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The European Water Framework Directive and Economic Valuation of Wetlands

The Restoration of Floodplains along the River Elbe

Von

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ABSTRACT / ZUSAMMENFASSUNG

The European Water Framework Directive and Economic Valuation of Wetlands
The Restoration of Floodplains along the River Elbe

This paper concerns the economic valuation within the European Water Framework Directive (WFD) of riparian wetlands ecological services, such as the protection of biodiversity. The Directive is an integrated approach to river basin management in Europe and aims at achieving a good water status for both surface and ground water. It introduces economic analysis as a core part of the development of integrated river basin management plans. Among other things, economic analysis should guide decision making on which measures are employed to achieve the good water quality or should justify possible derogation. However, currently the WFD does not clearly state to which extent wetlands should be used for the achievement of environmental objectives, nor which scope the economic analysis should have. While recently a guidance document on the role of wetlands stressed the importance of wetlands in river basin management, our study about the river Elbe shows that riparian wetlands provide significant benefits that should be considered in river basin management decisions. To neglect these benefits would lead to a misguided decision-making process and would result in an inefficient supply of ecological services.

Die europäische Wasserrahmenrichtlinie und die ökonomische Bewertung von Feuchtgebieten
Die Wiedergewinnung von Überflutungsflächen entlang der Elbe

Der Beitrag behandelt die ökonomische Bewertung ökologischer Leistungen, wie den Schutz biologischer Vielfalt, im Kontext der Europäischen Wasserrahmenrichtlinie (WRRL). Die Richtlinie stellt einen integrierter Ansatz zum Flusseinzugsgebietsmanagement in Europa dar, mit dem das Ziel, der gute ökologische Zustand sowohl für Grund- als auch für Oberflächenwasserkörper, erreicht werden soll. Entsprechend der Richtlinie ist die ökonomische Analyse ein Kernbestandteil des Flussgebietsmanagements. Sie soll zur Entscheidungsfindung darüber beitragen, welche Maßnahmen zur Erreichung der Ziele eingesetzt werden sollen und in welchen Fällen eine Abweichung von den Zielen begründbar ist. Jedoch wird in der WRRL weder eindeutig geregelt, welche Rolle Feuchtgebieten im Flussgebietsmanagement zukommt, noch, ob die ökonomische Bewertung auch die Betrachtung nicht-marktlicher Nutzen umfasst. Die Ergebnisse einer ökonomischen Bewertung von Überflutungsflächen entlang der Elbe zeigen, dass der Nutzen aus Feuchtgebieten beträchtlich sein kann. Wird dieser Nutzen beim Management der Flussgebiete nicht berücksichtigt, kann dies zu einem zu geringen Angebot an ökologischen Leistungen führen.

Contents

1. Introduction

By creating a new directive (2000/60/EC) to establish a common framework for the management of the aquatic environment, the European Union has entered a new phase in water policy. The Water Framework Directive (WFD) is an integrated approach to river basin management in Europe. Its main objective is to achieve a 'good water status' for both surface water and groundwater. Economic aspects of water management have been assigned a key role in the implementation process. Accordingly, the WFD requires an economic analysis for each river basin district (as it is stated in Art. 5) by 2004 and the use of economic methods and instruments to support decision-making.

However, several uncertainties concerning the implementation of the WFD still remain. Two things are of particular importance regarding wetland management. Firstly, the WFD does not clearly state the extent to which the protection, as well as the enlargement, of wetlands should be used for the achievement of the WFD environmental objectives. It is surprising that wetlands are not recognised explicitly in the WFD, considering that wetlands have been generally declining for decades - both in area and quality - in Europe, a trend associated with negative effects for the water environment in general. The European Environment Agency (EEA), for example, states that despite the global and national recognition of their importance, Europe's wetlands remain under severe pressure from land use and pollution (EEA 2000, CHAPTER 14).

Secondly, the WFD does not completely specify the scope of the economic analysis, which is also relevant for wetland management. Wetlands such as riparian flood plains provide many important services to society (Turner et al. 2003). In general, three kinds of services are recognised: hydrological functions (groundwater recharge, floodwater detention), biogeochemical functions (nutrient or carbon retention) and ecological functions (maintenance of habitats). As most of these services are not traded within any market, no market price is available which reflects their economic value. Thus it is not yet clear whether water related environmental benefits arising from the ecological services provided by riparian wetlands will be considered adequately in the economic analysis. Currently the scope of the economic analysis as required by 2004 is under discussion in several EU Member states (e.g. Strosser 2003). Among other topics, whether or not benefit assessment of non-market goods is a necessary input into the economic analysis is being discussed. The WFD's having a positive outcome for biodiversity and wetland protection depends on that regulation, too.

Present valuation studies have demonstrated that the economic value of ecological services of riparian wetlands can be significant (e.g. Loomis et al. 2000). Unfortunately, only a few studies concerning the non-market benefits of riparian wetlands relate to rivers in Europe. And those that do exist are distributed very asymmetrically over the EU member countries. For example, no such valuation study has been available for Germany so far. Furthermore, to our knowledge, a comprehensive benefit-cost-analysis for a biodiversity conservation program on riparian wetlands was conducted in only two studies. Kosz (1996) estimated the benefits of establishing a national park, "Donau-Auen", near Vienna in different variants. He compared the benefits arising from the different variants with the benefits which would arise, for example, from hydroelectric power stations. Amigues et al. (2002) determined the cost and benefits to provide a strip of riparian land for habitat preservation at the Garonne River near Toulouse. Both studies used the contingent valuation method (CVM) to determine the benefits from biodiversity protection.

A framework currently widely used for valuing natural resources is the Total Economic Value (TEV). It comprises not only use and non-use values people may derive from regained flood plains, for example, but also indirect use values (Turner et al., 2003). The indirect use values derive from services provided by wetlands and include the removal of nutrients or the prevention of downstream flooding. Therefore, in addition to the CVM, in the present study we applied the replacement cost approach (RCA). The CVM was used to assess the benefits arising from a biodiversity preservation program on riparian wetlands. It comprised the regaining of 15,000 ha flood plains by dike shifting at different sites along the Elbe, as well as the reduction of the negative environmental effects resulting from intensive agriculture on another 40,000 ha flood plains. In addition, the RCA was used to assess the benefits from nutrient removal provided by the 15,000 ha regained flood plains.

This paper has two main objectives. The first is to show that the restoration of riparian wetlands provides significant benefits of value to society. Furthermore, it emphasises the role wetland services can play for the achievement of the WFD's objectives. Accordingly, it argues in favour of considering the environmental benefits within the economic analysis of the WFD. The paper is structured as follows: Section 2 outlines the economic analysis within the WFD and the role environmental valuation might play. The next section presents the study area and the management actions. In Section 4 the valuation methods used and their application are introduced. Results from both methods used as well as from the ultimately conducted benefit-cost-analysis are presented in Section 5. Some conclusions are drawn in Section 6.

2. EU Water Framework Directive, wetlands and environmental valuation

The Water Framework Directive (WFD) applies a holistic approach in the management of water, integrating and balancing ecological, societal and economic demands at the river basin scale. The overall aim of the directive is to establish a framework for the implementation of sustainable water management strategies for long-term protection of water resources. The main objectives are to protect and enhance the aquatic environment and to achieve a 'good water status' for all surface waters and groundwater by 2015. To meet these requirements, the WFD integrates key principles in the water policy, such as the involvement and participation of stakeholders, the management at the basin scale (with implications for administrative change), and the integration of the economic dimension of water management.

Concerning the discussion as to whether wetlands have to be included in the river management plan or not, the WFD does not clarify whether wetlands, as important parts of the water environment, are relevant to the WFD implementation and in which way the protection and restoration of wetlands could contribute to the achievement of the environmental objectives. Although the WFD encompasses the protection of wetlands as a part of its purpose in Art. 1[1], it does not set recommendations for wetland restoration and management per se (CIS Wetlands Horizontal Guidance, 2003). The environmental objective 'good water status' refers only to the surface water and groundwater itself (Art. 4). But, as the WFD directs attention to the relationship among different elements and ecosystems of the water environment, a significant role of wetlands can be derived. The structure and the state of the riparian zone directly influences the biological and hydromorphological quality elements of the 'good water status', therefore adjacent wetlands such as flood plains can be identified as a part of the surface water body. Furthermore, wetlands are part of the terrestrial ecosystems directly dependent on groundwater bodies (UFZ 2002, CIS Wetlands Horizontal Guidance, 2003). Hence, management plans to achieve a 'good water status' can include protection or restoration measures for wetlands where restoration is required to ensure that the hydromorphological conditions are suitable for the biological quality elements. Wetlands are to be considered where wetland creation or restoration is the most cost-effective approach to achieve the environmental objectives compared to other alternatives. According to the justification of potential derogation, an extension of a flood plains area could be assessed as the 'significantly better environmental option' (Art. 4) to reach the same beneficial objective, for example flood

[1] Art. 1 states that the Directive will "…establish a framework […] which: prevents further deterioration and protects and enhances the status of aquatic ecosystems and with regard to their water needs, terrestrial ecosystems and *wetlands* directly depending on the aquatic ecosystem."

protection, compared to the existing physical alterations for flood protection like dikes or embankments (see below). In summary, the WFD does not set independent environmental objectives for riparian wetlands but identifies the use of their ecological services as a possible means for achieving good water status.

According to the integration of economics, the WFD requires the application of economic principles (e.g. polluter pays principle), the use of methods and tools (e.g. cost-effectiveness analysis) as well as the consideration of economic instruments (e.g. water pricing) to achieve the environmental objectives and aid decision-making (CIS WG WATECO, 2003). Within the context of the WFD, economics are mainly integrated by

- the economic analysis of water uses (Article 5),
- the assessment of the recovery of the costs of water services (Article 9) and
- the justification of possible derogation from the Directive's environmental objectives (Article 4).

The economic analysis should provide sufficient information to assess the cost-effectiveness of potential measures for reaching a good water status. Thus, the WFD focuses on the cost-side. The assessment of the benefits associated with improving water quality is not explicitly considered in the economic analysis. But, since the potential measures to achieve a good status are generally associated with high costs, many management options could be implemented only if the benefits are considered; otherwise they often will be averted by reason of expense.

From our point of view, the assessment of benefits is necessary to meet the requirements of the WFD in several cases:

(1) The recovery of the costs of water services should include environmental and resource costs. The value of riverside wetlands and their important role for biodiversity protection, as discussed in this article, are included in the WFD through the estimation of environmental and resource costs. As water services encompass the impoundment and storage of surface water for the purposes of hydropower generation and navigation, the associated negative impacts for flood plains, such as the disconnecting or loss of flood plains, can be considered as the environmental costs of the water service.

(2) The justification of possible derogation (time and objective) as well as the designation of Heavily Modified Water Bodies (HMWB) require the assessment of costs and benefits from an economic point of view, because the question whether or not measures to reach a good status entail disproportionate costs cannot be answered without considering benefits versus costs (Petschow & Dehnhardt 2004). If the measures to achieve a good water status will have significant negative effects on speci-

fied uses, a comparison of different alternatives to reach the same beneficial objective is required before a surface water can be designated as heavily modified (Art. 4). In this assessment, the overall net benefit to society of the specified uses which caused the changes in hydromorphology, and of the alternatives to reach the same beneficial objectives, is compared, including the environmental impacts and costs of these options. Accordingly, benefits gained from the higher ecological status are to be included (CIS WG HMWB). Thus, the case of potential derogation demands a benefit-cost analysis.

(3) The WFD does not provide any specification of the methodological framework that is required for justifying potential derogation or to assess the environmental and resource costs of water services. To support the implementation process regarding the role of economics, a specific group on water economics (CIS WG WATECO) was created. Their guidance stresses the importance of benefit assessment for the evaluation of disproportionate costs, as well as for the estimation of environmental and resource costs. Following WATECO, the economic analysis refers to the overall costs and benefits to society when the programmes of measures and their economic impact have to be identified. The evaluation of whether costs are disproportionate or not is a key input into the preparation of the river basin management plan, and accordingly is one of the important functions of the economic analysis within the WFD. Therefore, based on the WATECO guidance, one can argue that the value of flood plains should be included within the economic analysis.

As was already mentioned in the introduction, the scope of the economic analysis in the phase of the WFD implementation process after 2004 is currently being discussed in several EU Member states. One topic of this discussion is whether non-market benefits should be assessed, and if so, in which cases.

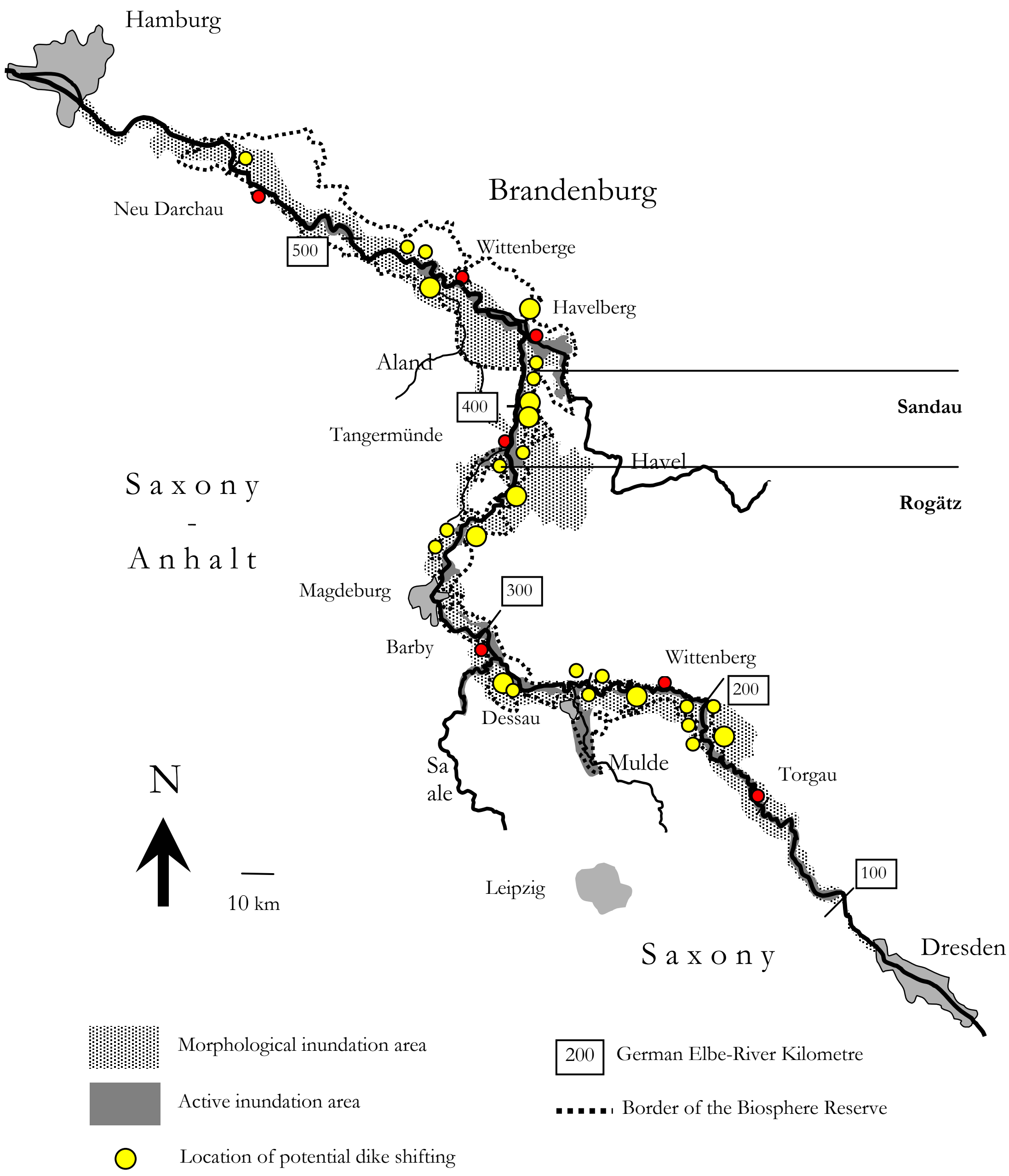
3. Study area and management actions

With a length of about 1.100 km and a catchment area of almost 150.000 $km^{2,}$ the Elbe River is one of the largest rivers in Central Europe. About one third of the catchment area and 365 km of its length are within Czech territory, two thirds of the catchment area and 727 km of its length are within Germany. Furthermore, some small parts of the catchment area can be found in Austria and Poland. Although today more than 80 % of the original inundation areas are cut off from the river by dikes, the Elbe riverscape still has many reaches in a near-natural state. Some of these reaches are protection areas of international significance. One major reason for this is that only one dam can be found on the Elbe, in Germany. Located in Geesthacht, it is used to regulate tidal fluctuations. The Elbe is therefore one of the last large rivers in Central Europe which is free flowing for several hundred kilometres. Accordingly, one of the few remaining continuous expanses of largely original flood plain forests, for example, can be found on the flood plains of the Elbe. Recognition of the Biosphere Reserve "Elbe riverscape" by UNESCO in December 1997 highlights the importance of the Elbe basin as a natural landscape and its importance for biodiversity protection in Germany, as well as in Europe.

Within the last decade, many national and international research efforts were initiated on the Elbe and its tributaries (Gruber & Kofalk 2001). For example, in 1990 the Federal Ministry of Research and Education initiated the Lead Project "Elbe 2000" constituting the contribution of research and development to the rehabilitation of the river Elbe. The focus of this initiative was primarily on water quality. The "Ecological Research in the Elbe Catchment Area (Elbe Ecology)" was started in 1995 as a successor to the "Elbe 2000" project. This time the research was mainly devoted to devising future-oriented use and development concepts that help to preserve or improve the ecological functionality of the river. The research priorities of the research programme were the "Ecology of Flowing Waters", "Ecology of the Flood plains", and "Land Use in the Catchment Area". As the later initiative also intended to provide a sound basis for decision-making over development alternatives within the Elbe catchment, several socio-economic projects analysed the economic impacts of various measures, such as regaining flood plains by shifting dikes along the Elbe. However, these projects focused only on the costs of certain management actions, such as land use changes and dike shifting.

Based on the results of the research projects concerning the ecology of the flood plains, three management actions were bundled into a program: regaining 15,000 ha riparian wetlands by dike shifting at certain locations along the Elbe (Figure 1), reducing the impact of the current land use (predominantly agricultural) on existing as well as future riparian wetlands, and constructing fish ladders in tributaries in order to allow migratory fish species to arrive at upstream reaches of some Elbe tributaries. This bundle of management actions was presented within the CVM as a measure to improve the conservation of endangered species and habitats along the Elbe. Furthermore, the assessment of the additional nutrient removal was based on the 15,000 ha of new flood plains. Due to the availability of site-specific data concerning the morphological structure of the flood plains and the inundation dynamics of the river in particular, the RCA was first applied at the sites of Sandau and Rogätz (Figure 1). Subsequently, the nitrogen reduction effects were estimated for the remaining locations.

Figure 1: The river Elbe between Hamburg and Dresden and locations for regaining riparian wetlands by dike shifting



Source: according to Scholten et al. (2004)

4. Methods

4.1. Contingent Valuation (CV)

The CVM is still the most often used stated preference technique in environmental valuation (Garrod & Willis, 1999, Bateman et al., 2002). As it is survey based, respondents are asked directly whether, and if so, how much they are willing to pay (WTP) in order to improve environmental quality or to avoid its degradation. In using hypothetical markets to determine people's WTP, CVM unites two advantages and a disadvantage. It allows a valuation of future stages of the environment as well as the measurement of those benefits which are not associated with any kind of use. These non-use values arise simply from the knowledge that, for example, a resource is maintained. They can form a substantial part of the TEV of the environmental commodity in question. On the other hand, respondents do not have actually to pay the amount of money they state in the survey. Accordingly, the hypothetical nature of the stated willingness to pay is still the starting point of the debate about whether CV produces accurate, reliable and consistent estimates of the value which individuals place upon environmental commodities (Bateman & Willis, 1999). Although this debate has been taking place now for more than twenty years, it is still not settled. But as Randall (1998) points out, there is no crucial experiment which allows us to conclusively answer the question "Does CV work?". Accordingly, one should not expect to get a final answer from a further CVM study. Each study will only provide a mosaic of something Randall called preference mapping.

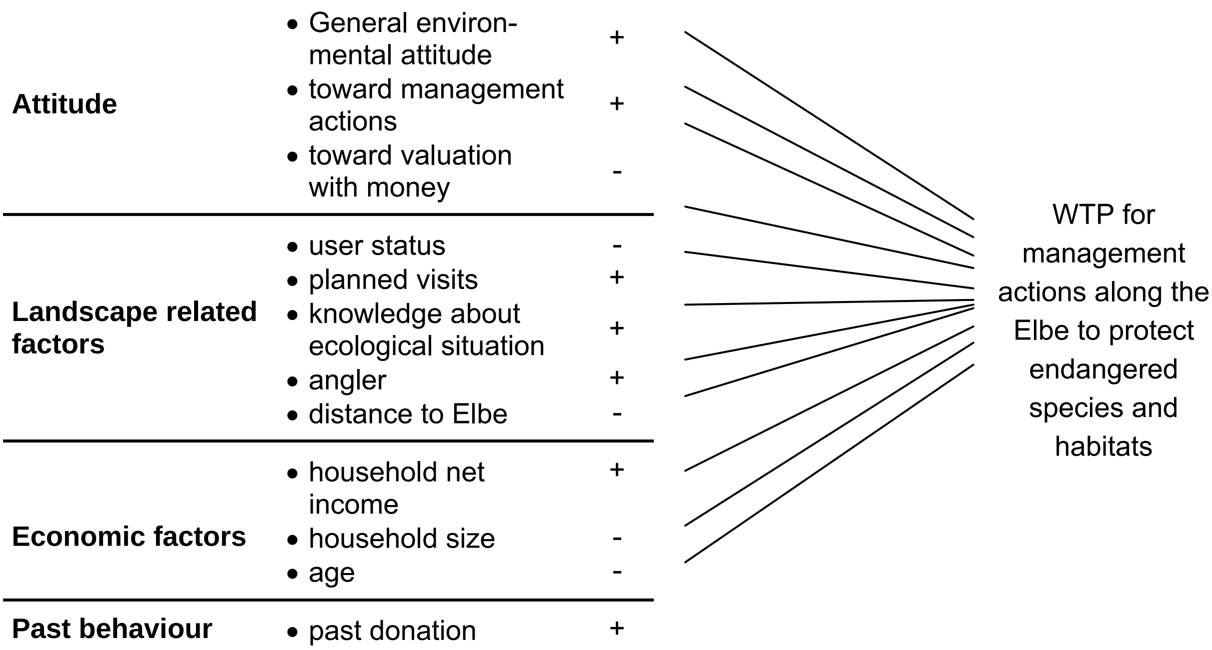
One objective of the present CVM was to investigate the influence of attitudes on stated willingness to pay. Attitudes are being used more frequently in CVM in order to increase construct validity. But so far it has not been clear which kind of attitude is more suitable for CVM. Are general environmental attitudes reliable predictors of the behaviour "paying money for public good", or should specific attitudes toward the commodity in question or toward the behaviour be determined? To analyse the influence of different kinds of attitudes, three types of attitudes were added to a set of independent variables comprising economic predictors, landscape related predictors and past behaviour (Figure 2).[2]

In order to elicit people's WTP for improving the preservation of engendered species and habitats along the river Elbe, 1,304 households within the catchment areas of the three German rivers, the Elbe (700), the Weser (300) and the Rhine (300), were interviewed based on random samples for each catchment (cf. Figure 3). The reason for interviewing respondents in the Weser and Rhine catchment areas was

[2] For a more detailed discussion of this topic and the estimation of a composite attitude-behaviour model within a CVM see (Meyerhoff, 2002).

twofold. The inclusion of non-users in the interviews was intended to determine the non-use value of the management actions. With this approach we followed Cameron (1992), who argues that in order to elicit the TEV, the WTP of non-users is crucial, rather than the non-use value users might derive from the commodity in question. We expected that the percentage of non-users would increase parallel to the distance to the Elbe. Moreover, we intended to investigate whether respondents in the latter two catchment areas would be willing to pay for similar management actions, but at the Weser and the Rhine respectively.

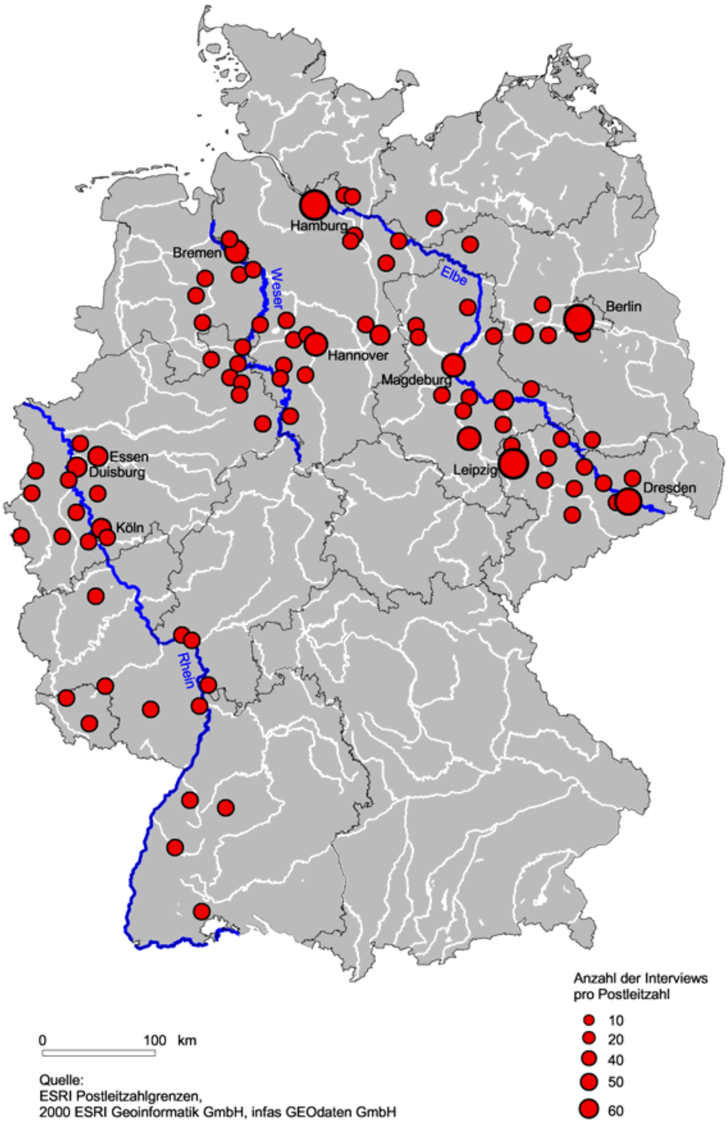
Figure 2: Conceptual model determining WTP



The questionnaire began with questions about the respondents' relation to the river Elbe, i.e. whether they visited the Elbe prior to the interview, and if so, which location they visited, or whether they planned to visit the Elbe within the coming 12 months following the interview. In order to assess their knowledge about the ecological state of the Elbe, they were presented with seven statements. For example, they were asked whether they thought the statement "The water quality of the Elbe has decreased during the last ten years" was correct or not.

Next, respondents were presented the items of a modified and shortened version of the New Environmental Paradigm. This scale was used as a measure of the respondents' general environmental attitudes. In the following section of the questionnaire, a description of the management actions was provided. Afterwards, respondents were asked about their attitudes toward the management actions and whether they would be willing to pay in principle in order to finance them. Those who responded positively were subsequently asked how much they would be willing to pay per month, using an open-ended question[3]. They were told that the biosphere reserve “Riverscape Elbe” would receive the money and will be responsible for the implementation of the program. In order to be able to correct for potential embedding, a disembedding question following Schulze et al. (1998) was used. Finally, standard socio-economic data such as people’s age, income, number of people per household, etc. were collected.

Figure 3: Distribution of sample points



The questionnaire was pre-tested in all three catchment areas. After minor adjustments had been made based on the results of the pretest, the main survey was carried out throughout the months of May and June 2001. People were interviewed face-

[3] In addition, respondents were also presented with a payment card to help them determine their WTP.

to-face, with an interview lasting on average 30 minutes. Both the pretest and the survey were undertaken by a professional market research firm.

4.2. Replacement Cost Approach (RCA)

The RCA was used to assess indirect use values, such as the benefits derived from the nutrient removal in flood plains. The CVM as a direct valuation method requires as a fundamental prerequisite that individuals are able to understand the commodity or service in question and have high perceptual skills. The suitability of the CVM remains unclear when highly complex and hardly perceptible ecological functions have to be valued. Accordingly, the CVM may not be the first choice for a sound valuation of defined ecosystem services such as the nutrient retention function. In general, using the RCA, the monetary value of a natural resource is assessed by how much it will cost to replace or restore it after it has been damaged (Garrod & Willis 1999). As a reference condition usually an environmental quality level is considered, e.g. a defined water quality. If the ecosystem is damaged and therefore can not provide this quality, suitable alternative measures have to be taken into account, e.g. the building of sewage treatment plants. Thus if specific functions of an ecosystem can alternatively be achieved by a technical substitute, then the costs of this substitute to replace this function can be regarded as the economic value of the ecosystem's service (Gren et al. 1994; Byström 2000; Mitch & Gosselink 2000). The RCA generally requires the following three steps:

1. Identification and quantification of the nitrogen reduction effects (estimating the ecosystem function),
2. definition of the reference scenario (i.e. the substitute and costs), and
3. economic valuation (assessing the ecosystem service).

Many studies which aim to assess the nutrient retention capacity of wetlands have shown that wetlands in general as well as river systems and flood plains in particular have a substantial potential for the improvement of water quality (Mitch & Gosselink 2000; Bystroem 1998; Jansson et al. 1994; Kronvang et al. 1999). The nutrient retention function refers to the self-purification potential of streams, which depends on the river's structure and its buffer strips (Kronvang et al. 1999). Due to the enlargement and the restoration of flood plains, the stream velocity is affected and thus, as a result of higher water residence time, the denitrification rate increases (Behrendt & Opitz 2000). Hence, one of the assumed benefits of dike relocation is an increase of nitrogen retention in the river system.

The amount of the nitrogen retention of restored flood plains is mainly affected by the additional flooded area as well as the denitrification rate. In the case of flood plains, the surface area available for nitrogen reduction is primarily determined by the inundation dynamic of the river, i.e. the duration and frequency of flooding, as well as the morphology of the flood plain surface. As a result of the enlarged area with its typical structure, the running velocity decreases as an important parameter known to influence the denitrification. Accordingly, the results vary substantially, depending on site-specific conditions that have to be taken into account. Therefore the nitrogen reduction effect was quantified for two defined relocation sites at the Elbe, in Sandau and Rogätz, in an initial step because of the availability of site-specific data.

A statistical model from Behrendt and Opitz (2000) is used to quantify the nitrogen retention within the flooded area resulting from the decreased running velocity. A model for the N-retention has been derived in this study, based on empirical data on the emission and transport of the nutrients of 100 different rivers in Europe. The nutrient retention is linked to the load (EN) and the load weighted nutrient retention (RL) by Equation (1).

$$R_N = E_N – [E_N / 1+R_L)] \qquad (1)$$

The load weighted nutrient retention (R_L) can be described by the following statistical model:

$$R_L = a * x^b \qquad (2)$$

where x is the hydraulic load and a and b are the coefficients of the model. Accordingly, the nutrient retention depends on the parameter specific runoff and hydraulic load. For river basins with a size of more than 10,000 km2 (such as the river Elbe), the specific coefficients are a = 10,9 and b = - 0,94. The model explains 75% of the variance of RL. The hydraulic load [m/a] is defined as the runoff in relation to one square meter water surface of the river system. Hence, the hydraulic load is lower in a free flowing river compared to a channelled one and could therefore be regarded as a measure for the morphological structure of the river. In order to determine the retention rate (RL), according to equation (2), the hydraulic load has to be estimated for the observed project areas. The total retention [t/a] is calculated following equation (1), using the specific runoff [m/s] in the flood plain and the nutrient concentration [mg/l] to determine the nutrient load E [t/a]. In a second step, the assessment of the replacement cost value requires the determination of the technical substitute and the costs to provide an equivalent service.

5. Results

5.1. Benefits due to an enhanced protection of endangered species and habitats (Contingent Valuation)

Of those interviewed, 66.3 per cent had visited the Elbe at least once prior to the interview and were classified as users. This group consists of 238 residents living close to the Elbe, i.e. only a few kilometres away, and 627 living more remotely. The remaining 33.7 per cent are non-users (439 respondents). Their percentage increased along with their distance to the Elbe. Of those living in the Elbe catchment, 15 per cent are non-users while 35 per cent of those living in the Weser catchment and 49 per cent of those living in the Rhine catchment respectively are non-users.

When asked if they would be willing in principle to pay in order to implement the management actions, 22 % responded positively (n = 289). The percentage of those WTP in principle varies from 23.4 % in the Elbe catchment to 18.3 % in the Rhine catchment. However, the chi-square statistic does not allow rejection of the null hypothesis that the place of residence has no influence on the likelihood that a respondent is WTP in principle. Those who were not willing to pay for the presented management actions were asked whether they would be WTP for similar measures at a different river. It is interesting that, contrary to our expectation, each time only a very small number of people in the Weser or Rhine catchment areas stated a WTP for similar management actions at the Weser or the Rhine respectively. Only 8 people each time were willing to pay for measures along one these rivers. The low number of people is perhaps due to the fact that people are aware of the unique value the Elbe has as a nature reserve. In the pretest some people stated as a reason for their WTP that errors once made with rivers like the Rhine by preparing them for inland navigation, for example, should not be repeated with the Elbe.

The mean willingness to pay for the total sample was 11.9 € per household per year. It varies from € 11.7 in the Elbe catchment to € 13.1 in the Weser catchment. In the Rhine catchment the mean WTP is € 11.2. The Kruskal-Wallis test shows that the catchment area has no significant influence on the stated amount of money. In contrast, whether a respondent is a user or not significantly influenced WTP. While 27.1 per cent of the users were WTP in principle, this percentage diminished to only 12.5 per cent of the non-users. Furthermore, mean WTP of the users was € 14.9. Non-users stated a mean WTP of € 5.9.

Table 1 reports the potential explanatory variables used in the analysis of the bid function underlying the WTP responses. The third column of the table indicates the expected influence of the independent variables on stated WTP[4], and the last column reports the mean for each variable.

Table 1: Potential explanatory variables and mean

Group	Variable	Expected influence	Definition	Mean
Economic variables	AGE	-	Age of interviewee	49.88
	INCOME	+	Available income per household (mid class)	3340.79
	HHZ	-	Number of members in household	2.25
Landscape related variables	USER	+	User-Status: 0 non-user, 1 = user	0.67
	PLAVI	+	Planned visit within next 12 month: 1 = Yes; 2 = No, but wish to go once, 3 = No	2.46
	KNOW	+	Knowledge about ecological situation along the Elbe, scale from 1 to 8 scores dependent on correctly answered questions	3.64
	ANGLER	+	Fishes at Elbe, 1=Yes, 0 = No	0.03
	DIST	-	Distance in km to the Elbe	180.19
Attitudes	GEA	+	General Environmental Attitude (NEP scale from 8 to 40 scores)	31.26
	ATMEA	+	Attitudes toward management actions (scale from 2 to 14 scores)	8.33
	ATMON	+	Attitudes toward valuing nature in monetary terms; (1) strongly disagree to (5) strongly agree.	2.53
Past behaviour	PD	+	Past donation to environmental projects within last five years, 1 = Yes, 0 = No	0.19

A two-part regression model was used to analyse the WTP responses. It composes a logit model and an ordinary least square (OLS) regression model. The logit model separates those respondents who are willing to pay from those who are not. Subsequently, the OLS model indicates the relationship between the independent variables and the stated amount of money for those who were willing to pay.

[4] For a discussion of the expected influence of the independent variables on WTP see Carson et al. (2001) and Bateman et al. (2002).

Logit model: $\Pr(Y=1) = 1/1[1 + \exp\text{-}(ß_1 + ß_2X_2 + ß_3X_3 + \ldots]$, where Y=1 if respondent is willing to pay in principle and Y=0 otherwise

OLS regression model: $WTP = ß1 + ß_2X_2 + ß_3X_3 + \ldots + \varepsilon_i$ if Y=1, where ε_i is the usual disturbance term.

At the beginning, all variables reported in Table 1 were taken into each regression. Next, all variables not significant at the 5 per cent level were eliminated.

Table 2 shows the results of the two-part model.[5] In the upper half the results of the logistic regression are reported. It fits the data quite well. The percentage of correctly predicted values is 83 per cent and the value of McFaddens pseudo-R^2 is 0.26. The reported relationships are as expected, except for HHZ. The number of people per household has an unexpected positive influence on payment principle responses. One explanation might be that households with a larger number of people normally include children. As several other CVM studies have shown, children have a positive influence on the WTP (e.g. Kosz 1998). The two landscape-related predictors USER and ANGLER show a positive influence. The attitude toward the management actions (ATMEA) as well as the attitude toward valuing nature with money (ATMON) have the expected influence. In addition, people who had donated money to environmental projects within the last five years are more likely to pay for the management actions.

The OLS is, as was mentioned earlier, confined to those who were WTP in principle. The results are reported in the lower half of Table 2. As can be seen, the goodness-of-fit is significantly lower. The OLS regression is only able to explain 11 per cent of the variance of the stated willingness to pay. Furthermore, two variables show an influence on the stated amount of money that is in contrast to the expected influence. The distance to the Elbe has a positive influence, but the general environmental attitude (GEA) has a negative influence. In line with expectations are the positive influence of income, knowledge about the ecological situation and the attitude toward the management actions. The two-step approach reveals that the chosen set of potential explanatory variables can much better explain the decision to be willing to pay in principle than to explain why people stated a certain amount of money. However, the attitude toward the management actions shows the expected influence in both regressions.

Finally, individual WTP is aggregated. CV secures estimates of the WTP for the investigated wetland service of the individuals sampled. In order to obtain the total WTP for the management actions, the individual values have to be aggregated for the respective population. We therefore calculated the number of households living in the three catchment areas using a GIS. Altogether this adds up to 29.1 Mio households in all three catchment areas. Next, mean WTP was adjusted. Following the ap-

[5] In order to avoid losing too much data due to missing values, the missing values for INCOME were imputed using the STATA command “impute”.

proach by Schulze et al. (1998), we corrected mean WTP for the embedding effect. Moreover, the upper 2.5% of the stated WTP amounts were excluded in order to limit the influence of outliers. This resulted in an adjusted mean WTP of 5.3 €. Multiplying this figure with the number of households of all three catchment areas amounts to a total willingness to pay of € 153 Mio. As this figure comprises both one-time payments as well as annual payments, a second figure was calculated using only those respondents who were willing to pay on a yearly basis. Accordingly, willingness to pay decreased to € 108 Mio. from the second year on.

Table 2: Results of the two-part model

(1) Logistic regression (wtp(yes/no)), n = 1.278

variables	*coef.*	*odds ratio*	*s.e.*	*z-value*	
CONSTANT	-6.799			-15.30	***
HHZ	0.279	1.322	0.073	3.80	***
USER	0.690	1.994	0.193	3.58	***
ANGLER	1.082	2.951	0.443	2.45	*
ATMEA	0.271	1.311	0.030	8.88	***
ATMON	0.580	1.786	0.067	8.65	***
PD	1.215	3.371	0.182	6.66	***

X^2 (6) = 341.39, p<0.001, McFad-R^2 = 0.26, adj McFad-R^2

(2) OLS regression (wtp), n = 278

variables	*coef.*	*s.e.*	*t-value*	
CONSTANT	58.913	44.702	1.32	
INCOME	.011	.005	2.33	*
KNOW	6.929	3.357	2.06	*
DIST	.101	.0402	2.52	*
GEA	-4.375	1.439	-3.04	**
ATMEA	8.278	2.200	3.76	***

F(5, 272) = 6.41 ***, R^2 = 0,11; adj. R^2 = 0,10

*p < 0.05, **p < 0.01, ***p<0.001

5.2. Benefits due to increased nutrient retention

Based on the site-specific data and available models, the additional flooded area during an inundation event was determined for Sandau and Rogätz. The estimation of these changes includes information about the annual discharge (m^3/s) and the average inundation duration (d/y). The results show that the total available surface area (approx. 1,800 hectares) as a result of dike relocation would be only inundated with a

HQ 100[6]. For the mean annual flood discharge, the retention area for Sandau would be enlarged by 737 hectares, for Rogätz the effects are less (171 hectare) as a result of adverse site conditions. Accordingly, the area available for denitrification and the nitrogen retention effects depend on the specific discharge and morphological structure of the flood plains. Using the model from Behrendt and Opitz (2000) described above, the nitrogen retention rate is determined according to the increased hydraulic load as a result of structural changes in the regarded areas.

Figure 4: Annual nitrogen retention

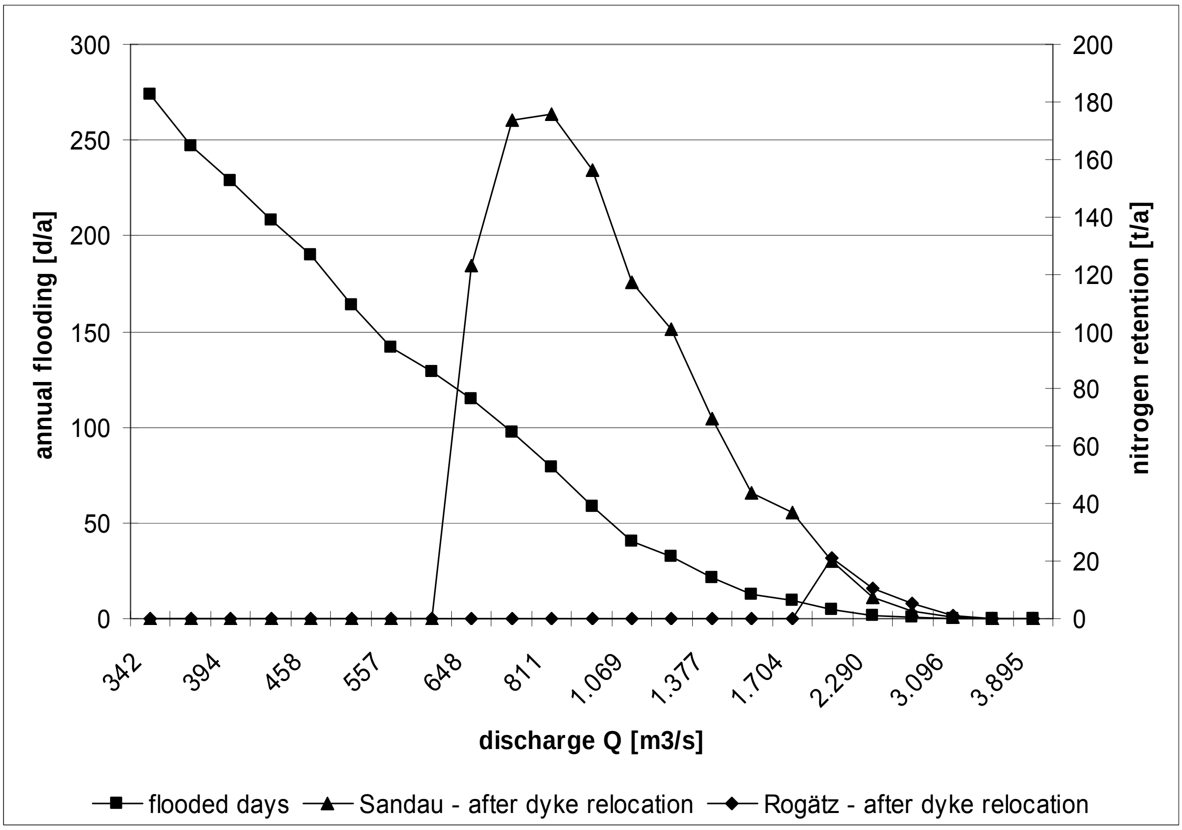


Figure 4 shows the results for the nitrogen retention depending on the annual inundation days and the discharge for Sandau and Rogätz. Following these results, the total effects for the additional retention area in Sandau adds up to a total nitrogen retention of 650 t/a, for Rogätz, 40 t/a. The comparably high retention effects in Sandau indicate an appropriate morphological structure of the flood plain which results in a higher inundation duration over the year.

Because of the high variability of the retention effects dependent on site-specific conditions, it is difficult to scale up this result for the investigated area of 15,000 hectares in total. On the basis of different denitrification rates for wetlands found in the literature (e.g. Mitch & Gosselink 2000, Kronvang et al. 1999, Jansson et al. 1994) and considering appropriate as well as adverse site conditions (i.e. the actual flooded

[6] Flood discharge with a probability of 100 years.

area), potential retention rates range from 50 to 450 kg N/ha. Therefore, for the two dike relocation sites next to Sandau and Rogätz, a mean retention effect of 200 kg N/ha is assumed.

Next, two substitutes for the service ‘improvement of water quality’ are considered: the building of sewage treatment plants and measures to reduce nitrogen emissions. Although both of these alternatives are based on different functional processes in detail and refer to different sources of nutrient load compared to the self-purification in flood plains, the final effect is the same. On a closer look, the denitrification process in sewage treatment plants is quite similar to the denitrification in flooded areas and is therefore the technical substitute most commonly used in the case of nutrient retention (Byström 2000; Gren 1995a, b; Bräuer 2002). On the other hand, measures to avoid non-point source nutrient loads can be taken into account. Since the main source of nitrogen pollution in aquatic ecosystems are emissions from agriculture, political strategies which aim, for example, at reducing fertiliser input could be a cost efficient alternative in order to improve water quality.

Hence, for the valuation of the ecosystem service following marginal costs considering the different scenarios described above are assumed:

(a) the marginal costs of waste water treatment in sewage treatment plants: mean 7.7 € / kg N (ranges from 5-8 € / kg N, see Grünebaum 1993)
(b) the marginal costs of avoidance of nitrogen loads by agricultural measures: mean 2.5 € / kg N (with a wide range depending on measures and production systems considered, see Dehnhardt 2002, Bräuer 2002).

Table 3: Value of the nitrogen retention of the restored flood plains (2000)

Scenario		*Sandau*	*Rogätz*	***Other sites***
Retention area (totally 15,000 ha.)	*[ha]*	*830*	*860*	*13,310*
Sewage treatment plant	[1000 €]	5,136	293	20,497
Agricultural measures	[1000 €]	1,734	99	6,921
Results per hectare				
(a) Sewage treatment plant	[€/ha]	6,188	340	1,540
(b) Agricultural measures	[€/ha]	2,089	115	529

The monetary value of restored flood plains is estimated by assessing the replacement costs to provide the same service in kg N retention. The results are given for both the project areas analysed in detail and the remaining area.

As Table 3 shows, the results differ considerably according to the site conditions and the scenario considered. The replacement value varies between approx. € 100,000 and € 290,000 for adverse site conditions (Rogätz) and € 1,7 mill and € 5,1 mill for appropriate conditions (Sandau). For the total amount of 15,000 ha additional flood plain area, the value ranges from 6.9 to 20.5 mill. €. These values are considered as annual benefits concerning the indirect use value of restored flood plains within the benefit-cost analysis.

5.3. Total economic value (TEV)

In this study only the value of two ecosystem services, maintenance of endangered species and nutrient retention, was determined. Accordingly, this value does not reflect the total economic value of the management actions. There are several services, such as flood protection or recreation, which may be affected by dike shifting, but were not evaluated in this study. Hence, the TEV of the management actions is expected to be larger than the sum of both examined services. This has to be kept in mind when the benefit-cost-analysis is conducted. Table 4 reports the benefits of the two estimated services that amount to the TEV. In the column labelled 'low' the benefits from the CVM were corrected for both the embedding effect as well as for outliers. As a yardstick to determine the benefits from nutrient retention, agricultural measures to avoid nitrogen loads were used. In the column labelled 'high' the benefits from the CVM were corrected only for the embedding effect. As a yardstick to determine the benefits from nutrient retention, the costs of wastewater treatment in sewage treatment plants were used.

Table 4: Total economic value of management actions in million €

Wetland service	low	high
Endangered species and habitats	153.0	252.0
Nutrient retention	8.7	26.0
Total	161.7	278.0

A problem that may occur in the summing up of both values is double counting. Turner et al. (2000), with reference to Barbier (1994), point out that if the nutrient retention function is integral to the maintenance of biodiversity, then if both functions are valued separately and aggregated, this would double count the nutrient retention already captured in the biodiversity value. In the description of the management actions, interviewees were only informed about the effect the increased area of riparian flood plains would have on the protection of endangered species and habitats. Therefore their WTP can be interpreted as a benefit arising only from the enhanced protection of these species and habitats. On the other hand, in some manner nutrient retention is integral to the maintenance of biodiversity, because regaining flood plains will provide both services at once. In order to be at least aware of the effect of some form of double counting on the benefit-cost analysis, we treated both services separately in the sensitivity analysis. This will show whether the benefits of both services are crucial to pass the benefit-cost test.

5.4. Benefit-cost and sensitivity analysis (BCA)

To carry out the BCA, several scenarios of cost and regained flood plain area combinations were created in order to cover the range of possible developments. For example, the size of the regained flood plain area was varied (10,000 ha / 15,000 ha). Costs for dike shifting or land use changes were also varied. This resulted in eight different scenarios (Table 5).

Based on the results of other research projects of the Elbe-Ecology program (Dehnhardt & Meyerhoff, 2002), we were able to determine the costs of the management actions. In detail, it was assumed that the costs for dike construction vary between 1.68 Mio. € and 1.86 Mio. € per kilometre dike. In the case that 10,000 ha flood plains are to be regained, 83 kilometres of new dikes would have to be rebuilt. In the case that 15,000 ha are to be regained, 136 kilometres of new dikes would have to be rebuilt. Furthermore, it was assumed that costs of the intended land-use change on the new flood plains will vary between 1,012 €/ha and 2,945 €/ha uniquely and between 56 €/ha and 75 €/ha annually. Finally, the costs arising from land-use changes on the already existing flood plains was set at 326 €/ha as the low amount and at 506 €/ha as the high amount.

Table 5: Scenarios underlying benefit-cost analysis and results

	Scenarios							
	I	II	III	IV	V	VI	VII	VIII
10,000 ha new flood plains	X		X		X		X	
15,000 ha new flood plains		X		X		X		X
High costs dike removal	X	X	X	X				
Low costs dike removal					X	X	X	X
High costs land-use change	X	X					X	X
Low costs land-use change			X	X	X	X		
Net present value (in Mio. €)	955	854	1,062	972	1,074	986	967	876
Benefit-cost ratio	3.1	2.5	4.1	3.2	4.1	3.3	3.2	2.6

In addition, several assumptions have been applied to all scenarios. The project lifespan was set at 20 years and the discount rate at 3 per cent. Cost for both land acquisition and dike shifting already occur in the first year. On the other hand, the benefits will not arise before the construction work is completed. The time period needed for dike shifting was set at three years. As it was reported in Section 5.3, we differentiated between the WTP in the first period after construction is completed (the fourth year of the project lifespan) and the WTP in the following years. The reason for this is that several respondents were only WTP once. Therefore, their WTP was only recognised in the first year after the construction period ended. The WTP of all other respondents is incorporated for all years between the fourth and twentieth year of the investigated lifespan.

The next to last row of Table 5 shows the net present value of the different scenarios. It varies between € 854 Mio. and € 1,074 Mio. As the last line of the table indicates, the benefit-cost ratio is for all scenarios. Even when 15,000 ha new flood plains are regained and costs for dike relocations as well as for land use changes are high, the net present value is still positive. Furthermore, the results show that even the scenario with the highest costs and the lowest benefits results in a benefit-cost ratio of 2.5 : 1. In contrast, the scenario with 10,000 ha flood plains regained and low cost for dike relocation as well as land use change shows the highest benefit-cost ratio (4.2 : 1).

As deficiencies keep the results of the ecological as well as economic analysis from being predicted with certainty, sensitivity analysis helps to illustrate the uncertainties arising from these deficiencies (Merrifield, 1997). We therefore carried out a sensitivity analysis for those two scenarios which showed the lowest BCR for a 10,000 ha regain of flood plains (scenario I) and for a 15,000 ha regain of flood plains (scenario II). The analysis proceeded in two steps. In the first step, both benefit measures as well as the costs for dike shifting and land-use change on the already existing flood plains were each doubled and halved, leaving all other factors constant. Both measures can change strongly during the project lifespan. For example, intervening factors such as a decrease in economic development may influence people's principle willingness to pay as well as the amount of money they are willing to pay. Moreover, it is well known that the costs for this kind of infrastructure are almost always higher in the end than originally estimated. We also varied the discount rate and extended the time period needed to complete the construction work by two years.

As can be seen in Table 6, the benefit-cost ratio remains in all cases above 1:1.0. One Euro invested always results in benefits higher than one Euro. Even when the benefits of biodiversity conservation are halved, the benefit-cost ratio is still 1:1.5 for scenario I and 1:1.3 for scenario II. Compared to the benefits from biodiversity conservation, the benefits from the service nutrient retention have only a minor influence on the BCR. In both scenarios the BCR decreases by 0.5 when the benefits are halved. On the other hand, the costs due to land-use changes on already existing flood plains have a larger influence on the BCR than the cost due to dike shifting. If the former are doubled, the BCR decreases to 2.0 : 1 for scenario I and to 1.7 : 1 for scenario II. Moreover, the last three rows of Table 6 show that, all other things being constant, neither a higher discount rate nor an extended construction period cause a huge decrease of the BCR.

To conclude, the sensitivity analysis indicates that the results of the BCA are stable. Even for those scenarios of the management actions with the lowest BCR, the assumed changes of important factors do not lead to a BCR beneath one.

Table 6: Results of the sensitivity analysis

		Scenario I old BCR: 3,1 10.000 ha		Scenario II old BCR: 2,5 15.000 ha	
Changed benefit and cost components		halved	doubled	halved	doubled
Benefits due to maintained biodiversity		1.5	6.2	1.3	4.9
Benefits due to nutrient retention		3.1	3.2	2.5	2.6
Cost due to dike shifting/construction		3.8	2.4	3.2	1.8
Costs due to land-use change on already existing flood plains		4.3	2.0	3.3	1.7
Changed discount rate (originally 3%)	-> 1%	3.3		2,7	
	-> 5%	2.9		2,2	
Extended construction period (5 years)		2,7		2,2	

6. Conclusion and policy implications

By creating the WFD, the EU has entered a new phase in water policy. The Directive is an integrated approach to river basin management in Europe that aims at good water status for both surface water and groundwater. However, the WFD does yet not clearly state the extent to which wetlands should be used for the achievement of environmental objectives. Furthermore, the scope of the economic analysis that is intended to play a key role in river basin management is not clearly defined. At the present time, for example, whether or not benefits arising from good water quality should be assessed is under discussion.

It is widely accepted that, from an ecological point of view, riparian wetlands as an integral part of the water environment play an important role for the environmental quality of river systems. In this context, the objective of the present study was to show that the non-market benefits of a good water quality environment can be substantial. The results reveal considerable economic value of wetland restoration and biodiversity protection along the river Elbe. Accordingly, the management actions to improve the services provided by wetlands would easily pass a benefit-cost-analysis. The sensitivity analysis also shows that the results of the benefit-cost-analysis are stable even if the estimated certain benefits are halved or the cost are doubled.

Based on these results we therefore argue that both the importance of wetlands for good water quality as well as the assessment of non-market benefits of a good water environment should be integrated into the WFD. Accordingly, during the implementation process the WFD should not only be extended in order to explicitly recognise wetlands, but the scope of the economic analysis should also be extended in order to determine the benefits of management actions. The exclusion of the non-market benefits from river basin management could be a missed opportunity to promote a sustainable development of European rivers. In order to integrate the benefit assessment into the WFD and its economic analysis, several things have to be clarified. For example, there is no guidance on the choice of appropriate valuation methods that can be used by the implementing authorities in order to incorporate the benefits of a good water quality environment into the decision-making process. Two approaches for benefit assessment were presented in this paper. Indirect methods, such as the replacement cost approach, might be suitable for indirect use values. If the relationships between ecological and economic systems are well known and an appropriate technical substitute can be identified, the benefits of ecosystem services will be comparatively simple to estimate. The CVM is especially relevant when non-use values form a large part of the total economic value. Faced with the situation of wetlands in Europe, it is reasonable to expect a significant non-use value of wetlands. Furthermore, the CVM allows valuation of future states of the environment. This as well is an important property for river basin management. However, the use of the CVM is often very expensive and its use will therefore be limited.

Hence, the benefit transfer might become an important tool within the WFD. The basic idea of this approach is to assign economic values to ecosystem services by transferring results from a valuation site to a policy site (Brouwer, 2000; Dehnhardt, 2004). For implementing the WFD, the benefit transfer approach seems to be an interesting method because the authorities usually do not have a large budget for site-specific studies. However, the suitability of this approach has been debated in recent years and there is no consensus at the moment as to whether it delivers valid numbers or not. Furthermore, as we already mentioned in the introduction, only a few studies about the economic values of riparian wetlands exist as yet. The successful use of the benefit transfer in the WFD will not only depend on methodical progress but also on a larger number of primary valuation studies. Accordingly, the Elbe study might serve as another step toward compiling a database for transferring values.

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