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Lessons learned? Effects of nutrient reductions from constructing wetlands in 1996–2006 across Sweden

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ABSTRACT

Water authorities are currently preparing the second river basin management plans to improve water status in Europe according to the Water Framework Directive (WFD). The on-going efforts of wetland constructions are related to historical efforts in draining land and lakes for agriculture. In Sweden, one of the major problems for surface water is eutrophication caused by diffuse pollution of nitrogen and phosphorus from agriculture. For the whole country, more than 2000 constructed wetlands covering in total 76,000 ha are now suggested to improve the water status. However, this is a small number for the size of the country. This study presents detailed calculations of effects from previous wetland constructions during the years 1996–2006, in which 1574 wetlands (in total 4135 ha) reduced the load to the sea by 0.2% for nitrogen and 0.5% for phosphorus. Even with more optimal allocation, increasing the efficiency, the maximum effect on the total river load would have been small. The simulated efficiency of wetlands varied between catchments in a range of 0.1–340 kg ha^{-1} yr^{-1} for N and 0.01–37 kg ha^{-1} yr^{-1} for P. The variation between wetland efficiency could be explained by large-scale patterns of nutrient concentrations and water discharge or by the specific location of the wetland within a catchment. A sensitivity study showed that each assumption behind the model and the model set-up contributed to uncertainty in the simulation results, but the largest uncertainty refers to the estimation of wetland-catchment area, model parameters and nutrient load. The study involved a lot of data-mining as most wetlands constructed were not accompanied with monitoring programmes, metadata and information on wetland characteristics. It is shown that Sweden still is very far from reconstructing natural conditions, and that more radical measures and combinations of measures are needed to achieve the WFD goals of good water status.

1. Introduction

During the last century, human activities such as agriculture, industry and transportation have increased the mobility of nutrients from land to aquatic ecosystems (e.g. Meybeck, 2002; Vörösmarty et al., 2010; Falkenmark, 2011) and as a consequence eutrophication is a major issue affecting most surface waters (Smith and Schindler, 2009; Grizzetti et al., 2012; Romero et al., 2012). However, during the last decades, measures to reduce riverine nutrients have been implemented across Europe in form of waste-water treatment, better agricultural practices and restricted use of phosphorus (P) products (EEC, 1991a,b). This have resulted in decreased phosphate and/or nitrate concentrations since the mid-1990s in several European rivers such as the Elbe (Lehmann and Rode, 2001), the Seine (Billen et al., 2007), the Rhine (Hartmann et al., 2007), Mediterranean rivers (Ludwig et al., 2009), the Thames (Howden et al., 2010), the Danube (Istvánovics and Honti, 2012), as well as for Swedish rivers (Grimvall et al., 2014).

In spite of the improvements, the current most common pressures to surface waters in EU member states are still diffuse sources and hydromorphological alterations (EEA, 2012), which causes nutrient enrichment and altered habitats (Künitzer, 2013). For nutrient load in rivers, the main sources across Europe are agriculture and point-sources (Donnelly et al., 2013). At present, European water authorities are preparing measure plans to improve the water status, which should be reported by the end of 2015 according to the second cycle of the Water Framework Directive, WFD (EC, 2000). In Sweden, the measure plans are currently available at http://www.vattenmyndigheterna.se for participatory dialogues with stakeholders. Especially the southern districts are very concerned with eutrophication (Vattenmyndigheterna i samverkan, 2014a,b,c) both for inland and coastal waters. Moreover, Sweden has committed to the Helsinki Commission (HELCOM) to implement the Baltic Sea Action Plan (BSAP), which

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is an ambitious program to restore the Baltic Sea in good ecological status by 2021 (HELCOM, 2007).

Among the measures proposed to fulfil both the WFD and the BSAP, are various wetlands constructed in the landscape. Natural wetlands are known for reducing diffuse nutrient loads (Mitsch et al., 2002; Fisher and Acreman, 2004; Verthoeven et al., 2006; O'Geen et al., 2010) and in Sweden large wetland areas disappeared during the 19th and 20th century to increase agricultural yields. In total about 2500 lakes were shrunk or dried out, and 30,000 soil-drainage projects were performed (SMHI, 1995). At the end of the 19th century some 10,000 ha per year were transformed into arable land and agricultural practices became more efficient. As a consequence the nutrient leaching, load and natural removal processes changed over time (e.g. Hoffman et al., 2000; Andersson and Arheimer, 2003).

The opposite trend to construct wetlands in the landscape started in Sweden during the late 1980s. The reasons were several, including bird watching or hunting, biodiversity, landscape reconstruction and nutrient reduction. To begin with, very high nitrogen (N) reduction rates were proposed, based on empirical data including wetlands for wastewater treatment with high concentrations and regulated flow (e.g. Fleischer and Stibe, 1991). A national research programme on wetlands and lakes as N traps was reported in 1994 (Jansson et al., 1994). From a hydrological point of view, however, critical questions followed whether these results could be realistic for natural conditions (Bergström, 1991). Subsequent modelling efforts showed that natural fluxes and allocation strategies had a significant impact on the wetland efficiency on river load (Arheimer and Wittgren, 1994, 2002).

Hence, some guide-lines were proposed for wetland constructions to reduce the load on Swedish inland waters and coastal zones efficiently; the wetlands should be allocated in coastal agricultural areas, downstream from lakes and where summer load is a large part of the annual load (Arheimer and Wittgren, 1994). Moreover, >1% of the catchment area should be converted into wetlands to receive >10% N reduction. This latter statement was found unrealistic in practice and empirical field studies suggested 0.4% of the arable land to be converted in southern Sweden (Holmström, 2003). The importance of wetland design for efficient nutrient reduction in Nordic climates has also been high-lighted repeatedly, e.g. impact of water residence time (Koskiaho et al., 2003), flow uniformity and effective volume (Persson et al., 1999) and plant species (Kallner Bastviken et al., 2005).

In Sweden the government has, among other measures, allocated more than 1300 million SEK (approximately 130 million Euros) for wetland constructions in the landscape during the period 1996–2011 (Table 1). When subsidies were given by the Swedish government for wetland constructions, however, no clear guidelines for allocation strategies or design were given, nor requests of control programs or monitoring of effects. Subsidies were mainly given within a rural development program (RUP: 1996–2006), and a local investment program (LIP: 1998–2002). In total, more than 5000 ha of wetlands were constructed for different purposes, and about half of them were dedicated primarily to nutrient reduction.

For the wetlands constructed 1996–2006 with a total connected area of 2952 km^2, the cost was on average €51,000 per wetland when calculated for a total cost of €80 million. The cost was €271 per ha of connected catchment area and €19,000 per ha of wetland area.

In this study, we explore the total effects of wetlands constructed between 1996 and 2006 for reducing nutrient pressure on inland and coastal zones in Sweden. We assume that nutrient load reduction by wetland constructions and variability in efficiency can be estimated by using a dynamic and integrated catchment model with spatial distribution. Similar landscape analysis have been performed for single catchments using various model concepts (e.g. Tonderski et al., 2005; Hattermann et al., 2008; Hansen et al., 2009; Passy et al., 2012; Pärn et al., 2012). However, no study of our knowledge has yet encompassed the national scale using a multi-basin approach for thousands of wetlands as in this paper. Three questions are addressed in the study:

(1) How effective in reducing nutrient load are the wetlands that were constructed at a large-scale across Sweden in 1996–2006?
(2) How robust are the results and conclusions from this modelling study?
(3) What lessons did we learn for the future?

Table 1
Planned and constructed wetlands across Sweden from 1996 and onwards.

	No.	Area (ha)	Cost (Euro)
Planned: WFD 2nd programme of measures (2015–)	2184	76,080	3650 million
Constructed in 2009–2011	564	1468	70 million
Constructed in 1996–2006	1574	4135	60–100 million

2. Material and methods

The wetlands are all constructed in southern Sweden (Fig. 1) and the study is limited to an area of 164,000 km^2, of which about 17% is agricultural land and 11% surface water (i.e. lakes). Less than 2% of this area is connected to the wetlands constructed. The constructed wetlands show higher density in southernmost part and south of Lake Vänern. In these areas agriculture is dominating and the nutrient leakage to surface water is high. Detailed information

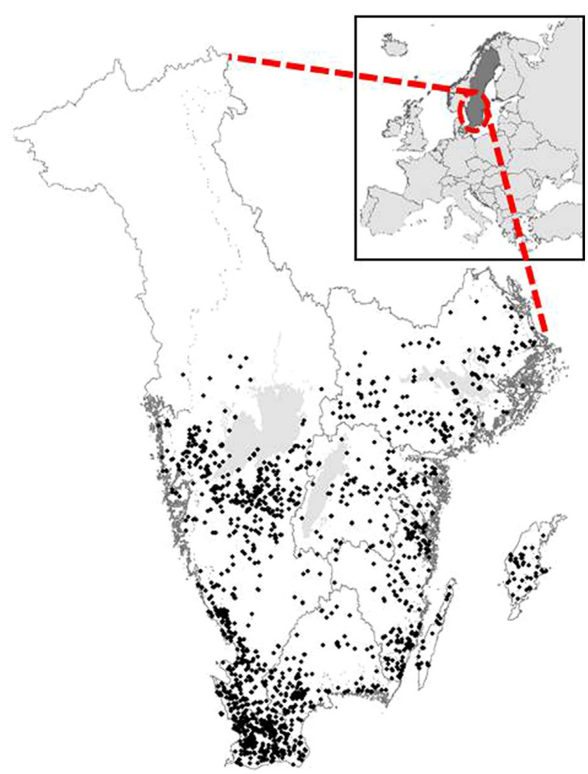

Fig. 1. Location of wetlands constructed in Sweden during the years 1996–2006.

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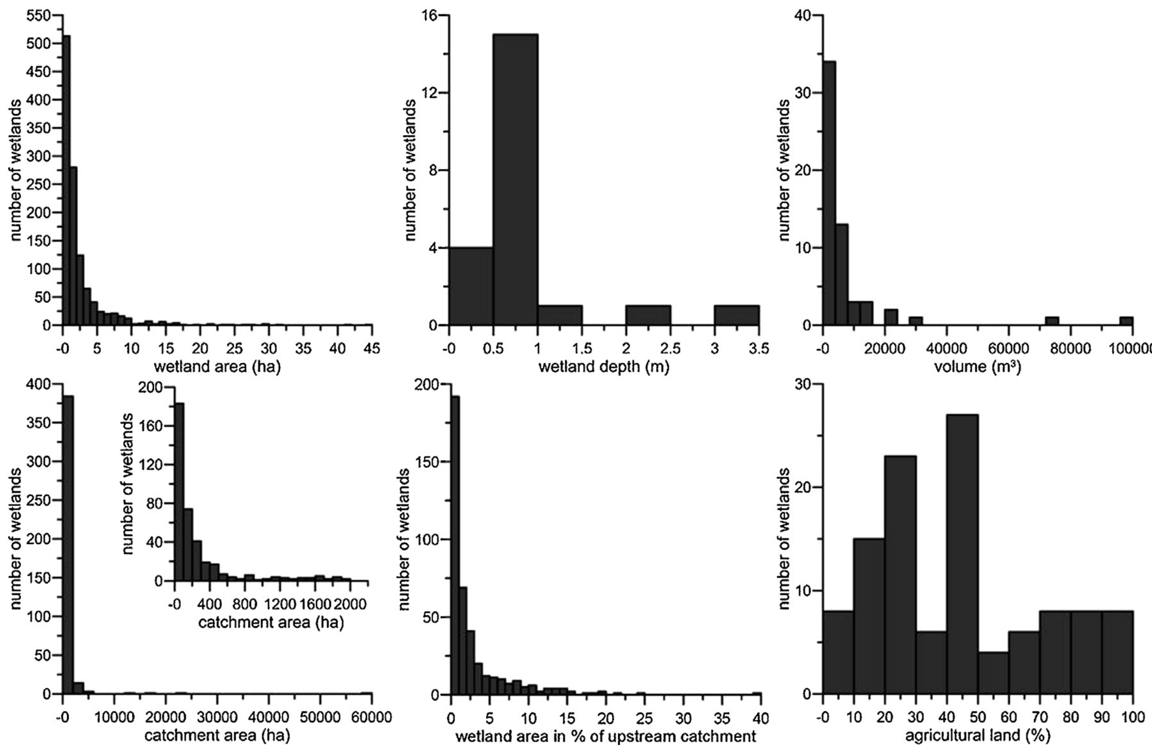

Fig. 2. Distribution of wetland properties among the ones with known data.

about the wetlands is generally sparse but publically available on the web at http://vattenwebb.smhi.se/wetlands/. The constructed wetlands properties vary among the wetlands as shown in Fig. 2.

Catchment-based modelling of water and nutrients for the entire country of Sweden has been performed since the year 2000 on request by the Swedish Environmental Protection Agency, e.g. to calculate the load, retention and source apportionment of nutrients to the surrounding seas for international reporting to the marine commissions HELCOM and OSPAR. A catchment-based approach is appropriate as the direct groundwater inflow can be neglected; the amount of direct groundwater flow to the Baltic Sea compared with total runoff is most likely very small probably around or even less than 1% (Peltonen, 2002). At present, the HYPE model (Lindström et al., 2010) is used for the WFD measure planning and simple estimates of measure effects can be made on-line at http://vattenwebb.smhi.se/scenario/. However, for the present study an earlier model system was used to quantify the effect of constructed wetlands and for investigating the robustness of results in the sensitivity analysis (Brandt et al., 2008, 2009). Note: The unit metric ton is referred to as tonnes or t throughout the paper.

2.1. Integrated catchment modelling

The model concept includes dynamic process-based descriptions of nutrient leaching in agricultural soils, using the models SOILNDB (Johnsson et al., 2002) and ICECREAM (Tattari et al., 2001), linked to a rainfall-runoff model called HBV-NP. The latter integrates water balance, water flow, soil nutrient leaching, additional nutrient sources, nutrient residence and transformations in ground and surface water as well as typology, extrapolation and routing functions to cover the national scale (Arheimer, 2003; Pers, 2007). The HBV-NP model simulates the hydrology of coupled sub-catchments called subbasins. The model is based on the semi-distributed conceptual HBV model (Lindström et al., 1997), which is a hydrological model that first was equipped with a N module (Arheimer and Brandt, 1998) and later also with a P module (Andersson et al., 2005) and applied to the entire country (e.g. Brandt et al., 2007, 2008). The selected sizes of subbasins provide the spatial distribution and in the present application the subbasin areas range between 200 and 700 km^2.

The model was run on a daily time-step for the period 1985–2004 to account for weather fluctuations and daily variations of water and nutrient inflow to the wetlands. The dynamic model concept thereby account for variations in hydraulic residential time and nutrient storage in water bodies (river, wetlands and lakes). In this study, the results are given as accumulated average load, which are thus flow normalized. The original national set-up of the HBV-NP model was modified to simulate several types of wetlands, according to their position in the landscape. The constructed wetlands were added to the national model set up and the original model application (Brandt et al., 2008) was used for base-line conditions.

2.2. Wetland modelling

The basic model assumption in HBV-NP is that wetland turnover of nutrients is driven by nutrient concentrations and load, nutrient residence time in the wetland, temperature and water flow, according to Eqs. (1)–(4). We assume a perfectly mixed reactor. For every day the current nutrient concentrations and temperature determine the nutrient retention, while daily water flow determines the hydraulic residential-time and how long the nutrients stay in the wetland subjected to the retention processes. This accounts for temporal variations in retention and nutrient storage, although, when presented in this paper, the results are aggregated over time.

$$\frac{\mathrm{d}(cV)}{\mathrm{d}t} = c_{\mathrm{in}}Q - \Phi - cQ \tag{1}$$

$$\Phi_{\mathrm{IN}} = wret \times c \times \overline{T_5} \times A \tag{2}$$

$$\Phi_{\mathrm{TP}} = wsedp \times c \times A \times 10^3 \tag{3}$$

$$\Phi_{\mathrm{TP}} = -wrel \times c_{\mathrm{in}} \times 1.2^{\left(\overline{T_{30}}-20\right)} \times A \times 10^3 \tag{4}$$

where A is the wetland area (km^2), c is the concentration of nutrient in wetland (g m^{-3}), c_{in} is the concentration of nutrient in inflow (g m^{-3}), dt is the time step; one day, Q is the inflow and outflow (m^3 d^{-1}), V is the volume of wetland (m^3), Φ is the retention of inorganic N (IN) and total P (TP) (kg d^{-1}), $\overline{T_n}$ is the average air temperature over n days (°C), *wret* is a model parameter; inorganic N retention rate (mm °C^{-1} d^{-1}), *wsedp* is a model parameter; total P sedimentation rate (m d^{-1}) and *wrel* is a model parameter; total P release rate (m d^{-1}).

Information about the wetlands has been compiled from overlapping databases; the register for area based subsidy (Ararat) at the Swedish Board of Agriculture (SBA), which contains all RUP-wetlands, and Svensson et al. (2004), which contains wetlands from both RUP and LIP programs up until 2002. Information regarding wetland size, depth, position, wetland catchment area, and land use in the catchment was needed for calculations of wetland effect on the nutrient transport.

Additional wetland information could be achieved from the authorities in charge for subsidies, such as the Swedish Agricultural Board and County Boards. In some cases, additional information was available at the Wetlands Centre in Halmstad. Some wetlands appeared several times in the databases and a cleaning procedure was necessary to compile a harmonised database, which eventually included 1574 wetlands (totally 4135 ha) of unique identities. However, for many wetlands the information needed was not available (Table 2, Fig. 2) and had to be estimated according to a 'best guess' strategy. Thus, when wetland area and depths were missing, median values for the known material were assumed (Table 2).

Catchment boundaries were defined individually for each wetland as this information was not in the databases. This was a very labour intensive part of the work. Automatic routines were first applied using ArcHydro in ArcGIS, although the available DEM had a spatial resolution of 50 m and an average error of 2 m in height. However, this resolution was found to be insufficient to capture the small features in the landscape that constituted the borders of the catchments, as the wetlands were often located in flat areas. Hence, manual corrections were needed considering hydrography and realistic judgements. For some wetlands the catchments had been estimated by field observations at the County Boards, and this information was used for validation of the estimates. It was then concluded that the automatic method generally underestimated the size of wetland catchments, often 5 times (Brandt et al., 2009). Some of the wetlands, which were allocated very closely to each other were aggregated and simulated as one single wetland, so that the original 1574 wetlands were merged into 1504 calculation units in the national model. This means that in reality, at most 4% of the wetlands could be allocated in series or parallel but this is thus neglected in the simulation.

The code of the HBV-NP model was modified to simulate several wetlands at different locations within each subbasin without changing the original subbasin division. In that way, each wetland was given unique characteristics and specific load of water and

Table 2
Number of wetlands (out of total 1574) with available information, and minimum and maximum values found. "Best guess" values of wetland area, depth and type used when data is not available. "Best guess" values of wetland area and depth are corresponding to median values of the available information.

	No	Minimum	Maximum	"best guess"
Wetland area (ha)	1538	0.03	44.1	1.2
Depth (m)	22	0.25	3.5	0.7
Catchment area (ha)	405	0.3	60,000	
Agriculture in catchment (%)	421	5	100	
Wetland type	794			
• Small wetlands (swet)				60%
• Parallel wetlands (lpwet)				40%

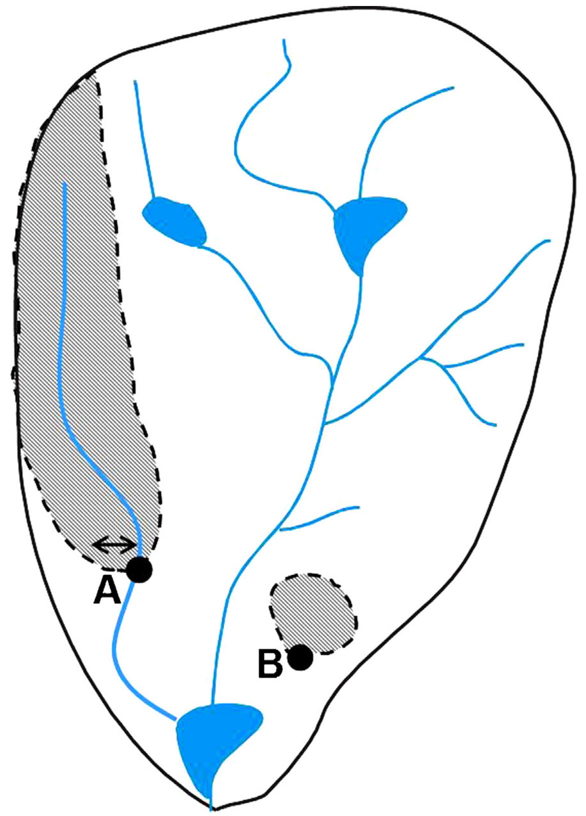

Fig. 3. Two types of wetlands are modelled in a subbasin; (A) wetlands receiving part of the stream water from the wetland catchment or (B) wetlands receiving all input from the wetland catchment.

concentrations, while in earlier versions subbasin average conditions were used (e.g. Arheimer and Wittgren, 2002; Tonderski et al., 2005). This was introduced since the subbasins were rather large and average conditions were not considered representative for the small catchments of the constructed wetlands. In the model, each subbasin can include numerous wetlands, which are all calculated individually. The constructed wetlands always receive inflow from the local stream. Two kinds of wetlands were identified (Fig. 3): (A) wetlands in connection to a stream, but only receiving part of the water, (B) isolated wetlands in the subbasin, receiving all flow in the stream of the wetland catchment. According to the information gathered, 60% of the wetlands belonged to this latter category, and the same distribution was assumed also for remaining uncategorised wetlands (Table 2). Based on a survey of 6 wetlands by the consultant group Ekologgruppen (pers. com.), in average 30% of the river discharge was assumed to pass through the wetlands of category A. In the model we assumed that 30% of the flow each day was diverted to the wetland.

Calculations of the wetland catchment load were simplified compared to the original HBV-NP, because of several wetlands per subbasin. Daily amount of water generated was determined by the size of the wetland catchment, assuming an areal-proportional part of the daily water discharge of the subbasin, i.e. no spatial variation of runoff within the subbasin was simulated here. Nutrient concentration in water inflow from arable land was received from the national field-scale modelling of agricultural regions, considering land use, soil type and crop distribution the same way as normal (Johnsson et al., 2008; Brandt et al., 2008). These catchment characteristics were specifically extracted for each wetland catchment by using GIS and national databases. Land use in the wetland catchments was to most part arable land (45%), but forests were also a large part (37%). On arable land, the most common crops

(80%) were ley, pasture, green fallow, winter wheat and spring barley. Wetland inflow concentration was then adjusted to include other sources (such as rural household emissions) and retention in soil/groundwater/small streams, by assuming equal distribution over the subbasin.

The model parameters in Eqs. (2)–(4) were estimated through calibration of the wetland model against empirical time-series (1–3 years) of weekly values from 14 wetlands for N and 9 wetlands for P. Most monitored wetlands were found in the southern part of Sweden, but also south-eastern and south-western conditions were represented. The model parameters showed large ranges when calibrated individually for each wetland. Attempts were made to categorise results by finding relations between parameter values and region or wetland characteristics, but without success. Finally, median values were used for the national calculations, i.e. *wret* = 2.8 mm °C^{-1} d^{-1}, *wsedp* = 0.09 m d^{-1}, *wrel* = 0.05 m d^{-1} (Brandt et al., 2009).

2.3. Sensitivity analysis

As described above, essential information of the wetlands were missing. Hence, the estimation of the wetland effect on nutrient load was based on many assumptions. To investigate the robustness of results and conclusions a sensitivity analysis was performed. The following assumptions were tested with regard to the first "best guess" assumptions:

- *Wetland area* was only missing for a few objects (2.5%). The 'best guess' assumption used the median value for all wetlands as a proxy for wetlands without observed area. However, in the sensitivity study wetlands with missing area were assumed to have the maximum (44.1 ha) or minimum (0.03 ha) size according to the database. The size of the wetland is important for the retention processes but also for wetland volume and hydraulic residential-time.
- *Wetland depths* were missing in most cases (Table 2), despite its importance as it determines wetland volume and hydraulic residential-time. Median values for the wetlands with information were used as 'best guess' for missing values, while maximum (2 m) and minimum (0.2 m) values were applied for missing values in the sensitivity analysis.
- *Wetland type*, i.e. whether the wetland received all water from the wetland catchment or not, were not given in most cases and was difficult to decide from maps. In the 'best guess' assumption the distribution 60–40% were used for the two types, while in the sensitivity analysis all wetlands without classification were set as the one and the other, to distinguish the two extremes.
- *Parallel water flow* not going through the wetland will of course affect the wetland capacity to reduce the riverine load. Wetlands taking water from a nearby stream was considered to receive 30% of the flow in the 'best guess'. In the sensitivity study this figure was increased to 70% to investigate the influence of this variable on overall results and conclusions.
- *Active wetland* area. The 'best guess' approach assumed that the whole wetland was available for nutrient turnover processes to take place. However, it has been argued that only part of the wetland is active, due to hydraulics (Persson et al., 1999) or active surface (Svensson et al., 2004). Based on these studies, the sensitivity analysis assumed only 80% of the wetland area to be active.
- *Model parameters* are describing the rates of processes influencing nutrient removal in wetlands. Few wetlands had monitoring programmes and it was difficult to find parameters with optimal fit compared to empirical data for all wetlands. In the sensitivity analysis the whole range of values were tested (*wret*: 2.3–10; *wsedp*: 0005–0.5; and *wrel*: 0.03–1), using the highest and lowest values from each individual calibration on all wetlands.
- Size of the wetland *catchment area* was difficult to estimate due to databases with low resolution (see above). The catchment size determines the water flow, the load on the wetland and the water turnover-time, which are essential for nutrient removing processes. In the sensitivity study the catchments were increased by 25 times or decreased by 50 times, which covered the large range in error estimate.
- Inflow *nutrient concentration* was determined by land use. In the 'best guess' assumption the concentration was mainly the result of agriculture in the wetland catchment according to available data of crops. Three other assumptions were made for nutrient concentrations in water inflow by changing crops in wetland catchments to give: (i) *mean* concentrations, same as for the whole subbasin, (ii–iii) the *range* between the highest and lowest leaching concentrations possible. In the latter two simulations, the small wetlands had priority when crops present in the subbasin were re-allocated to the wetland catchments. Crop distribution in the overall subbasin remained the same.

3. Results and discussion

3.1. Modelled effect of constructed wetlands

The catchment modelling showed that the 1574 wetlands constructed in 1996–2006 reduce the nutrient transport to the sea by 110 t of N per year and 9 t of P per year. This corresponds to less than 0.2% of the total N transport from the 164,000 km^2 of southern Sweden, (66,000 t) and 0.5% of the total P transport (1700 t). Most of the reduction is found in the southern and western parts of the region (Fig. 4) where most of the wetlands are located. Hence, as a combat measure to save the Baltic Sea from eutrophication, the efforts in the past to construct wetlands across Sweden have not been very effective. Only 1.8% of the southern Sweden area was connected to wetlands.

The local effect on river concentrations close to the wetlands is higher than for nutrient transport to the sea. For the whole study area, the gross removal was estimated at 140 t N per year and 12 t P per year. Wetlands with high local efficiency but located upstream of large lakes do not have equal effect on reducing the transport to the sea due to natural retention of nutrients in the lakes. The presence of lakes in the northeast regions (and partly the western region), gave up to 60% less efficiency of nutrient reduction in wetlands on the total transport to the sea compared to the local average efficiency (Fig. 5). However, specific wetlands could still be efficient to reduce eutrophication locally in specific inland waters, if allocated properly. We found a large range in efficiency of nutrient reduction among individual wetlands.

The effect of wetlands varied over southern Sweden, but the average local efficiency for all simulated wetlands was estimated to 33 kg ha^{-1} wetland yr^{-1} for N and 3 kg ha^{-1} wetland yr^{-1} for P. However, the efficiency of wetlands within a subbasin varied between catchments in a range of 0.1–340 kg ha^{-1} yr^{-1} for N and 0.01–37 kg ha^{-1} yr^{-1} for P. These estimates are the combined efficiency of all wetlands within a subbasin, thus the ranges might be larger for individual wetlands. The highest efficiency for N was found in the southern and south-western parts (Fig. 6), where the average efficiency in wetland reduction was more than twice that of the other regions. For P the spatial pattern was more scattered.

The modelled range in this study is within the large variation of wetland efficiency to reduce diffuse nutrient load that have been reported in previous work for Nordic countries, for N, e.g.: 0–1300 kg N ha^{-1} yr^{-1} (Arheimer and Wittgren, 2002), 0–7000 kg N ha^{-1} yr^{-1} (Fleischer et al., 1994), 11–280 kg N ha^{-1} yr^{-1} (Koskiaho et al., 2003), and for P, e.g.: 0–24 kg P ha^{-1} yr^{-1} (Koskiaho et al., 2003), 16–50 kg P ha^{-1} yr^{-1} (Tonderski et al., 2005),

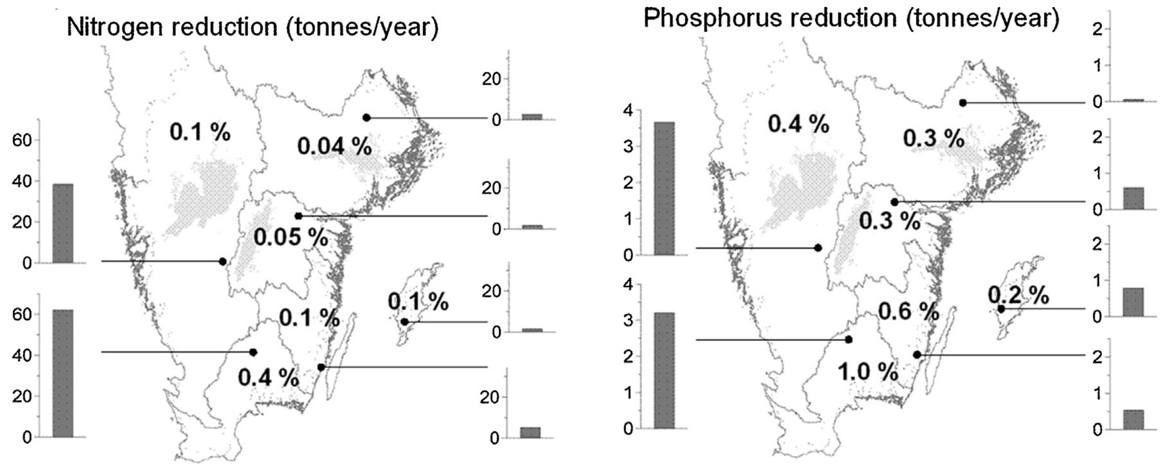

Fig. 4. Reduction of nitrogen (left) and phosphorus (right) transport to the sea by constructed wetlands. The map shows the reduction in percent of the transport for different regions in southern Sweden, and the bars the total reduction in tonnes per year for the same regions.

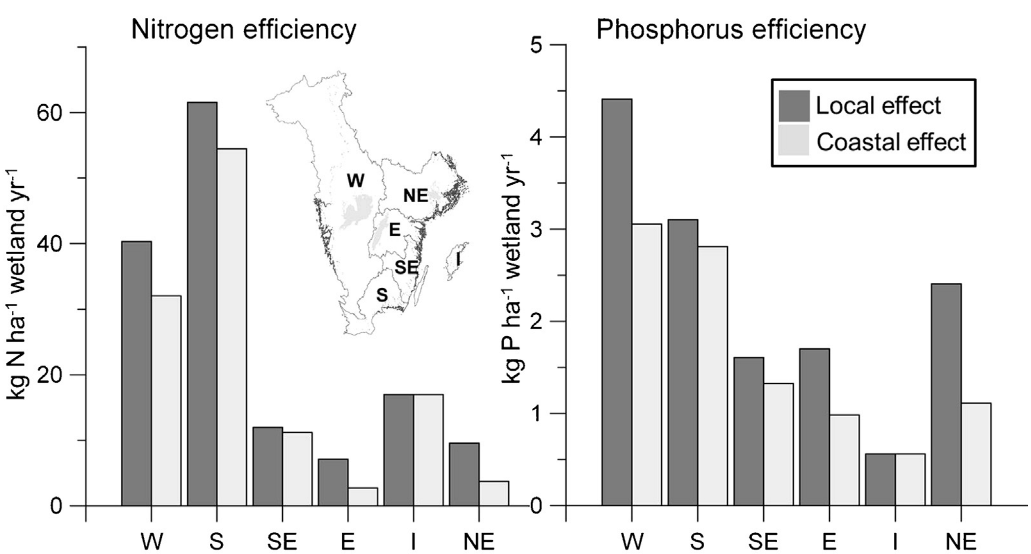

Fig. 5. Average effect of wetland reduction of nitrogen (left) and phosphorus (right) for different regions. From left and counter clockwise: W—western region, S—south region, SE—southeast region, E—eastern region, I—island region, and NE—northeast region.

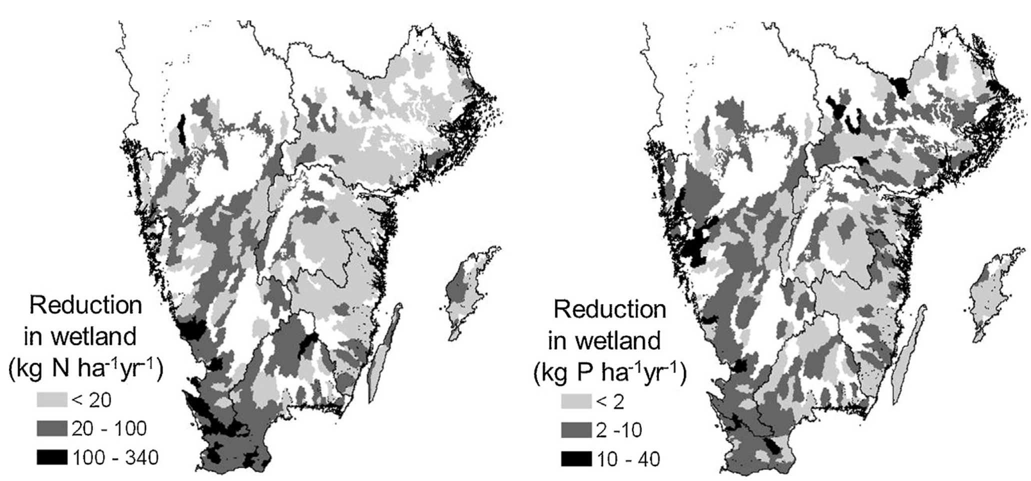

Fig. 6. Average efficiency in reduction of nitrogen (left) and phosphorus (right) per constructed-wetland area in each subbasin of southern Sweden. In white areas no wetlands were constructed in 1996–2006.

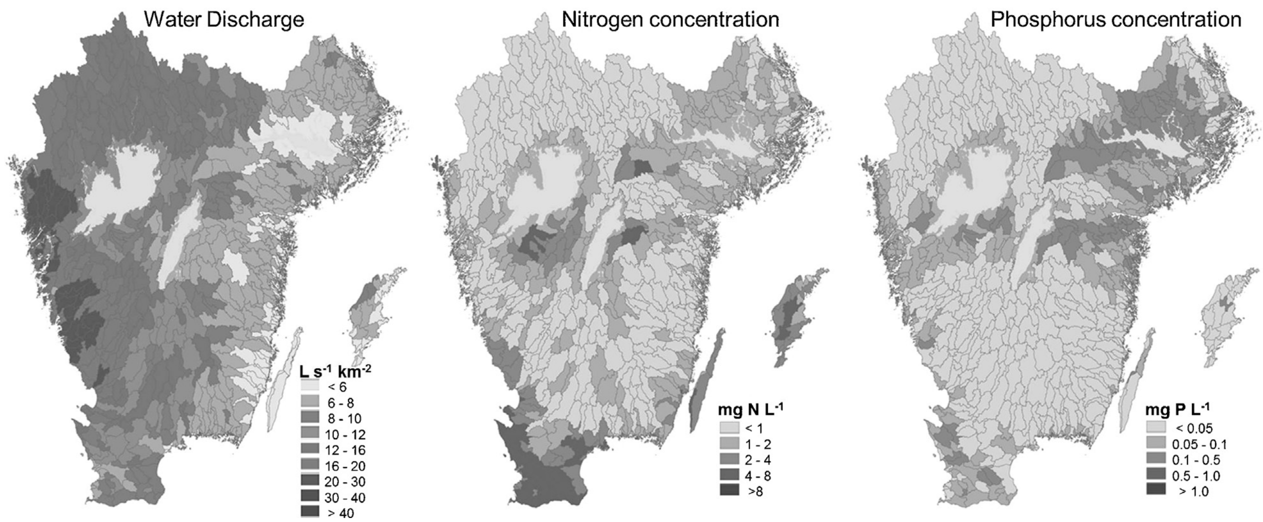

Fig. 7. Spatial patterns of water discharge (left) and riverine concentrations of nitrogen (middle) and phosphorus (right), as mean values per subbasin for the period 1986–2006.

20–460 kg P ha^{-1} yr^{-1} (Uusi-Kämppä et al., 2000). Braskerud (2002) reported as much as 260–710 kg P ha^{-1} yr^{-1} for P reduction in Norwegian wetlands, but this may be linked to landscape characteristics and the high surface erosion from fields on steep hillsides.

The spatial pattern of wetland efficiency for nutrient reduction is related to the spatial pattern of drivers like water flow and concentrations. In our study, low N efficiency is generally found in the regions NE, E, SE and I (Figs. 5 and 6). These areas have low runoff (Fig. 7) and thus low wetland load. Some of the subbasins in this area do have higher efficiency due to higher concentration of inflow, thus slightly higher load by this reason. Likewise higher N efficiency is found in the southern and western parts as these areas have high N load due to high runoff and/or high concentrations. These areas have sandy soils, which gives high N leaching from agricultural land (Johnsson et al., 2008). For P, on the other hand, no clear regional pattern can be discerned (Fig. 6). Clayey soils in NE and E regions result in higher P concentrations in surface water (Fig. 7).

Variation between subbasins within these large-scale patterns can be caused by the specific location of the wetland(s). The land use distribution of the wetland's catchment in relation to the subbasin land use distribution (i.e. nutrient load) and the wetland catchment area in relation to the wetland area (i.e. the hydraulic residential time) are factors that make wetland efficiency in a specific subbasin stand out among its neighbours. Nitrogen removal efficiency is also affected by site specific conditions such as organic carbon and anaerobic conditions (e.g. Passeport et al., 2012). The spatial pattern of wetland effect on the river load to the sea mainly reflects the presence of lakes downstream of the wetland. Natural nutrient reduction in lakes can be considerable in Sweden (Arheimer and Brandt, 1998) and that makes up-stream regions less important for the load on the sea. The reduction of load by a wetland is reduced by any lake retention between the wetland and the sea. Since the gross load would anyway be reduced in the lake, only part of the wetland nutrient reduction will affect the load to sea compared to for a similar wetland downstream the lake. In addition, if the nutrient reduction is large above the lake, the lake itself becomes less efficient as a natural nutrient traps if the load is reduced. Downstream of a lake, however, no such compensation between these different retention traps in the landscape will appear. The present study thus supports the previous conclusions that wetlands (and other nutrient reducing measures) should be allocated in coastal agricultural areas with high concentrations and downstream from lakes to be effective as nutrient traps for the sea (Arheimer and Wittgren, 1994; Arheimer et al., 2005).

3.2. *Robustness of results*

The sensitivity study shows that the largest uncertainty in the simulated wetland efficiency refers to the estimation of wetland-catchment area, model parameters and nutrient load (Fig. 8). Each assumption behind the model and the model set-up contribute to uncertainty in the simulation results, but the model sensitivity is less for other input data. The uncertainty of the simulated efficiency related to assumptions of catchments areas is of the same magnitude as the calculated wetland efficiency using the best guess assumption. The changed load simulations show the efficiency of the wetlands if they were allocated differently in the landscape. It is thus not actually an estimate of the uncertainty in our best guess simulation, but a boundary of possible loads. Still, even in these extreme scenarios the maximum effect of the wetlands on nutrient transport to the sea is only 0.5% for N and 2% for P.

The simulations' sensitivity to missing information of wetland area and depth are of similar magnitude in the model (Fig. 8). Yet, in the gathered database a lot more wetlands miss information of depth than of area (Table 2). Both these input data influence the modelled volume of the wetland and thus the hydraulic residential-time in the wetland and reduction rates (cf. Eqs. (2)–(4)). The simulations with changed wetland type and water inflow to parallel wetlands also influence the retention time in the wetland by shifting the amount of water passing through and thereby the load. The sensitivity to these changes is similar or larger than the previous two (Fig. 8). For these four simulations only wetlands that lack information have their assumptions changed, which means all wetlands for the water intake simulation. The study of input data sensitivity shows the importance of sufficient metadata and characteristics to be able to estimate nutrient reducing effects of constructed wetlands.

The wetland simulation is also sensitive to the chosen model parameter values (cf. Eqs. (2)–(4)). In the 'best guess' simulation, we used median values of the parameters from individual calibration of empirical time-series. For N the best guess result was close to the result of using minimum values (Fig. 8). For P the minimum values have no wetland reducing effect at all, because some of the wetlands used for calibration show no annual net reduction on P.

The present wetlands were not allocated according to any guidelines on efficiency for nutrient reduction and the results show that a random allocation would have given slightly more efficient

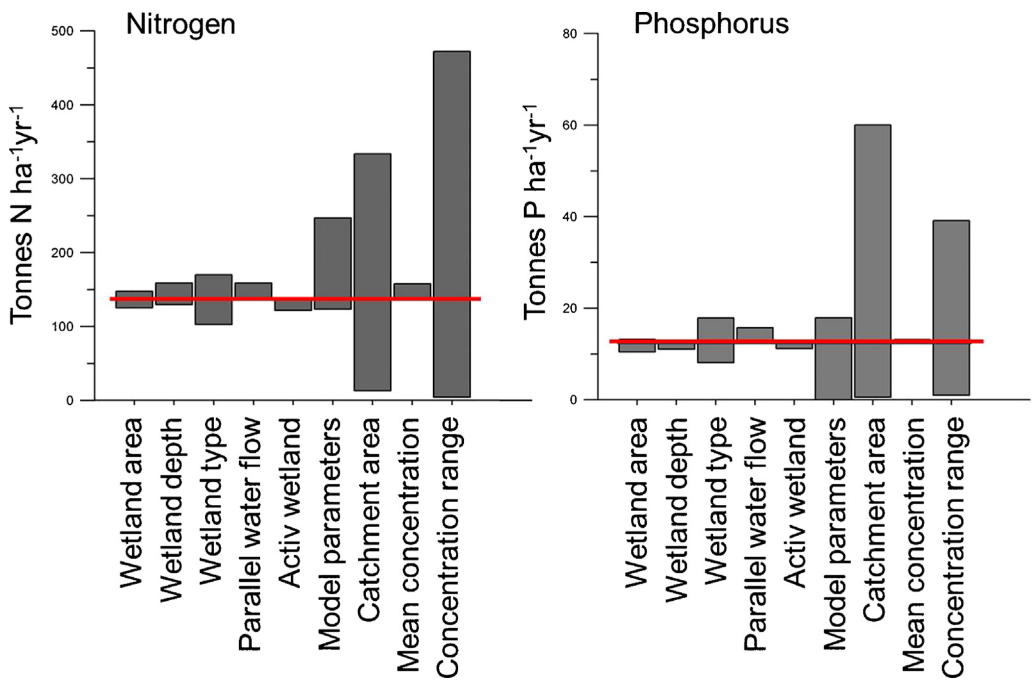

Fig. 8. Simulated nutrient reduction in wetlands in southern Sweden (t yr^{-1}) using different model assumptions. The straight line indicates the result of the 'best guess' simulation.

wetlands than the present locations (i.e. mean concentration gave higher effect than the "best guess" based on crop distribution in wetland catchment).

To sum up, the sensitivity analysis show that the assumptions in the calculations could result in at most five times underestimation of wetland effects. When adding the extremes of sensitivity assumptions for several variables, the maximum modelled annual effect on nutrient reduction was 645 t N per year and 66 t P per year (compared to 110 and 9 t per year, respectively). The minimum effect was zero. The maximum effect corresponds to 1% of the total N load and 4% of the total P load (compared to 0.2% and 0.5%). Although this is a large uncertainty, it is still a very small part of the overall nutrient load to the sea and hardly noticeable when considering the total nutrient flow from land to sea (Fig. 9). The overall conclusion that the wetlands constructed during 1996–2006 in Sweden only have had minor influence on the nutrient transport to the Swedish coast is thus valid, regardless of the uncertainty of the assumptions made in the calculations. The results also indicate that even if the wetlands would have been more optimally allocated according to guide-lines, the total effect would still only be at most one or a few percent of the total load.

3.3. *Lesson learned: The minuscule effect of the wetlands*

It may appear that constructing 1574 wetlands of in total 4135 ha is a lot, and that the effect on nutrient reduction is surprisingly low, when it results in a reduction of 0.2% for N and 0.5% for P transport to the sea. However, the small impact of the wetlands on the total load is logical when the wetlands are put in the perspective of overall nutrient fluxes in the region (Table 3). Less than 2% of Southern Sweden is connected to the wetlands and the wetland area constructed corresponds to 0.02% of the total area. This is far below the recommendations that have been suggested for the region in previous studies (cf. Arheimer and Wittgren, 1994, 2002; Tonderski et al., 2005). In fact, 400 times more wetlands must be constructed to reach 1% of the total area or three times more to reach 0.4% of the agricultural area. In addition, the latter should of course be allocated on arable land, which is seldom the case for the previously constructed (cf. Fig. 2).

Compared to wetlands, lakes were considered to be half as effective nitrogen traps and three times less effective for phosphorus reduction in the modelling. Nevertheless, Table 3 shows that the natural reduction in surface water is high due to large amount of lakes in the region. Compared to modelled nitrogen reduction of agricultural leaching in groundwater and small creeks, the wetland effect only correspond to 1% of this natural nutrient reduction. The change in large-scale nutrient fluxes that is caused by wetlands constructed 1996–2006 is thus miniscule.

The base-line for the calculations in this study is the total load from land to sea (Fig. 9). However, in many national environmental objectives and in the BSAP, a distinction is made between natural and anthropogenic load, as it is only the latter that can be affected by control measures. The natural background load has been calculated for the region (Brandt et al., 2008) and compared to the anthropogenic load the wetland effect would be 0.25% for N and as much as 1.8% for P. Phosphorus thus have a higher effect on anthropogenic load than on total load (0.2% for N, 0.5% for P). This is an effect of higher natural load of P compared to N. Yet, the effect of constructed wetlands remains small as long as they are not implemented more extensively. Compared to pristine conditions, the present efforts are minor and the suggested new 76,080 ha to be converted into constructed wetlands in the WFD measure programmes, only corresponds to a few years of drainage activities a century ago.

Accordingly, the Swedish WFD measure programmes mostly rely on other measures to reduce riverine nutrient load and eutrophication (Gyllström et al., 2014). A whole battery of measures in agriculture are proposed, as well as more efficient treatment plants of point sources (Vattenmyndigheterna i samverkan, 2014a,b,c). Constructing wetlands is only one measure among these. Hopefully, the water authorities will also see to that guidelines are followed regarding where and how to construct the wetlands to be efficient and establish monitoring programmes to be able to better judge their effect in the future. In fact, wetlands are not more expensive to reduce diffuse pollution than other

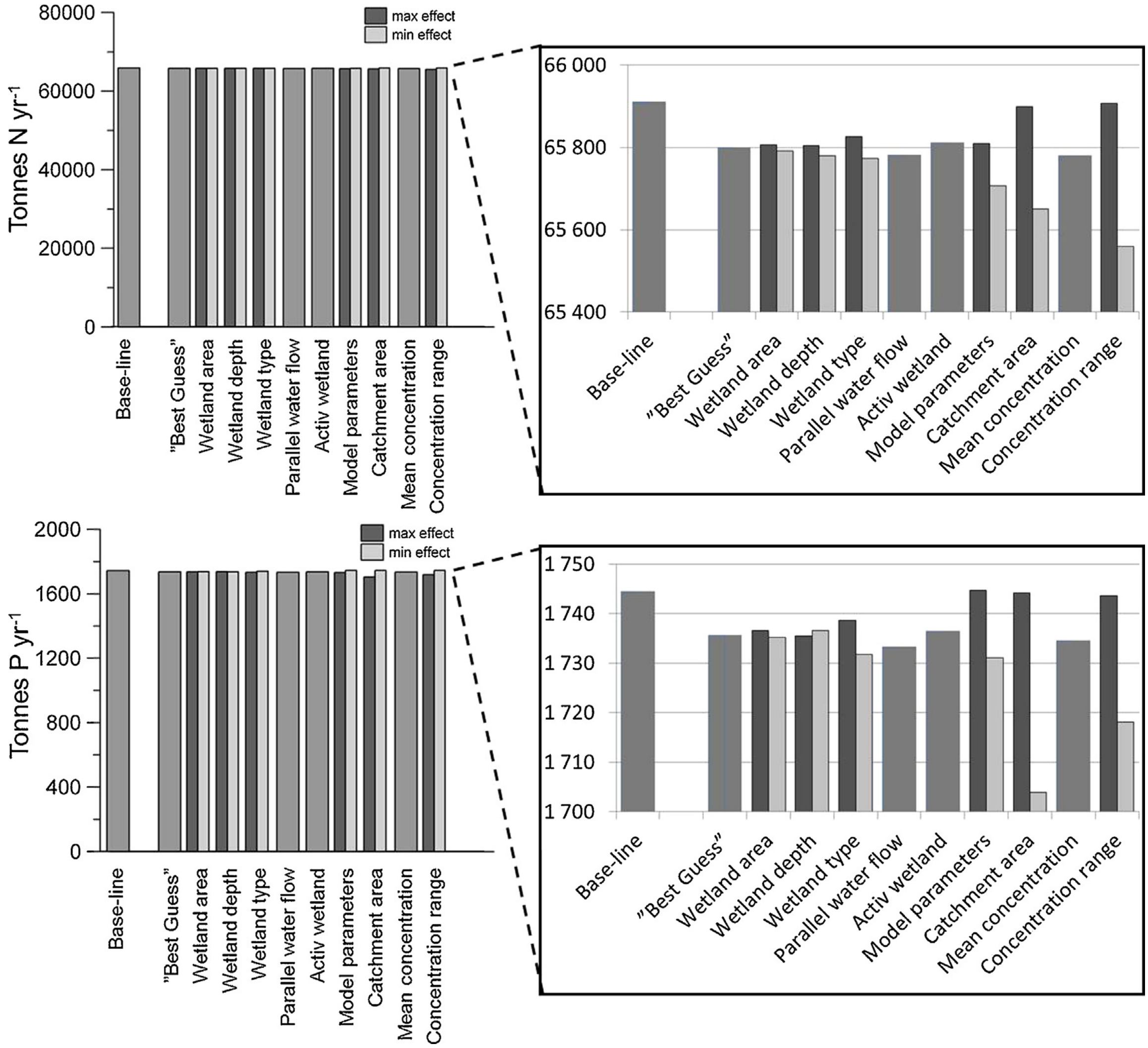

Fig. 9. Nitrogen (left) and phosphorus (right) transport to the sea for the simulations without (base-line) and with wetlands, according to all the sensitivity simulations.

Table 3
Effect of constructed wetlands in relation to total nutrient fluxes for the region.

	PLC5 (Brandt et al., 2008)			Present study
	Total southern Sweden[a]	Agriculture	Surface water	Constructed wetlands
Area (km^2)	164,100	27,720	18,300	41
Part of total area (%)	100%	17%	11%	0.02%
Gross N load ($t\,yr^{-1}$)	91,700	48,000	11,000[b]	
Gross P load ($t\,yr^{-1}$)	2520	800	70[b]	
N removal ($t\,yr^{-1}$)	31,900	9600	22,040	110
P removal ($t\,yr^{-1}$)	1090	0	1090	9

[a] Including drainage to the marine basins: Baltic Proper, Danish straights, Kattegat and Skagerrak (i.e. slightly different to this study).
[b] Atmospheric deposition on lake surfaces. Internal load from sediments are not included.

measures (Arheimer et al., 2005) and there are many other benefits (biodiversity, beauty and hunting) and potential for ecoservices to take into account (e.g. De Steven and Lowrance, 2011; Bostian and Herlihy, 2014). As the wetlands constructed in Sweden between 1996 and 2006 were not optimally allocated, the nutrient reduction was more expensive (to the sea: €700/kgN and €9000/kgP; locally: €600/kgN and €7000/kgP) than in previous studies. The costs for measures to improve the environmental status may seem high, but in fact it is only a minor cost compared to the overall gains from exploiting nature through agriculture, forestry and fishing. Moreover, the removal of algal blooms will in turn benefit industries of tourism and water related ecoservices.

4. Conclusions

Lessons learned from investigating the 1574 wetlands across Sweden and their effect on riverine nutrients during 1996–2006 were:

- The wetlands had minor effect on nutrient transport to the sea (−0.2% for nitrogen and −0.5% for phosphorus). Wetlands could be beneficial for water status locally; however, the implemented wetlands were not allocated to the most optimal areas for nutrient reduction.

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- The model results and conclusion are robust as the wetlands studied were so small compared to the total area and load in the studied region (41 km^2 vs. 164,000 km^2). Substantially larger areas need to be converted to wetlands for them to have any major impact on large-scale nutrient load.
- Most wetland constructions were not accompanied with monitoring programmes, metadata and information on wetland characteristics, which is needed to follow-up on measure programmes and estimate their effects.

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