048/52	2008/12	2048/52	2008/12	2048/52	2008/12	2048/52	2008/12	2048/52
Thermal power	26,1	1,5	26,9	12,3	-0,2	-0,1	-0,4	-0,9	-0,5	-1,7	-0,2	-0,3	-0,4	-3,6	-0,5	-7,2
Industry	14,9	12,4	16,7	16,8	-0,7	-0,7	-0,8	-1,9	-0,9	-3,0	-0,8	-1,0	-1,0	-2,7	-1,1	-4,1
Water supply utilities	10,7	9,5	10,9	9,8	-0,2	-0,3	-0,2	-0,9	-0,3	-1,4	-0,2	-0,3	-0,2	-0,9	-0,3	-1,5
Pond fisheries	10,0	9,5	10,0	9,5	-0,3	-0,4	-0,3	-1,0	-0,4	-1,5	-0,3	-0,3	-0,3	-0,6	-0,4	-1,0
Irrigation	8,6	10,8	8,6	10,8	-18,0	-35,6	-19,9	-44,4	-20,2	-51,4	-16,9	-23,5	-18,7	-28,7	-19,0	-33,0
Wetland	120,4	176,7	120,4	177,0	-17,7	-26,5	-21,6	-37,7	-22,4	-47,0	-8,0	-12,4	-9,8	-18,1	-10,1	-22,8
Hydropower	1292,6	1292,6	1292,6	1292,6	-9,0	-15,1	-9,1	-17,5	-9,3	-19,4	-9,0	-11,4	-9,2	-13,3	-9,3	-14,8
Recreational boating	8,1	8,1	8,1	8,2	-0,4	-0,8	-0,5	-1,2	-0,6	-1,6	-0,4	-2,6	-0,6	-4,2	-0,6	-5,6
Transport shipping	0				-1,9	-12,9	-2,2	-15,3	-3,1	-17,7	-2,0	-28,0	-2,2	-30,4	-3,1	-39,1

Table 5: Percentage of water users by sector and sub-basin that are at risk from low flows for the A1o baseline in the period 2008-2012 (P1) and 2048-2051 (P9).

	Total			**Upper Elbe***			**Havel ****			**Mulde**			**Saale**			**Spree-Schwarze Elster**			**Lower Elbe**		
	N	**P1**	**P9**	**N**	**P1**	**P9**	**N**	**P1**	**P9**	**N**	**P1**	**P9**	**N**	**P1**	**P9**	**N**	**P1**	**P9**	**N**	**P1**	**P9**
Thermal power	17	24	29				1	0	0	1	0	0	4	0	0	11	36	45			
Industry	47	17	21	5	40	40	4	0	25	12	17	17	20	20	25	4	0	0	2	0	0
Water utilities	57	14	18	9	11	22	6	0	0	16	25	19	12	25	33	14	0	7			
Pond fisheries	62	53	65				18	33	44				5	20	40	36	67	75	3	67	100
Irrigation	91	56	69	11	82	82	41	51	61				10	10	30	18	56	83	11	91	100
Wetlands	35	95	96	1	100	100	12	100	100	0			0			9	93	96	13	91	91
Hydropower	53	100	100	5	100	100				31	100	100	16	100	100	1	100	100			
Recreational boating	36	8	14				21	14	24										15	0	0

N = total number of water users, * Upper Elbe from Schmilka to mouth of Havel; ** Havel without Spree.

Spatial distribution across sectors and basins

This emerging general picture corresponds to the findings regarding the share of demand sites of a sector that are at risk from low flows presented above. These are summarised by sub-basin and sector in Table 5. The share is larger than 75 % of the demand sites for hydropower and wetlands and larger than 50 % of demand sites for sprinkler irrigation and pond fisheries. Less than ca. 25 % of the demand sites for thermal power plants, industry and water supply are at risk from low flows.

The regional pattern reflects both the differences in water availability and the distribution of the water users within the basin. A large share of the sub-irrigated wetlands and agricultural sprinkler irrigation sites are concentrated in the Havel basin. Its main rivers are part of the waterway system that is used for recreational purposes. The Mulde River and the Saale River hold a major share of the hydropower capacity and industrial water uses, while thermal power generation is concentrated in the Spree River sub-basin.

6 Conclusions

This paper has presented a climate impact assessment method using the economic risk associated with changes in surface water availability as an indicator. Risk is assessed based on loss functions that are integrated into a stochastic water resource management simulation framework. The method can provide information for water resources planning that in this form is currently unavailable in German river basins. Previously this kind of economic assessment was conducted using a loose mix of separate approaches or was considered infeasible at the presented scale and detail. The outlined method incorporates a wider range of benefits from water use in the economic assessment, such as recreational and ecosystem service benefits, than has been customary in water resource planning to date.

One of the key challenges in building an assessment model that covers such a large area at a high level of spatial disaggregation as presented here, is to compile sufficiently reliable information on the water using processes regarding key determinants of actual demand (such as capacities, irrigated areas) and their water rights (such as allocations, ranking priority, limitations). Such information is not readily available in a trans-boundary and federally organised water management setting as is found in the Elbe River Basin. The systematic assembly and reconciliation of data for such a large river system is a major product of the model development process in its own right and serves as the first step to systematically analyse the system of water resource management at the chosen scale.

The results presented in this paper provide a first overview of the vulnerability of the surface water using processes in the German part of the Elbe River Basin to climate change. The study has demonstrated that analysing vulnerability in terms of economic risk has the advantage that a single, monetary criterion can be used to compare losses and benefits effects across a wide range of water uses. It has been shown that a focus on the water use benefits (output of water use process) facilitates the incorporation of various coping mechanisms in the assessments of water shortages compared to an analysis based only on the security of water supply (input perspective). However models are unlikely to be able to represent the full range of coping options and so might be pessimistic in many regards. On the other hand, because we use a time-step of one months in the simulation, extreme values for temperature or low flows potentially affecting the water users might be missed. Whilst the approaches for thermal power plants or wetlands are already of a high complexity, there is room for improvement. Especially the representation of irrigation and municipal water utilities could be improved, for example regarding a more detailed consideration of substitution mechanisms with groundwater and possibilities for conjunctive management. Important water uses that are missing completely from the economic assessment include benefits from environmental flows related to water quality, the multifunctional benefits of reservoir water use for example for recreation and pumped hydropower plants.

The results of the model based assessment of vulnerability to climate change, despite the limitations, point to some practical conclusions for the long term water planning process in the Elbe River Basin. One of the implications of climate change for water resources management is that long-term planning can no longer be based on static assumptions regarding climatic conditions and resultant water availability. The study provides an ex ante assessment on the magnitude of expected impact of important factors that shape future water scarcities and their spatial distribution. As expected, the effects of climate change generally increase the water scarcity related risks for those water uses that are either affected by an increasing water demand induced by increased evapo-transpiration demands (irrigation, wetland landscapes) or that are directly susceptible to variations in in-stream flows (hydropower, shipping). Climate risks for industrial, thermal power generation and municipal uses are mitigated by reduction in demand associated with the projected population, economic and technological developments. This study also highlights that substantial economic value is associated with public good benefits from recreation or wetland ecosystem services that should not be left out of the assessment of water management options.

The results show that in general less than 50 % of the demand sites are at any risk from water scarcity over the simulation period and that higher risks are concentrated on less than 20 % of the demand sites. Exceptions are hydropower plants and wetlands, of which nearly all site are affected by changes in water availability. A further implication of the results is that the effects of reduced water availability will tend to exacerbate existing water scarcities. Anticipated additional shortages are expected for example from the closure of open cast mining operations. This is an interesting result, as it reinforces the arguments for addressing already manifest water management problems. Without adaptation, this could potentially raise the intensity of existing water management conflicts.

The risk based climate impact assessment presented here must be considered as a first and scoping step to identify potential needs for adaptation to climate change in the basin. In a second step, adaptation options need to be analysed in detail. The methodology presented here has primarily been designed for the comparative assessment of water management options in a cost benefit analytical framework, as was outlined in the methods section of this paper. Whilst adaptation options may reduce the climate change impact, it is to be expected that there will be much less leeway in the system compared to current operations. The challenge to adequately capture the whole range of possible adaptation options such as improvements in infrastructure operation, demand management or changes in water allocation mechanisms remains for the further application of this modelling approach.

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SUPPLEMENTARY MATERIAL

To manuscript entitled: Economic risks associated with low flows in the Elbe River Basin (Germany): an integrated economic-hydrologic approach to assess vulnerability to climate change.

Description of water demand - and loss functions

Overview

In the following section the approaches developed to model water demand and economic losses within the framework of the integrated economic hydrologic model are described in detail for each sector. Key characteristics of the selected water users are described before the demand- and loss functions are presented. In addition an attempt is made to clarify the key input parameters that can be varied in the context of scenario analysis and the main coping and adaptation options that have been implemented in the model to date. DQ, SQ and RQ generally denote demand, actual supply and actual return flows. General subscripts used are ds, m and t: ds denotes a specific demand site, t denotes a specific year, and m a month of the year. These subscripts indicate the spatial and temporal resolution for the variables and parameters required by the model.

Thermal power plants

Water demand and water consumption

The key factors that determine water demand are fuel, the installed plant capacity and the available cooling technology. Additional factors influencing cooling water use are the actual temperature of the river water that determines the maximum additional heat load before permissible water temperatures are reached. Air temperature and humidity are important variable that determine the amount of water evaporated in the cooling tower. The approach is described in detail in Koch and Vögele (2009).

The monthly water demand of a power plant with once-through cooling is estimated according to the following approach:

$$DQ_{(ds,m,t)} = \frac{\Phi_{(ds,m,t)}}{\vartheta \cdot c \cdot AS_{(ds,m,t)}} \tag{1}$$

where DQ is the cooling water demand in [m³], Φ is the heat flow that has to be conducted in [MJ], ϑ is the density of water in [t/m³], c is the specific heat capacity of water in [MJ/t*K] and AS is the available rise in temperature of the cooling water in [K].

The available rise in temperature of the cooling water is calculated as the difference of the estimated actual water temperature T in [K] and the maximal allowable temperature Tmax in [K] at a demand site:

$$AS_{(ds,m,t)} = \max(T\max_{(ds)} - T_{(ds,m,t)};0) \quad (2)$$

The demand for cooling water is thus the quotient of the heat that has to be conveyed and the capacity of water to absorb that heat. The heat that has to be conveyed is dependant on the energy efficiency of the power plant and the amount of heat that is either lost and released to the environment via other pathways such as steam production or is used for heating purposes. The heat that has to be conveyed is estimated as follows:

$$\Phi_{(ds,m,t)} = BR_{(ds,m,t)} \cdot (1-\eta_{ges}) \cdot (1-\alpha) \quad (3)$$

where BR is the fuel use in [MJ], η_{ges} is the efficiency of fuel use of the plant, α is a correction factor to account for lost heat that does not have to be conveyed via cooling water.

The fuel use is estimated as follows:

$$BR_{(ds,m,t)} = nKW_{(ds,t)} \cdot h_{(ds,m,t)} \cdot 3{,}6 \cdot \frac{1}{\eta_{elek}} \quad (4)$$

where nKW is the nominal capacity of the power plant in [kW], h is the capacity utilisation of the power plant in [full load hours] and η_{elek} is the electricity energy efficiency.

The demand for water of power plants with a cooling tower is the sum of the water required for evaporation in the cooling tower and the water required to keep the concentration of solubles in the water of the cooling system below the required level. In contrast to through-flow cooling water is thus permanently taken from the river and lost through evaporation. The heat that is conveyed from a cooling tower is contingent on the air temperature and humidity. Taking the conduction of heat through the cooling tower into account, the equation for once-through cooling is extended and becomes:

$$DQ_{(ds,m,t)} = \frac{\Phi_{(ds,m,t)} \cdot (1-\beta) \cdot \omega_{(ds,m,t)}}{\vartheta \cdot c \cdot AS_{(ds,m,t)}} \cdot EZ \quad (5)$$

where β is the average share of heat that is conducted via the cooling tower, ω is a correction factor that takes air temperature and humidity into account and EZ is the densification factor.

The value for β is around 0.993. The correction factor ω is calculated on a monthly basis in order to take effects of temperature and humidity into account. The factor ranges between 0.75 (cold dry winter month) und 1.25 (hot humid summer month). The densification factor is a measure for the concentration of solubles in the cooling water system. It is calculated as:

$$EZ = \frac{SQ}{RQ} \tag{6}$$

and is to be maintained in a range between 1 und 3. The return flow can then be calculated as:

$$RQ_{(ds,m,t)} = \frac{SQ_{(ds,m,t)}}{EZ} \tag{7}$$

Loss function

If water availability is limited or if the temperature of the discharged cooling water may not be raised further, we assume that the electricity production of the power plant has to be reduced. The actual monthly production capacity of the power plant can be estimated by rearranging the water demand equation and to solve for the capacity of the plant using available water supply and available temperature rise as an input. For plants with once-through cooling this gives:

$$KW_{(ds,m,t)} = \frac{SQ_{(ds,m,t)} \cdot 4{,}2 \cdot AS_{(ds,m,t)}}{h_{(ds,m,t)} \cdot 3{,}6 \cdot \frac{1-\eta_{ges}}{\eta_{elek}} \cdot (1-\alpha)} \tag{8}$$

and for plants with a cooling tower:

$$KW_{(ds,m,t)} = \frac{SQ_{(ds,m,t)} \cdot 4{,}2 \cdot AS_{(ds,m,t)}}{h_{(ds,m,t)} \cdot 3{,}6 \cdot \frac{1-\eta_{ges}}{\eta_{elek}} \cdot (1-\alpha) \cdot (1-\beta) \cdot \varpi_{(ds,m,t)} \cdot EZ} \tag{9}$$

where KW is the actual capacity of the power plant in [kW], SQ is the available water in [m3], AS is the available rise in temperature of the cooling water in [K], all others as above.

Valuation of losses in the thermal power plant sector is based on a change in net rents approach. Under the assumption that the total annual electricity demand to be supplied by the power utility is constant, it is assumed that additional electricity is bought on the electricity market to compensate a reduction of a power plants electricity production. Annual loss is then calculated as the reduction in net rents by calculating the costs of the required electricity purchases, net of savings in variable costs, as follows:

$$L_{(ds,t)} = \sum_{m} (nKW_{(ds)} - KW_{(ds,m,t)}) * h * (P_t - sC_{ds}) \quad (10)$$

where nKW is the nominal capacity of the power plant in [kW], KW is the actual power production capacity in [kW], h is the capacity utilisation of the power plant in [full load hours], P is the purchasing price for electricity in [€/kWh] and sC are the saved variable costs of production in [€/kWh].

Coping and adaptation options

The major long term adaptation options for power plants that can be modelled is a change of the cooling technology. Short term coping mechanisms implemented in the model are the utilisation of the full allowable range for raising the temperature of the cooling water. An additional coping mechanism that is implemented, but not fully explained here, is that regularly scheduled maintenance can be shifted to low flow periods to avoid losses.

Key variables for scenario and sensitivity analysis

The key demand shift variables are the generating capacity and cooling technology at a power plant site. The key price variable is the assumed price of base load electricity at demand site and the saved variable costs of production. Furthermore future prices of CO2-allowances have a significant influence on the type of fuel used and electricity prices.

Industry

Water demand and water consumption

The current annual abstraction volumes and their monthly distribution are scaled by the annual change in demand of industrial water demand. The monthly demand then equals that month's fraction of the adjusted annual demand:

$$DQ_{(ds,m,t)} = (nQ_{(ds)} * DR_{(ds,t)}) * mFrac_{(ds,m)} \quad (11)$$

where DQ is the water demand in [m^3], nQ is the nominal water demand in [m^3], *DR* is the factor by which water demand differs from the nominal demand and mFrac is the month's fraction of average annual water use.

The effective water use ratios are assumed to be constant for the specific sectors, so that return flows are determined as:

$$RQ_{(ds,m,t)} = SQ_{(ds,m,t)} * (1 - effFrac_{(ds)}) \quad (12)$$

where effFrac is the share of water that is lost in the production process by evaporation or inclusion in the product.

Loss function

The approach is described in detail in Mutafoglu (2010). In order to estimate effects of water shortage on production in different industrial sub-sectors, a survey of industrial firms was conducted in relevant water intensive sectors in the Elbe River Basin. Because only a few of the industrial firms have ever experienced water shortages, a series of contingent questions was posed in order to elucidate a possible functional relationship between reduction in water supply and production. More specifically, the firms were asked about the potential impacts of a one month water shortage of 7 %, 15 %, and 30 % respectively, compared to the usual water intake during summer months. Furthermore, the firms were also requested to quantify that level of water shortage which would force the firm to stop operations completely. For simplicity reasons a linear relationship between water availability and the level of output is assumed between any two points on the respective curve, so that

$$\begin{aligned} & rPR_{(ds,m,t)} = \\ & \text{if}(\mathrm{d}_{(\mathrm{ds,m,t})} < \mathrm{T1}_{(\mathrm{ds})}; 0; \\ & \text{if}(\mathrm{d}_{(\mathrm{ds,m,t})} > T2_{(\mathrm{ds})}; 1; (\mathrm{d}_{(\mathrm{ds,m,t})} - \mathrm{T1}_{(\mathrm{ds})}) * \frac{1}{\mathrm{T2}_{(\mathrm{ds})} - \mathrm{T1}_{(\mathrm{ds})}})) \end{aligned} \quad (13)$$

where rPR is the proportion of maximum production that can be maintained with the supplied water, d is the proportion of water demand that is supplied, T1 is the threshold above which production is not reduced and T2 is the threshold below which production ceases.

Valuation of losses in the industrial sector is based on a change in net income approach. It is based on the assumption that a reduction in production will lead to a proportional reduction in sales revenues. Practically all large industrial plants included in the model operate 24 hours per day and may not compensate reduced production during a period of water scarcity by increased production in another month. The reduction in net rents is estimated as the reduction in gross income net of the savings of variable costs related to production such as resource inputs to production. Because the industrial sectors produce a very wide range of different products, the revenue for each industrial production site was estimated using statistical data on the average revenue per employee. This information is available for sub-sectors and size classes from a regular survey of the industrial sector conducted by STABU (2007). Also available from this data source is the proportion of production related variable costs to total revenue. In addition information on the number of employees at each production site included in the WBalMo Elbe model was collected. Using this data, annual loss is defined as follows:

$$L_{(ds,t)} = \sum_{m} (gI_{(ds)} * \mathrm{E}_{(\mathrm{ds})}) * \mathrm{rPR}_{(\mathrm{ds,m,t})} - (gI_{(ds)} * \mathrm{E}_{(\mathrm{ds})}) * \mathrm{rPR}_{(\mathrm{ds,m,t})} * \mathrm{C}_{(\mathrm{ds})} \qquad (14)$$

where gI is the sector and size class specific gross revenue per employee in [€/employee and month], E is number of employees at a production site, rPR is the proportion of maximum production that that can be maintained with the supplied water and Cv is the sector and size class specific share of variable costs in relation to gross revenue.

A total of six manufacturing sectors are differentiated within the valuation framework. These comprise chemicals, paper and pulp, metals, food and beverages, textiles, and non-energy mining and quarrying. To take structural differences between different firm sizes into consideration, the firms are also differentiated with respect to employment size classes. Generally speaking, firms with a smaller number of employees show a larger share of fixed costs. Also of relevance for the modelling is the fact that gross output values per employee typically increase with increasing number of employees.

Coping and adaptation options

Currently internal coping options in the production process in the short-run, such as a temporarily more intense circulation of water during drought periods are implicitly included in the parameters of the sector specific loss functions, as derived from the responses of the survey respondents. Further adaptation or coping options such as procuring water from other sources or switching to other largely closed water circuits especially with respect to cooling are not considered to date. A short term substitution is not considered feasible without investment in necessary conveyance and storage infrastructure in most cases.

Key variables for scenario and sensitivity analysis

The key demand shift variables are the rate of change in demand for each manufacturing sector. The key price variable is the sector and size class specific gross income per employee.

Municipal water supply utilities

Water demand and water consumption

The current annual extraction rates and monthly distribution at the demand sites are scaled by the determined annual demand increase. The monthly demand is then equal to the specific month's fraction of the adjusted annual demand:

$$DQ_{(ds,m,t)} = (nQ_{(ds)} * DR_t) * mFrac_{(m,ds)} \tag{15}$$

where DQ is the water demand in [m³], nQ is the nominal water demand in [m³], *DR* is the factor by which water demand differs from the nominal demand and mFrac is the month's fraction of average annual water use.

The return flows that occur via the waste water treatment plants were kept constant for this analysis, because the linkages between treatment plants and water abstraction sources could not yet be established. The discharge from waste water treatment plants includes water abstracted from groundwater sources and includes surface runoff in areas where surface areas are drained by the sewerage system.

Loss function

Valuation of losses in the water supply utilities sector is based on a change in net income approach. In case of water scarcity we assume that bank filtration water is a substitute when surface water sources do not suffice. It is assumed that the additional costs of water provisioning incurred from additional pumping and water treatment costs reduce the net income of the affected water works and annual loss is then calculated as:

$$L_{(ds,t)} = \sum_m (DQ_{(ds,m,t)} - SQ_{(ds,m,t)}) * aC_{ds,t} \tag{16}$$

where DQ and SQ are the water demand and supply in [m³], aC are the additional provisioning costs in [€/m³] at a demand site.

Coping and adaptation options

The endogenous coping option included in the model to date is a substitution of surface water by bank filtration water, which is in the current model version assumed to be available in unlimited amount at the surface water demand sites.

Key variables for scenario and sensitivity analysis

The key demand shift variables are the rate of change in demand for each demand site. The key price variable is the additional costs for substitution of surface by bank filtration water.

Pond fisheries

Estimation of water demand and water use

Fish ponds are filled in spring and drained in autumn. Monthly water demand is estimated based on the amount of water required to fill the ponds and to compensate evaporation from the pond surface. Water demand of the pond fisheries is estimated as follows:

$$DQ_{(ds,m,t)} = (A_{(ds)} * ETw_{(ds,m,t)}) + V_{(ds)} * rf_{(m)} \tag{17}$$

where rf is the proportion of the total pond volume that is refilled in a specific month, A is the total pond area in [ha], ETw is the monthly evaporation from water surfaces [m^3/ha] and V is the total volume of the ponds in [m^3].

The demand estimate corresponds roughly to the effective water use. Return flows occur mostly in the autumn months when ponds are drained.

Loss function

The approach is described in detail in Messner et al. (2007). Fish production is estimated from the available surface area of standard fish ponds. Carp pond fisheries typically consist of a number of ponds with a target water depth of 1.1 m. Insufficient water supply causes falling water levels, rise in water temperature, and finally oxygen shortage in the pond waters. As a consequence, there is an increased risk that the fish stock dies. If the water level falls below 0.7 m ponds are therefore usually taken out of production and the water is pumped to stabilise other ponds within the group. For simplification it is assumed that the total water supplied in the production period is proportional to the available pond volume and that the volume is proportional to the water depth of the standard pond. Thus in order to maintain minimum water levels in the range between 1.1 and 0.7 m at least 63.3 % of the demand has to be supplied. If the water deficit is greater, the pond area has to be reduced proportionally to ensure that the remaining pond area can be operated with water levels at the threshold level of 0.7 m. The proportion by which the pond area is reduced in size as a result of a water deficit is therefore calculated as follows:

$$Ared = if\,(\frac{\sum_{m=3}^{10} SQ}{\sum_{m=3}^{10} DQ} < mcrit; \frac{\sum_{m=3}^{10} SQ}{\sum_{m=3}^{10} DQ} * \frac{1}{mcrit}; 1) \qquad (18)$$

and

$$mcrit = \frac{WL\min}{WLn} \qquad (19)$$

where mcrit is the threshold defining the proportion of the standard pond volume below which the operational area of the ponds has to be reduced, WLmin and WLn the minimum required and the nominal pond water levels in [m] respectively.

Valuation of losses in the pond fisheries sector is based on a change in net income approach. Changes in fish yield are calculated on the basis of the specific productivity per pond area (SSL 2005). Changes in net income are estimated as follows:

$$L_{(ds,t)} = Apond_{ds} \cdot (1 - Ared_{ds,t}) \cdot ((Y \cdot P_t) - sC_t) \qquad (20)$$

where Apond is the total available pond area in [ha], Ared is the proportion of the pond area actually available, Y is the fish yield in [kg/ha], P is the price in [€/kg] and sC are the saved variable production costs in [€/ha].

Coping and adaptation options

Coping mechanisms in production are implicitly implemented by assuming that water deficit up to the specified threshold level can be compensated without a reduction in pond area. Beyond this threshold, the remaining water is used to ensure the operation of the remaining pond area.

Key variables for scenario and sensitivity analysis.

The key demand shift variable is the pond area. The key price variables are the produce price for fish and the saved variable costs.

Sprinkler irrigation of agricultural crops

Estimation of water demand and water use

The annual irrigation water demand for the irrigated areas is calculated on the basis of the FAO Method (Doorenbos and Pruitt 1975). Crop water requirements are calculated assuming a demand site with simplified evapo-transpiration and crop growth processes.

The relationship between evapo-transpiration of a specifc crop and the reference condition is integrated into a single coefficient kc.

$$ETp_{crp} = ETp_{ref} * Kc_{crp} \tag{21}$$

where ETp_{crop} is the potential crop evapo-transpiration in [mm] for crops, ETp_{ref} is the reference evapo-transpiration in [mm], kc is the specific FAO crop evapo-transpiration coefficient [-].

Based on this approach, the additional amount of water needed to supply the evapo-transpiration demand of the crops at a demand site is calculated as follows:

$$DQ_{ds} = \sum_{crp} ((\frac{1}{IrrEff}) * (Max:(0;(ETp_{ref} * kc_{crp}) - P)) * A_{crp,ds} * 0{,}1 \tag{22}$$

where A_{ds} is irrigated the area per demand site in [ha], $Fraction_{crp}$ is the fraction of the area covered by a specific crop, P is precipitation in [mm], IrrEff is the share of supplied water available for ET (i.e. irrigation efficiency) [-], DQ_{ds} is the total irrigation water demand at demand site ds in [m^3/s].

It is assumed that water demand for irrigation corresponds to effective water use, as all of the modelled water demand is evapo-transpirated and there are no return flows.

Loss function

The key effect of reduced water availability for irrigation is a reduction of crop yield. Crop yields are also calculated on the basis of the FAO Method (Doorenbos and Kassam 1979) in which a linear crop-water production functions is used to predict the reduction of crop yield when crop stress is caused by a shortage of soil water according to the following relationship:

$$1 - \frac{Ya}{Y\max} = ky * (1 - \frac{ETa_{crp}}{ETp_{crp}}) \tag{23}$$

where ky is yield response factor (-), ETa_{crp} is adjusted actual evapo-transpiration in [mm], ETp_{crp} is crop evapo-transpiration for standard conditions (no water stress) in [mm], Ya is actual crop yield in [dt] and Ymax is maximum expected or agronomically attainable crop yield under no stress [dt].

The adjusted actual evapo-transpiration is calculated for every month for irrigated (i) and optimally irrigated (io) conditions as follows:

$$ETa_{crp,irr=i} = \min(ETp_{crp}; P + (IrrEff * \frac{SQ_{crp}}{A_{crp}} * 10)) \tag{24}$$

$$ETa_{crp,irr=io} = \min(ETp_{crp}; P + (IrrEff * \frac{DQ_{crp}}{A_{crp}} * 10)) \qquad (25)$$

where SQ_{crp} and DQ_{crp} are the available and demanded irrigation water respectively for a specific crop in [m³/s] at a demand site calculated by the WBalMo model.

The actual yield can then be calculated from the annual sum of potential and actual evapo-transpiration for each irrigation condition:

$$Ya_{crp,irr} = Y\max_{crp} * \max(0; (1 - ky_{crp} * (1 - (\frac{\sum_{month} ETa_{crp,irr}}{\sum_{month} ETp_{crp}})))) \qquad (26)$$

Loss is calculated using a change in net income approach as follows:

$$L_{ds,t} = \sum_{crp}((Ya_{crp,irr=io,t} - Ya_{crp,irr=i,t}) * A_{crp} * (P_{crp,t} - sC_{crp,t}) - ((DQ_{ds} - SQ_{ds}) * sIC_t) \qquad (27)$$

where Ya is the actual yield in [dt/ha] under optimally irrigated condition (irr=io) and actual conditions (irr=i), A is the irrigated are in [ha], P_{crp} is the produce price in [€/dt], sC are the saved variable (harvest and post harvest) production costs in [€/dt], sIC are the saved variable costs for irrigation and water in [€/m³] and DQ and SQ are the water demand and supply in [m³] respectively.

Coping and adaptation options

There are no coping mechanisms implemented, as yield changes in proportion with the water deficit. Possible adaptation options that can be modelled include changes in irrigation efficiency or crop mix.

Variables for scenario and sensitivity analysis

The key demand shift variable is the irrigated area and crop mix. The key price variables are the produce price for crops and the saved variable irrigation costs.

Sub-irrigated and water table regulated lowland (fen) wetlands

Water demand and water consumption

Water demand is estimated with the wetlands water balance sub-model WABI. This model is described in detail by Dietrich et al. (2007). WABI is a water balance model for groundwater influenced areas with drainage and sub-irrigation systems. The wetlands areas are divided in sub-areas. A sub-area is the smallest area in which the groundwater level can be regulated separately and is a water user in the WBalMo model. One important assumption is a horizontal ground water level in each sub-area. The model needs target water levels, climatic conditions and inflows for each sub-area as an input and then calculates evapo-transpiration, demand for additional water, actual water levels and return flows.

Loss Function

Water table regulation in wetlands impacts simultaneously on a multitude of ecological functions associated with the wetland, all off which have to be considered in management decisions (Turner et al. 2003). The loss function takes the agricultural production function, habitat and biodiversity conservation function and greenhouse gas regulation function into account. Loss is calculated from the difference in benefit accruing from these functions at target water levels and actual water levels.

The loss of agricultural production function is based on the difference in yields. Five agricultural cropping systems are identified on the basis of a combination of land use data and water table levels. Arable land is assumed to be planted with the dominant crop, which is corn. Grassland is classified into four subtypes: intensive grassland (groundwater levels >= 0.45 m), extensive grassland (groundwater levels >= 0.45 m), extensive wet grassland (groundwater levels < 0.45 and > 0.2 m) and reed / conservation grassland (groundwater levels > 0.2 m). Yield is calculated using the FAO method as outlined above, whereby the sum of annual actual evapo-transpiration (ETa) and potential evapo-transpiration (ETp) is provided from the wetlands water balance sub-model. Yields are calculated for three irrigation situations: target water level, actual water level and without groundwater influence. Annual energy yield is calculated using the specific energy density of each crops biomass and the potential biomass as follows:

$$YE = \sum_{crp} Y\max_{crp} * \max(0; (1 - ky_{crp} * (1 - (\frac{\sum_m ETa_{(crp,m)}}{\sum_m ETp_{(crp,m)}}))) * ED_{crp} * A_{crp} \tag{28}$$

where YE is the annual energy yield in [MJ], Ymax is maximum expected or agronomically attainable crop biomass yield under no stress in [dt/ha], ky is yield response factor [-], ETa_{crp} is adjusted actual evapo-transpiration for the specific crop in [mm], ETp_{crp} is crop evapo-transpiration for standard conditions (no water stress) in [mm], ED is the specific energy density of the crop in [MJ ME/dt] and A is the area of crop in [ha].

Total annual energy yield is calculated for target and actual water levels for all crops in the wetland. Loss is calculated on the basis of the change in net income approach and the loss in gross margin from a production activity is used to reflect the loss to the farm enterprises. The loss in gross margin is calculated as the value of the lost output less any savings of variable costs from reduced harvest and storage costs. As the energy yield of biomass produced for fodder is not directly tradable in the market, a substitute price is used. It is assumed that deficit in energy yield of the fodder grown in the wetland during dry years is compensated by maize that would alternatively have been used for biogas production. The loss in energy yield is then valued at the shadow price of biomass for biogas production as follows:

$$L_{t,ds} = \sum_{crop,ds} (tYE_{crop,ds} - aYE_{crop,t,ds}) \cdot SED \cdot cf \cdot P_t) - (tY_{crop,ds} - aY_{crop,t,ds}) \cdot \Delta VC_t \quad (29)$$

where YE is the annual energy yield in metabolic energy [MJ ME], Y is the annual biomass yield in dry mass [dt DM] at target t and actual a water levels, SED is the energy density of the substitute crop's dry matter [MJ $dtDM^{-1}$], P is the shadow price of biomass for biogas production [€ dt FM^{-1}], cf is a correction factor (usually 1/33) to convert from dry matter (DM) to fresh matter (FM) and ΔVC is the saving in variable harvest costs of the crop [€ dt DM^{-1}].

We use estimates of the shadow price of carbon to assess the value of the greenhouse gas emission externalities associated with the wetland water management. Peat carbon sequestration is a result of low biomass decay rates under anaerobic conditions in water logged wetland sites. When wetlands are drained the peat is no longer conserved but decomposed, because lowering the water table stimulates aerobic decomposition of the peat. The main drivers controlling greenhouse gas fluxes of fen wetlands are related to aspects of hydrology. In order to estimate the greenhouse gas flux from wetlands we use a method presented in Grossmann and Dietrich (2011). This method combines water level dependant emission functions for the major greenhouse gases to estimate the global warming potential (GWP) measured in carbon dioxide equivalents. The GWP balance of the greenhouse gas exchange for wetlands is calculated as:

$$GWP_{CO2e} = (NEE_{CO2-C} \cdot GWP_{CO2-C} + F_{CH4-C} \cdot GWP_{CH4-C} + F_{N2O-N} \cdot GWP_{N2O-N}) \cdot 44/12 \quad (30)$$

where GWP is measured in CO_2e, $NEECO_2$-C is the net ecosystem exchange of carbon measured in CO_2-C, FCH_4-C is the mean annual flux of methane measured in CH_4-C, FN_2O-N is the mean annual flux of nitrous oxide measured in N_2O-N and GWP are the corresponding elementary global warming potentials. The factor 44/12 converts from Ce to CO_2e which is the usual accounting unit. The GWP balance is an atmospheric balance, so that emissions from wetlands have a positive and sinks a negative sign.

The shadow price of carbon can be used to measure the scale of the externality from greenhouse gas emissions (or the benefits of abatement) which needs to be incorporated into cost-benefit and policy appraisal. The rational of using a shadow price is to make policy and investment decisions across sectors comparable and has been proposed by growing number of governments (cf. DECC 2009, UBA 2007). The economic loss from increased greenhouse gas emissions of fen soils compared to the water level regulation target in year t are then calculated as:

$$L_{t,ds} = -SPC_t \cdot (\sum_{wl} tA_{wl,ds,} * GWP_{wl} - \sum_{wl} aA_{wl,ds,t} * GWP_{wl})) \tag{31}$$

where SPC is the shadow price of carbon in € t CO_2e^{-1}, GWP is the average annual global warming potential in t CO_2e^{-1}, wl is the mean annual groundwater floor level and A_{wl} is the fen area with water level wl under target tA and actual aA conditions and GWP_{wl} is the global warming potential at water level wl.

Changes in habitat or biodiversity quality of fen peat wetland sites are estimated using as an indicator the area of peat soils with an average annual water level less than 40 cm below surface. This water level corresponds to the minimum target water levels for wetland conservation as set out in the regional wetland management strategy (LUA 1997). Fen wetlands with lower water levels will degenerate at a fast rate and will ultimately loose their wetland status. The values attached by the general public to the conservation of wetland biodiversity and habitats are potentially considerable; however they are also extremely difficult to measure. Stated preference methods, such as contingent valuation or choice experiments, are the only techniques suited to derive estimates of the general public's preferences for habitat and biodiversity conservation that include non-use value components (cf. Haab and McConnell 2002). Non-use values derive from preserving natural heritage for future generations independent of any personal use of a site for example for recreation. However, it is difficult to separate non-use values from non-consumptive use values, because improvements of the wetland habitat and biodiversity quality may also improve the recreational use value of that part of the population that uses the wetland areas for recreational purposes.

All of the stated preference methods require interview surveys to elicit primary value estimates for a proposed policy or measure. Where there are no primary studies available, the transfer of benefit estimates either by direct value transfer or functional transfer is a second best strategy (cf. Bergstrom and Taylor 2006 for theoretical and methodological overviews). For this study we apply a meta-functional transfer method that is based on the systematic quantitative summary of evidence across empirical studies (cf. Grossmann 2010). The resultant meta-function is then adapted to fit the specifics of the policy site such as socioeconomic characteristics of the population, extent of market and scope of the wetland quality change.

Total annual loss from wetland habitat degradation is then assessed on the basis of the difference of area with high water table levels under target and actual conditions as follows:

$$L_{ds,t} = (nAwet - Awet_{t,ds}) * (\left[WTP \cdot \frac{POP}{HH} \right] / A) \tag{32}$$

where nAwet and Awet are the area of peat wetland with groundwater level below surface < 40 cm in [ha] for nominal water levels and actual conditions respectively, WTP is the average annual willingness-to-pay estimate for wetland habitat protection per household in [€ per HH], Pop is the population of the evaluated region in [persons], HH is the average household size in [persons per household] and A is the total area in[ha] for which the WTP estimate is valid.

The total loss for wetlands is the sum of losses from changes in agricultural production, greenhouse gas emissions and habitat functions.

Coping and adaptation options

The major adaptation options that can be modelled are changes of water level regulation targets and the associated changes in land use. Additional options include alternative water allocation rules within each of the wetland sites, for example to give preference to certain areas.

Key variables for scenario and sensitivity analysis

The key demand shift factor is the water level regulation targets. The key price variables are the produce price for substitute crops and saved variable costs for agricultural biomass production, the shadow price of carbon for the greenhouse gas emissions and the willingness to pay for wetland conservation.

Hydropower

Water demand and water consumption

As all reservoir release and run of river hydropower plants considered in this study are producing base load electricity which is in constant demand, monthly water demand is equated to maximum turbine flow capacity:

$$DQ_{(ds,m,t)} = QT\max_{ds}$$

where DQ is water demand in [m^3/s] and QTmax is the specific maximal turbine capacity in [m^3/s] of the hydropower plant. Return flows are equal to inflows.

Loss function

Electricity production for run of river hydropower plants is calculated as

$$kWh_{(ds,m,,t)} = \min(SQ_{(ds,m,t)} - (MNQ_{(ds)} \cdot rateMin_{(ds)}); QT\max_{(ds)}) \cdot h_{(ds)} \cdot 7 \cdot hrs \quad (33)$$

and for reservoir hydropower plants as:

$$kWh_{(ds,m,t)} = \min(SQ_{(ds,,m,t)}; QT\max_{(ds)}) * hres_{(ds,m,t)} * 7 * hrs \qquad (34)$$

where kWh is the monthly power production in [kWh], QTmax is the maximum turbine flow capacity in [m³/s], h is the fixed water head for run of river hydropower plants in [m], hres is the month's average head of water in the reservoir in [m], 7 is the efficiency factor in [kN/m³], SQ is the actual flow in month m and year t, *hrs* are the monthly operation time in [hours], MNQ is the average low flow at a run of river demand site [m³/s] and rateMin is the share of the MNQ that is required as minimum flow in the river bed.

Loss is defined as the difference in value of annual electricity produced at hypothetic maximal production and actual production levels. The price of base load electricity is:

$$L_{(ds,t)} = \sum_{m} (nkWh_{(ds)} - kWh_{(ds,m,t)}) * P_{(ds,t)} \qquad (35)$$

where nkWh is the nominal electricity production capacity of the hydropower plant in [kWh/month], kWh is the actual electricity production in [kWh/month], P is the applicable price in [€/kWh] at a demand site.

Key variables for scenario and sensitivity analysis

The key price variable is the assumed price of base load electricity at a demand site.

Recreational boating on inland waterways

Estimation of water demand and water use

The demand for water at locks is a function of the rate at which a lock operates and the specific volume of the lock. Locks may operate at regular intervals or on demand. From observed data on number of boats and the number of locking events per month, an empirical relationship between the number of boats passing a lock and locking events was determined, whereby the locking frequency increases with increasing number of boats and asymptotically approaches the operational maximum locking rate Cmax. It follows that:

$$LE_{ds,m} = ((\mathrm{Min}((\mathrm{N_{ds,m,t}} * \mathrm{PAR\_A_{ds}} / \mathrm{PAR\_B_{ds}} + \mathrm{N_{ds,m,t}}); \mathrm{C_{ds}} * 13 * 30) \qquad (36)$$

where LE is the number of locking events per month, N is the number of boats passing a specific lock, C is the maximum frequency of locking events per hour [LE/h] multiplied by the daily operating period of 13 hours and 30 days a month, PAR_A and PAR_B are specific parameters of the function. The monthly flow requirement DQ in [m³/s] for locking can then be determined from the volume and number of locking events plus a height dependant seepage factor of 0.005 m^3/s per m as follows:

$$DQ_{ds,m} = ((LE_{ds,m} * \frac{V_{ds}}{2}) / (60*60*24*30)) + (0{,}005*\mathrm{H}_{\mathrm{ds}}) \tag{37}$$

where V is the volume of the lock in [m^3] and H is the difference in water level to be overcome [m]. The return flow is equal to the supplied flow.

Loss function

The major effect of reduced availability of water for locking is a reduction in the number of possible locking events per month, which can be determined from rearranging the equation for water demand:

$$LE_{(ds,m,t)} = (\mathrm{QS}_{\mathrm{ds,m,t}} - (0.005*\mathrm{H}_{\mathrm{ds}}) * (60*60*24*30)/\mathrm{V}_{\mathrm{ds}} * 2 \tag{38}$$

A reduction in the amount of locking events will lead to increased waiting times and congestion effects especially during peak usage in summer months. Changes in waiting time result from an increase in the time span between locking events. Assuming random arrival and deterministic service during the 13 daily hours of operation of the locks, the total waiting time is approximated by the average service time (Ts):

$$T_{ds,m,t} = \mathrm{Ts}_{(\mathrm{ds,m,t})} = \frac{1}{\mu_{ds,m,t}} \tag{39}$$

and

$$\mu_{(ds,m,t)} = \frac{LE_{ds,m,t}}{30*ot} \tag{40}$$

where μ is the service rate in [locking events per hour] and ot is the daily operation time of locks in [hours].

Increases in waiting time are valued using a consumer surplus approach. Annual changes of total consumer surplus induced by changes in water availability are then estimated as follows:

$$L_{ds,t} = \sum_{m} (\mathrm{nT}_{(\mathrm{ds})} - \mathrm{T}_{(\mathrm{t,m,ds})}) * \mathrm{N}_{(\mathrm{t,m,ds})} * \mathrm{WTP} \tag{41}$$

where nT is the nominal average waiting time under current conditions in [hours/boat] and T is the average waiting time under actual conditions in [hours/boat], N is the number of boats passing a lock and WTP is the willingness to pay to avoid waiting time in [€/hour and boat].

Consumer preferences for a reduction in waiting time at locks were estimated on the basis of an empirical on site survey of boaters at locks along the Havel waterway. Details of the survey are given in Meyerhoff & Grossmann (2007). Willingness-to-pay (WTP) for avoiding waiting time at locks was elucidated using a choice experiment format.

Variables for scenario and sensitivity analysis

The key demand shift variable is the annual lock utilisation (or visitation) rates and the key price variable is the per visit recreational value.

Transport shipping

Water demand and water consumption

There is no direct water demand function for shipping because it uses the residual available in-stream flows of the main river. However there are minimum flow requirements implemented along the Elbe trajectory that have a high ranking in the allocation priority list.

Loss function

With decreasing depth of the channel the load capacity of vessels decreases, while costs for personnel and fuel only decreases less than proportionally. This implies that transport costs increase with lower water levels. The change in transport costs is used to value the change in water availability. The economic assessment approach that we use in this study was originally developed for the benefit-cost analysis of infrastructure projects in the context of the compilation of the Federal Transport Infrastructure Plan (BMVBW 2005). The modelling approach is described in greater detail in Möhring and Grossmann (2010 in prep.).

The approach is based on a transport network model onto which the total goods to be transported are projected. The network model describes for every transport relation all important parameters like channel depth, locks, flow velocity characteristics and maximal speed. The total load is differentiated according to source and target region. From the load the required number of vessels turn around trips is calculated for every relation. This process takes the fleet structure and the maximal load capacity of different vessel types into account.

The number of required turn-around trips F (b,l) is calculated as follows:

$$F(b,l) = \frac{VH}{\sum_i AF_i \times TT_i \times AR_i \times \frac{\sum_{perz=1}^{100} \min(1; \frac{Tr_{perz} - C - To_i}{Ts_i - To_i})}{100}} \times (1 + LF) \tag{42}$$

where VH is the load in the direction with more goods to be transported (in t/a), LF is an empirically determined share of empty trips, AF describes the share of a vessel class i of the total capacity weighted fleet, TT is the maximum load of a vessel of class i (in t), AR is an empirically determined factor to account for operations, Tr is the channel depth (in m) for the percentile, C is the minimum required water depth (in m), To is the draught of the empty vessel (in m), Ts is the draught of the fully loaded vessel (in m).

The channel depth is calculated for each of 100 percentiles of the monthly discharges from 500 simulated months per five year period for the depth limiting river stretch for each transport relation. In a second step the maximum number of turn around trips a vessel of size class i can accomplish on each transport relation per year:

$$UJ_i = \frac{Bh_i}{Stz + Sz + Uz} \tag{43}$$

where Bh are the annual operating hours of a ship (in h/a), Stz und Sz are travel and locking time and Uz is the harbour time (all in h per trip). The number of required vessels (AS) per size class is then calculated from the share of the size class in the fleet and the required number of trips per year and size class (UJ_i).

The transport costs are determined as the sum of overhead or fixed costs VK and variable costs BK. The overhead costs are calculated as the product of required number of vessels (AS) and the overhead costs rate (VKS):

$$VK_i = \sum_i^n AS_i \times VKS_i \tag{44}$$

The total variable costs [BK in €] are calculated from the travel or motor time [Stz in h], the specific fuel costs per time unit [BKS in €/h], the time dependant variable costs [ZK in €/h] and the annual operating hours [Bh in h]:

$$BK_i = \sum_{i=1}^n Stz_i \times F(b,l)_i \times BKS_{i,w} + \sum_{i=1}^n ZK_i \times Bh_i \times AS_i \tag{45}$$

The total transport costs are therefore a function of the amount of goods transported and the water level dependent degree of capacity utilisation of the vessels. We calculate the average annual loss compared to a target water level, as it is described in the development goal of the Federal Transport Infrastructure Plan 2003 (cf. BMVBW 2005) and that is expected after completion of current infrastructure works such as channel dredging. This target (GLW_{1995}) is defined as a minimum draught of 1.6 m on 345 days, and of 2.5 m more than 50 % of the year.

Coping and adaptation options

The coping mechanism implemented in the model is the increase in vessel turn around trip frequency and for those relations that can use an alternative route using water regulated canals, the choice of route. The most important adaptation options that can be modelled are a change of the fleet structure and works on the waterway infrastructure.

Key variables for scenario and sensitivity analysis

The key demand shifting variables are the total goods that are to be transported by ship and the fleet structure. The key prices are the variable costs for ship operation.

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