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Assessing the cost-effectiveness of the water purification function of wetlands for environmental planning

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ABSTRACT

A reduction of nutrient input into freshwater and marine water bodies is a prerequisite for the restoration of healthy aquatic ecosystems and for the achievement of the ambitious goals of the European Water Framework Directive and the HELCOM and OSPAR convention. In Northern Germany, phosphorus inputs from point sources were reduced by nearly two-thirds compared to the phosphorus load in the period 1980–1985. This success was made possible by the investment of 2.1 billion € since 1985 in the development and enhancement of waste water treatment plants. However, these investments had only little effect on nitrogen inputs, which enter water bodies mainly from non-point sources. In 2002, the federal state Schleswig-Holstein implemented a peatland rehabilitation program aiming to restore the water purification function of this wetland type. The rationale of this plan is based on an economical analysis of the effects of the investments in the waste water action plan. A reduction of the P load by 1 kg has mean costs between 70 and 100 €. Based on model calculation, restoring wetlands for water quality purification will cost between 1 and 50 € per retained kg of nitrogen when considering a 10-year time span. Compared to the waste water action plan, wetland restoration is also a cost-efficient strategy for nutrient load reduction. For 10% of the investments of the waste water action plan, the nitrogen load from Schleswig-Holstein can be reduced by 10%; this will require rewetting of 20,000 ha of wetlands. The implementation of this strategy as well as the application of the concept of ecosystem services in environmental management is currently limited by a lack of political support and funding for such projects from higher political bodies such as the European Union.

1. Introduction

Providing society with clean and healthy water belongs to the challenges for the next decades to meet the demands both of humans and ecosystems (Millennium Ecosystem Assessment, 2005). Actions to improve water quality can be taken during all stages of water flow through the landscape. Cost-efficient strategies for water quality management aim on a reduction of quantitatively important inputs of harmful substances as well as on enhancing the transformation and retention capacity in the landscape (Kronvang et al., 2005; Trepel and Palmeri, 2002). For nutrients, quantitatively important inflows to the water system originate from diffuse sources and point sources. On the other side lakes, rivers and wetlands are landscape elements with a high nutrient transformation and retention potential (Verhoeven and Meuleman, 1999; Fisher and Acreman, 2004). Quantifying the services these ecosystems provide for society is gaining more attention in environmental decision making (Turner et al., 2008).

The pressure to improve water quality of groundwater and surface water bodies has increased in Europe since the Europe Water Framework Directive (WFD) came into force in December 2000. The ecological aims of the WFD can only be achieved if the nutrient loads to surface and coastal water bodies are reduced significantly. To achieve this environmental goal policy has to develop management strategies for nutrient load reduction for different sectors (Turner et al., 1999). For each sector measures have to be developed, their effect either on reduction of nutrient inputs or on improved retention assessed and their costs calculated. These data and information are needed by policy and decision makers to implement different cost-effective water management strategies for nutrient load reduction.

For this purpose in this paper, the effects of improved waste water treatment on nutrient load reduction are compared with effects of wetland restoration on improved nitrogen retention. The cost-effectiveness is used as an ecological–economical indicator to compare and evaluate both strategies. It is calculated for both strategies independently due to different data availability with two different approaches. These ecological–economical data are needed to apply and improve the concept of ecosystem services

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in environmental planning and to make it a useful tool for responsible management of natural resources.

This paper especially introduces a new method for the model-based quantification of ecosystem services provided by wetlands for water purification and improves understanding how human activities can control these services. Furthermore it offers examples how scientific knowledge on water and nutrient exchange in wetlands can be transformed for communication to decision makers in environmental planning.

2. Methods

This paper evaluates the cost-effectiveness of two water management strategies for nutrient load reduction in the state of Schleswig-Holstein, Germany. Cost-effectiveness (CE: € $kg^{-1}\,t^{-1}$) is defined as the costs (C: €) required achieving a load reduction (Δ_{LR}: kg) of 1 kg during a given time period (t); it is calculated with Eq. (1):

$$CE = \Delta_{LR}/C \tag{1}$$

The state of Schleswig-Holstein is located in northwest Germany between the city of Hamburg and Denmark. The land of Schleswig-Holstein drains with approximately equal proportions into the Baltic Sea, the North Sea basin, and the river Elbe, which mounds through the German bight also into the North Sea. Due to its position between the North Sea and the Baltic Sea, strategies for nutrient load reductions are since several decades an important part of the state's environmental policy (MUNL, 2005). During the 1980s action plans were implemented to improve waste water treatment of the population as well as the industry. These early plans were triggered by the OSPAR convention for protecting the North Sea and the HELCOM convention for protecting the Baltic Sea. In 2002, these plans were accompanied with a peatland action plan aiming on restoring peatlands for improving nutrient retention (Trepel, 2007). Peatlands are the dominant wetland type in Schleswig-Holstein, covering approximately 9.3% of the lands surface.

2.1. Assessing the effectiveness of the waste water action plan

The effect of the waste water action plan on nutrient load reduction is quantified with water quality monitoring data. Fig. 1 displays the change of the nitrogen and phosphorus load from Schleswig-Holstein to the Baltic Sea and North Sea for three 10 year time periods. The phosphorus load decreased exponentially by more than half during the last 30 years in both areas. The nitrogen load decreased linear by around 25% in this time period only.

The different behaviour in load reduction of both nutrients can be explained by two factors. First, the waste water action plan mainly funded measures to improve phosphorus retention in treatment plants. Second, point sources are of lower importance for nitrogen input into surface water systems than non-point sources.

For a quantification of the cost-effectiveness of the waste water action plan, the cumulative load reduction for nitrogen and phosphorus is quantified using transformed flow-weighted loads (Grimvall et al., 2000; Stalnacke et al., 1999) and a linear trend function for nitrogen and an exponential trend function for phosphorus. In the period 1985–2005 the cumulative nitrogen load reduction to the Baltic Sea was approximately 14,400 t and to the North Sea about 9700 t (Table 1). The cumulative phosphorus load reduction between 1985 and 2005 to the Baltic Sea was around 6400 t and to the North Sea 2900 t (Table 1).

According to a ministry report (MUNL, 2005), between 1985 and 2005 around 2.1 billion € were spent for improving waste water treatment. With the assumption that this money was spent proportional to the population of the three main drainage basins of Schleswig-Holstein, around 931 million € were spent in the Baltic Sea Basin and 310 million € in the North Sea basin (Table 1).

With these data for cumulative load reduction and their costs, the cost-effectiveness of the waste water action plan is quantified. Because the measures implemented affected the nitrogen and phosphorus load from point source differently, the cost-effectiveness is calculated for three cost allocation strategies. The first strategy, assumes that the costs were used either for nitrogen or phosphorus reduction only. The cost-effectiveness values calculated with this assumption represent the upper boundary of costs. The second strategy assumes that 67% of the costs were spent on phosphorus reduction and 33% on nitrogen reduction. This strategy is realistic, because the measures funded by the waste water action plan were designed mainly for phosphorus removal and thus had lower effect on reduction of nitrogen loads. The third strategy assumes that costs were used equally for reduction of nitrogen and phosphorus loads from point sources. This strategy presents the lower boundary of costs.

2.2. Quantifying nitrogen retention in wetlands

At the end of the 20th century it became clear in Germany that further reduction of nutrient loads to the sea is only possible if either inputs from non-point source are reduced or the retention

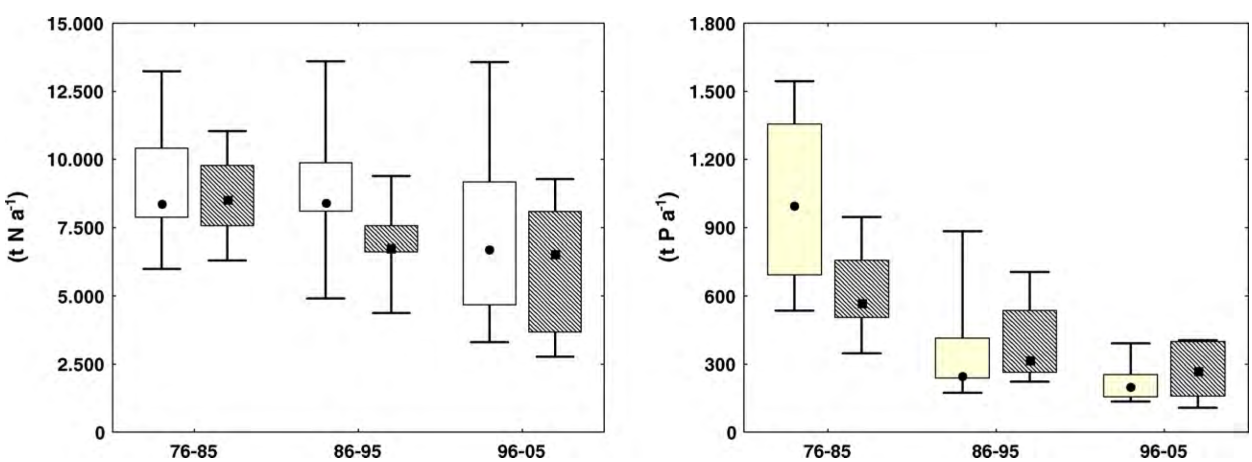

Fig. 1. Change of the nitrogen (left) and phosphorus (right) load from Schleswig-Holstein to the Baltic Sea (circle) and North Sea (squares) between 1976 and 2005. Box plots show median, upper and lower quartiles and minimum and maximum values of flow-weighted nutrient loads.

Table 1
Cost-effectiveness of the Schleswig-Holstein waste water action plan.

Parameter	Unit	Baltic Sea	North Sea
Area	km^2	5446	4221
Population	n	1,200,000	400,000
Cumulative Nitrogen reduction between 1985 and 2005	t	14,417	9773
Cumulative Phosphorus reduction between 1985 and 2005	t	6393	2918
Costs spent in waste water action plan between 1985 and 2005	Mio. €	931.6	310.6
Cost-effectiveness			
Investment only for N reduction	€ kg^{-1} N	65	32
Investment only for P reduction	€ kg^{-1} P	146	106
Investment 33% for N reduction	€ kg^{-1} N	21	10
Investment 67% for P reduction	€ kg^{-1} P	98	71
Investment 50% for N reduction	€ kg^{-1} N	32	16
Investment 50% for P reduction	€ kg^{-1} P	73	53

capacity in the landscape is increased (BMU, 2004). At this time, around 94% of the population of Schleswig-Holstein was connected to treatment plants. Thus the state developed a peatland action plan with the goal to rehabilitate the water purification function provided by this wetland type (Trepel, 2007). Peatlands influence the water quality of their basin by the way they are hydrologically connected with their surrounding. For quantification of the effect of different water management and land use scenarios on nitrogen retention, the web-based, flow-path-oriented decision support system (DSS) WETTRANS was developed (Trepel and Kluge, 2004).

2.2.1. WETTRANS model concept

The DSS WETTRANS follows a steady-state, flow-path oriented, mass budget approach and assumes homogenic stratigraphical condition in the peatland under question. A detailed model description is given by Trepel and Kluge (2004). According to a flow-path-oriented approach, peatlands receive water and nutrient input via several hydrological inflow pathways. The inflow pathways differ in their age and due to different transient times and origin in their nutrient concentrations (Table 2). Especially, younger, lateral inflow pathways show a high variability of nutrient concentrations due to differences in soil texture and land use (Dahl et al., 2007). The lateral water entering a peatland is a mixture of these different lateral inflow pathways (Table 2).

Table 2
Definition and characteristics of inflow and outflow pathways to and from a peatland; NTP°: nitrogen transformation potential.

Inflow pathways		Age	N ($mg\,L^{-1}$)
A	Precipitation	<hours to days	1.0–2.0
B	Surface runoff	<hours to day	2.0–15.0
C	Interflow	<30 days	5.0–30.0
D	Tile drainage	<30 days	5.0–30.0
E	Young aerobic groundwater	<20 years	1.5–20.0
F	Young anaerobic groundwater	>20 years	0.0–10.0
G	Old anaerobic groundwater	>100 years	0.0–1.0
H	River inflow	Mixed water of different age	2.0–7.0

Outflow pathways		Hydraulic detention time	NTP°
1	Evapotranspiration	–	–
2	Ditch outflow	<10 days	Medium
3	Drain outflow	<days	Low
4	Surface flow	<hours to days	Medium
5	Subsurface flow	<30 days to years	High
6	River throughflow	<10 days	Low
7	Groundwater bypass	<1 year	High

In the WETTRANS model, the peatland is considered as a system which distributes the inflow from different inflow pathways to outflow pathways. The outflow pathways are characterized by different hydrological retention times, oxygen status and carbon availability (Trepel and Kluge, 2004). Thus, they can be connected with different nitrogen transformation potentials (Table 2). In the current model version, each outflow pathway has a potential nitrogen transformation coefficient which is modified by the outflow path specific flow length and a site-specific land use intensity index.

2.2.2. Input data

The input data for the WETTRANS DSS are grouped in five categories. The first category contains basic information about peatland size, mean peatland width, peatland length, precipitation and evapotranspiration, and the size of the upstream basin and lateral basin. In the second category, the user has to choose a substrate type (peat type) and a depth for three layers. From this information, a peatland type is calculated based on transmissivity differences between (I) layer 1 and layer 2 and (II) layer 2 and layer 3. The third category includes information about the hydrological conditions in the lateral drainage basin adjacent to the peatland. The user is asked for data about, e.g. the proportion of adjacent slopes in the lateral basin, proportion of drained slopes, building area of young aerobic groundwater, building area of anaerobic ground water, and additional inflow of deep groundwater from a second aquifer. The data from these first three categories are used to calculate the water and nitrogen inflow for 8 inflow pathways. For the calculation of the nitrogen inflow, the programme suggests for each inflow pathways a mean nitrogen concentration which is based on a statistical analysis of available water quality data from the area (Trepel and Kluge, 2004).

In the fourth category, the user has to provide information about the water management inside the peatland. Data include proportion of drains on peatland perimeter, proportion of ditch drainage and tile drainage in the peatland, mean ditch depth, mean tile depth, mean annually flooded area and mean annual flood duration. These data are used to calculate together with information about the peatland stratigraphy the water exchange from the surrounding through the peatland to a receiving surface water body. Finally in a fifths category, a user has to provide data about the proportion of different land use types on the peatland. Each land use type is assigned with (changeable) fertilizer amount. Land use data are needed for the calculation of total nitrogen input and output by harvest.

2.2.3. Model results

For each simulation a water partitioning matrix is calculated, which distributes the inflowing water and nutrients from the inflow pathways to the outflow pathways. Fig. 2 visualizes the development of the numbers in the water partitioning matrix for a percolation peatland with a slightly decomposed peat layer above a thick, impermeable layer during increasing drainage intensity. When this peatland is not drained, the inflowing water leaves the peatland via surface and subsurface flow. Under poorly drained conditions, both the proportions of surface and the subsurface flow decrease while the proportion of ditch outflow increases. An excessively drained peatland is dominated by ditch and tube drainage outflow. Lowering of the river sole has resulted also in a reduced water exchange between the river inflow and the peatland.

Fig. 2 shows clearly the model concept. The water partitioning matrix presents the structure of the system and contains information of the peatland stratigraphy and water management. These values are newly calculated for each modelled site; thus the model approach accounts for the uniqueness of individual wetland (Joosten, 1993) regarding its connectivity and surrounding.

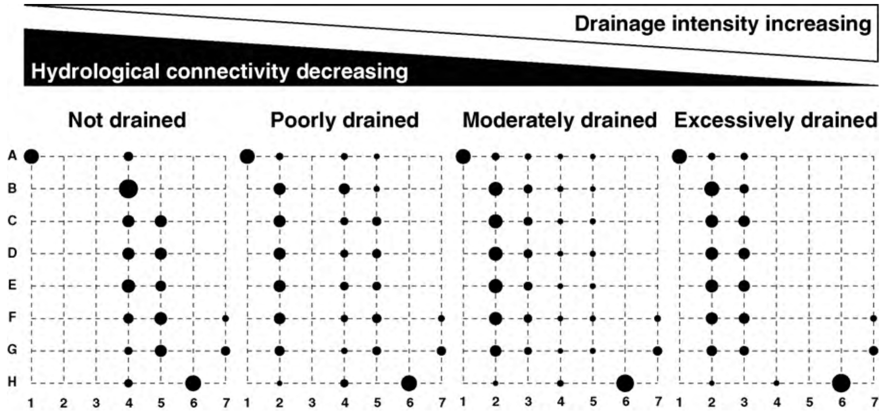

Fig. 2. Changes of the water exchange pattern of a percolation peatland with increasing drainage intensity. For explanation of inflow pathways (letters) and outflow pathways (numbers) see Table 1.

Next to the water partitioning matrix a nitrogen transformation matrix is calculated (Trepel and Kluge, 2004). The final model results are values for nitrogen input to the system, nitrogen output from the system, nitrogen retention in the system and nitrogen retention efficiency of the system. Additionally, nitrogen retention and retention efficiency is calculated for three subsystems: the vertical subsystem represents effects of water management, land use and mineralization on nitrogen outflow and harvest, the lateral subsystems represents effects of water management on nitrogen retention from lateral inflow pathways, and the longitudinal subsystem represents nitrogen retention via flooding.

2.2.4. *Effect of water management and land use on nitrogen outflow*

Fig. 3 summarizes the effect of water management and land use on nitrogen outflow from a percolation peatland. Increasing drainage intensity is often combined with increased land use intensity. Both aspects affect the nitrogen outflow from the whole system as well as nitrogen outflow from the three subsystems. The difference between the four system states can be best visualized with an amoeba diagram, where all values of the potential natural status are transformed to 1 and changes of the parameters from the other scenarios are calculated against this value.

Flooding index (FI) and vertical nitrogen input (VI) are indicators for the system itself. Under natural conditions, the system has a flooding index of 0.25 and receives a vertical input of 4.6 t N year^{-1} due to mineralization and atmospheric deposition. The Flooding index decreases first when the system is moderately drained, the vertical input increases immediately when the system is drained and fertilized. Nitrogen outflow (NO) and nitrogen retention (NR) are indicators for the reaction of the whole system. Nitrogen outflow increases slower as nitrogen retention decreases. This reaction is caused by the basin position of the peatland. In this example, a major nitrogen inflow is coming from the upstream basin of the peatland. Nitrogen retention of the system decreases considerably with increasing drainage intensity and land use intensity. Lateral retention (LAR) and longitudinal retention (LOR) are indicators for the reaction of these subsystems on water management and land use changes. The amount of lateral nitrogen retention decreases when the system is drained by ditches and drains. These outflow pathways have lower nitrogen transformation potentials as the natural outflow pathways overland flow and subsurface flow (see Table 2 and Fig. 2).

The amoeba diagram indicates how far the drained system has moved away from the pristine state.

2.2.5. *Cost-effectiveness of peatland rehabilitation for water purification*

The cost-effectiveness of planned peatland rehabilitation projects is calculated from the difference in nitrogen output obtained from a scenario analysis where a present situation is compared with a future scenario. The project costs are calculated from the costs required to purchase the peatland area plus 15% of project management costs. The value of additionally 15% for project management costs is based on regional experiences with ongoing wetland restoration projects (Kieckbusch et al., 2006; Kieckbusch and Schrautzer, 2007). The additional 15% cover costs for hydrological plannnig and consultancy as well as the rewetting measures itself. The costs do not cover any wetland management costs because most rewetted wetlands are either not used for agriculture anymore or are used with moderate grazing scheme.

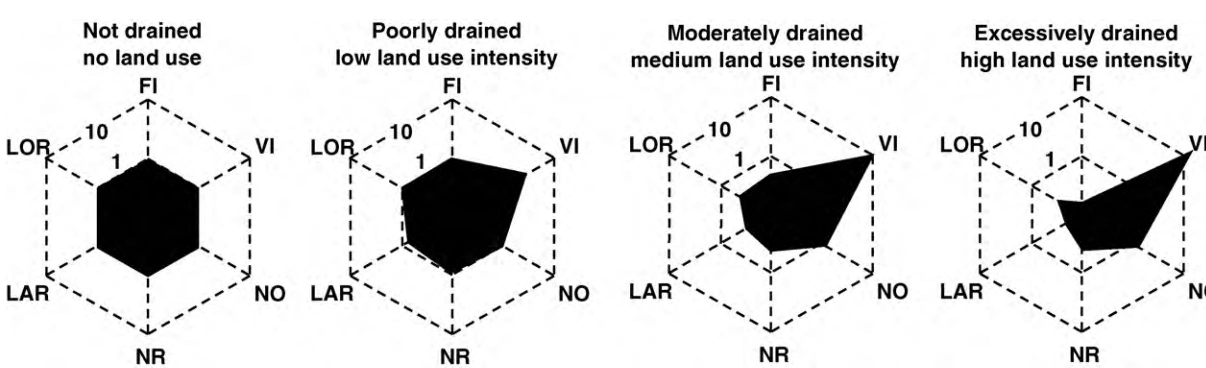

Fig. 3. Changes of nitrogen outflow from a percolation peatland with increasing drainage intensity. Note the scale of the amoeba diagram is logarithmic. Changes are calculated against the most natural state (not drained and no land use). FI = flooding index; VI = vertical nitrogen input (mineralization, fertilizer and deposition); NO = total nitrogen output from the system; NR = total nitrogen retention in the system; LAR = lateral nitrogen retention; LOR = longitudinal (flooding) nitrogen retention.

3. Results

3.1. Cost-effectiveness of the waste water action plan

The nutrient concentration in surface waters of Schleswig-Holstein has been decreased during the last 3 decades. Consequently, nitrogen and phosphorus loads from land to sea were reduced. In Schleswig-Holstein these changes are mainly the result of the waste water action plan which came into force at the end of the 1980s. The costs for the reduction of 1 kg of nitrogen vary between 10 and 65 € and for the reduction of 1 kg of phosphorus between 71 and 146 € (Table 1). The high variation in the cost-effectiveness is caused by different hydromorphological constraints of the North Sea and Baltic Sea drainage basin in Schleswig-Holstein and by the three different cost allocation strategies. According to the realistic cost allocation strategy, a load reduction of 1 kg of nitrogen was achieved in the Baltic Sea Basin for 21 € and in the North Sea Basin for 10 €. A load reduction of 1 kg of phosphorus was achieved in the Baltic Sea Basin for 98 € and in the North Sea Basin for 71 €.

3.2. Cost-effectiveness of the peatland action plan

Peatland rehabilitation can contribute to the water quality improvement if the rehabilitation measures are designed to restore the hydrological connectivity between a peatland and its surrounding (compare Fig. 2). With the WETTRANS model nitrogen retention and their cost-effectiveness was calculated for 34 potential project sites in Schleswig-Holstein (Fig. 4). For each project water management and land use was changed in two scenarios.

According to these data, cost-effectiveness for a reduction of 1 kg nitrogen can vary between 11 and >500 €, if only the load reduction of the first year is considered (left *y*-axis of Fig. 4). However, because the peatland action plan aims to rehabilitate fully functioning ecosystems and does not aim to overexploit one function; it can be assumed that the effect achieved by rewetting and lowering of land use intensity will last longer. If the load reduction of a 10-year period is considered, the cost-effectiveness for a reduction of 1 kg of nitrogen varies between 1 and >50 € (right *y*-axis of Fig. 4). This variation is caused by the different present conditions of the wetlands, which determine their potential load reduction.

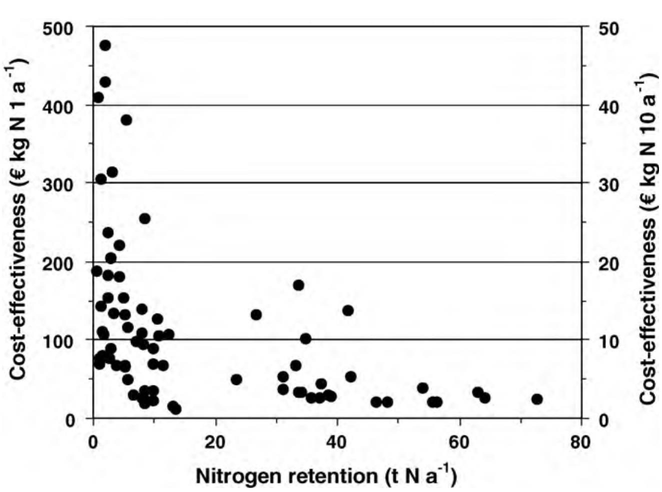

Fig. 4. Cost-effectiveness of wetland restoration projects for improving nitrogen retention calculated with WETTRANS. The left *y*-axis indicates the cost-effectiveness in the first year after project implementation, the right *y*-axis the cost-effectiveness after 10 years.

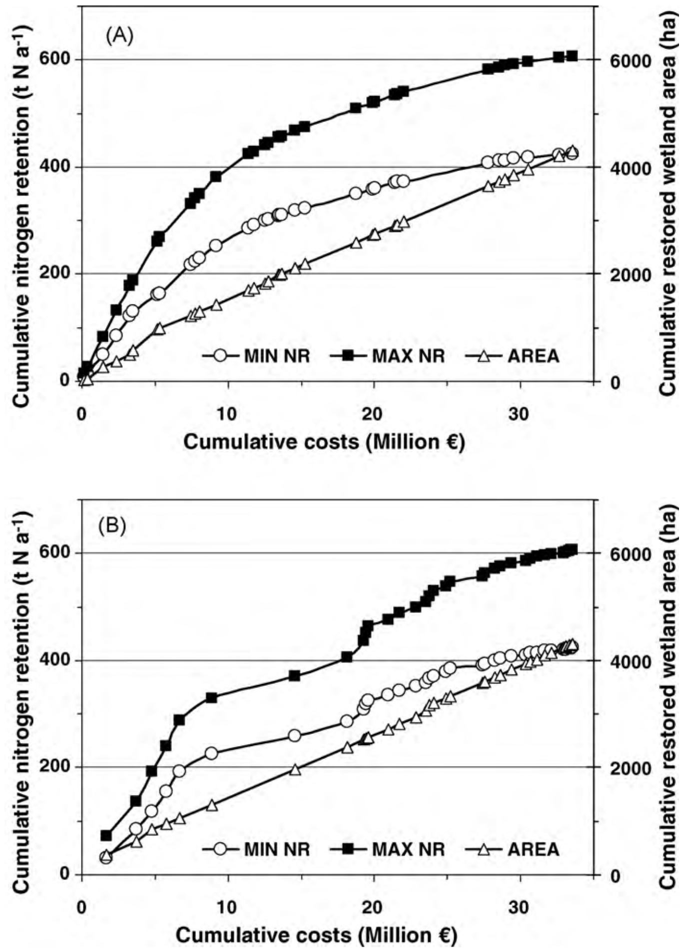

Fig. 5. Cumulative costs, cumulative annual nitrogen retention and required wetland area for two different ranking mechanisms. In 5A sites are ranked according to their cost-effectiveness, in 5B sites are ranked according to maximum nitrogen retention. Data are used for cost-effective site selection on regional scale.

The results from the WETTRANS model are used for ranking the potential wetland restoration projects and to decide which projects are funded first. In Fig. 5a, the projects are ranked after their cost-effectiveness. In this case, for 15 million € 20 projects can be funded. These projects require purchasing approximately 2100 ha of wetland area. They will reduce the nitrogen load by 320–450 t per year due to improved nitrogen retention. This reduction is equal to a load reduction of 1.2–1.7%. If the projects are ranked after maximum nitrogen retention (Fig. 5b), for 15 million 7 projects can be funded. These projects require purchasing approximately 2000 ha of wetland area. They will reduce the nitrogen load by 260–370 t per year due to improved nitrogen retention. This reduction is equal to a load reduction of 1.0–1.4%.

4. Discussion

Reducing eutrophication in surface and marine water bodies is a well accepted need in European policy to achieve the goals formulated by the Water Framework Directive. In the past, reduction of nutrient loads from surface waters to the Sea were achieved in several European countries mainly by improving waste water treatment. Based on data and experiences from Schleswig-Holstein, the nutrient load reduction in the waste water action plan could only be achieved because a large amount of money (2.1 billion €) was spent over a long time (more than 20 years) to improve waste water treatment. The technical measures had reduced mainly the phosphorus load. A reduction of 1 kg phosphorus has cost between 71 and 98 €. This value is at the upper boundary of the cost-effectiveness of wetland restoration projects aiming to improve nitrogen retention. If the cost-effectiveness of wetlands is considered for a 10-year period, then this value represents the maximum costs.

Mannino et al. (2008) report an average service cost improvement of 2.1–8 when seminatural wetlands are used for waste water treatment instead of traditional waste water treatment systems. They calculated that the development cost of free surface flow wetlands are six- to ninefold higher than traditional wetlands systems; however due to lower maintenance and service costs seminatural wetlands are economically competitive with technological treatment plants. The cost-effectiveness of wetland restoration for a 10-year period calculated with the WETTRANS model are in the same order of magnitude as costs calculated by Gren et al. (1997) for wetland restoration in different countries around the Baltic Sea. In most countries costs for the retention of 1 kg of nitrogen were around 1.5–5 €; however, in some northern countries costs increased up to 30 or 50 €. Mitsch and Gosselink (2000) argue that the cost-effectiveness of wetlands for water purification is largely influenced by the wetland position in the landscape and by the wetland type; both factors are considered in the WETTRANS model and allow the quantification of site-specific costs.

Using natural biogeochemical processes in wetlands for water purification requires at first a quantification of the possible effects and their costs (Verhoeven and Meuleman, 1999). Both factors can be determined with the model WETTRANS for nitrogen. This tool calculates site-specific retention rates and cost-effectiveness values. While many studies calculate retention rates with rule of thump values (Kronvang et al., 2004; De Groot, 1992), these studies may over- or underestimate the effect of an individual wetland for water purification. The WETTRANS model is used by the state environmental agency (I) for the selection of most effective wetlands sites and (II) for the selection of the most effective water management measure for nutrient load reduction.

Selection of sites and measures are crucial for an effective implementation of any environmental programme aiming to improve the environmental conditions.

Using ecosystem services as a guiding principle in environmental policy requires a policy shift towards an ecohydrological landscape perception. While ecological services receive increasing attention within the scientific community (De Groot, 1992; Maltby et al., 1994; Joosten and Clarke, 2002; Turner et al., 2008) and in international environmental policy (Millennium Ecosystem Assessment, 2005). Their consideration in the preparation of future management concepts for single wetlands is still in the beginning. This imbalance has several reasons. (I) Quantification of non-direct uses is often uncertain (Rosenberger et al., 2003). (II) The effect of a single wetland to regional water quality improvement appears to be quantitatively not relevant for local decision makers. (III) The political pressure for improving surface water quality is in most European countries minimal. All three reasons are related with the quantification of ecosystem services. In an informed society, environmental administration and policy come to decisions on knowledge and numbers. Thus, an influence of the decision making process towards the implementation of multifunctional land use concepts for wetlands requires understanding of the involved processes and quantification of the effect of a changed land use or water management on aspects of the regulation and information function. Mathematical models and decision support systems are tools for this purpose (Bunnell and Boyland, 2003). Developing decision support tools for application in environmental agencies is a tightrope walk between scientific accuracy and end-user requirements. A good DSS represents the scientific understanding of the problem structure and takes from the beginning data availability and accuracy into account. To meet these requirements, the DSS WETTRANS was developed in close co-operation with potential end-user as web-based, user-friendly software. All equations are documented and explained in an online help functions.

Model calculations and many scientific studies indicate that nutrient retention in wetlands can be managed if rehabilitation projects aim to reduce human control on water and nutrient exchange processes in wetlands (Fisher and Acreman, 2004; Trepel and Kluge, 2004). The results from this study indicate that it is more effective evaluating and ranking wetland restoration projects before they are implemented than to implement restoration projects just by what is offered. This stage requires definitions of clear goals such as improving nitrogen retention through changes in the hydrological connectivity. In this study, the goal can be better reached if the proposed sites are ranked according to their cost-effectiveness instead of their maximum effect on load reduction.

At the same time the calculations point out that a significant reduction of the nitrogen load through wetland restoration is only possible if a higher proportion of wetlands is restored. A nitrogen load reduction of 10% through wetland restoration in Schleswig-Holstein will require approximately 20000 ha of rewetted wetlands and will cost around 10% of the budget (210 million €) spent for the waste water action plan.

This amount will be only available if funds for water management are shifted towards the implementation of wetland rehabilitation projects. Supporting this concept, the European community should co-finance such projects, as they did with measures taken during the waste water action plan and in the 1950s with the drainage plan. Only, if this strategy of applying environmental services to solve recent ecological problems is supported with funding from a higher political body such as the European Union it will be in the long time successful.

5. Conclusion

1. Compared to measures taken during the waste water action plan in Schleswig-Holstein, wetland restoration is a cost-effective strategy for nitrogen load reduction.
2. The cost-effectiveness of restoration projects can be quantified for individual projects with wetland nutrient retention models such as WETTRANS. When evaluating all project proposals beforehand it is possible to rank the projects and fund only projects which are most cost-effective.
3. Applying the concept of ecosystem services in environmental management requires a policy shift on European level to support governments with funding for the implementation of such cost-effective strategies to achieve a significant effect on the desired goal.

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