The Contribution of Marsh Zones to Water Quality in Dutch Shallow Lakes: A Modeling Study

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Abstract Many lakes have experienced a transition from a clear into a turbid state without macrophyte growth due to eutrophication. There are several measures by which nitrogen (N) and phosphorus (P) concentrations in the surface water can be reduced. We used the shallow lake model PCLake to evaluate the effects of three measures (reducing external nutrient loading, increasing relative marsh area, and increasing exchange rate between open water and marsh) on water quality improvement. Furthermore, the contribution of different retention processes was calculated. Settling and burial contributed more to nutrient retention than denitrification. The model runs for a typical shallow lake in The Netherlands showed that after increasing relative marsh area to 50%, total phosphorous (TP) concentration in the surface water was lower than the Maximum Admissible Risk (MAR, a Dutch government water quality standard) level, in contrast to total nitrogen (TN) concentration. The MAR levels could also be achieved by reducing N and P load. However, reduction of nutrient concentrations to MAR levels did not result in a clear lake state with submerged vegetation. Only a combination of a more drastic reduction of the present nutrient loading, in combination with a relatively large marsh cover (approximately 50%) would lead to such a clear state. We therefore concluded that littoral marsh areas can make a small but significant contribution to lake recovery.

Introduction

Eutrophication in shallow lakes is a worldwide problem. Many lakes have experienced a transition from a clear state with abundant macrophyte growth to a turbid state characterized by the excess growth of microscopic algae. Submerged macrophytes can hardly survive under these conditions and the overall ecological quality of the lake ecosystem is poor (e.g., Korner 2001; Irfanullah and Moss 2004). A moderate reduction of nutrient loading often does not lead to a clear state due to internal loading (Sondergaard and others 2003) and resilience of the turbid state (Scheffer 2001). Water quality in the shallow lakes of the Netherlands is still far from meeting ecological standards (Gulati and Van Donk 2002). The objective of Dutch lake managers generally is the restoration of the macrophyte dominated clear-water state.

There are several measures by which nutrient concentrations in the surface water can be reduced. The further reduction of nutrient inputs from sewage treatment plants and from diffuse agricultural sources is the most effective, source-oriented measure. Although this has led to a significant reduction in nutrient loading in the past decades

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(Gulati and Van Donk 2002), even further reduction is possible using 4-step purification and new methods to reduce agricultural run-off (Muscutt and others 1993). Scheffer (2001) already stated that a prerequisite to achieve this is a reduction of the nutrient loading to below a critical value. Another approach is the restoration of littoral marsh zones because of their significant potential for nutrient retention by the marsh vegetation (Coveney and others 2002). Vegetation can take up nutrients directly from the surface water (Coops and others 1996) or indirectly decrease surface water nutrient concentrations by uptake from the soil (Spears and others 2007). This could result in reduced nutrient concentrations in the lake, low enough for a return to a clear water state. Also, increased sedimentation, other physical or chemical processes (Ishida and others 2006; Olde Venterink and others 2006), and bacteria and algae uptake (Jeppesen and others 2005; Noges and others 2003) in the marsh zone may contribute. Artificial marsh zones can be constructed at the lake shore or in shallow open water zones of the lake (Kovacic and others 2006).

The effect of marsh zones will depend on their surface area and the intensity of the exchange of water between the lake and the marsh zone. Another critical parameter for the effectiveness and total area of marsh zones is the water level regime in combination with shore morphometry. When the difference in water level between summer and winter is minimal, only a narrow band of lake shore is alternately flooded and drained, leading to only a narrow zone of shore vegetation. However, when the difference in water level is increased, the total vegetated marsh area may be enlarged several times (Coops and others 2004). The steepness of the shore zone is also of importance (Rea and others 1998). The rate is dependent on the situation of the marsh zone in relation to wind direction, wave action, flow through rate of water in the lake, width and shape of the marsh zone, water level fluctuations, and the presence of dams or other structures blocking free water flow. To our knowledge, no studies exist in which the water exchange between open water and natural littoral zones was measured directly, but many experiments showed the importance of hydraulic retention time for the removal efficiency of (constructed) wetlands (Ishida and others 2006; Jing and others 2002).

Research on the nutrient retention function of littoral zones has often focused on processes within the vegetated zone itself. For a proper evaluation of the functioning of these zones in the context of the nutrient dynamics of the lake as a whole, the nutrient dynamics of the littoral zone have to be considered at a larger scale and linked with the functioning of the open water section of the lake. Dynamic modeling is a suitable approach to achieve insights in the consequences of vegetation for overall lake water quality

(e.g., Jorgensen and Bendoricchio 2001). In this study, we used the ecosystem model PCLake developed for shallow lakes (Janse 1997; Janse and others 2001; Janse 2005). The model simulates the food web and nutrient dynamics in the open water and in a littoral marsh zone, as well as the hydrological exchange between these two lake compartments. We evaluated different management options with the model: increasing the marsh area, increasing the water exchange rate, and reducing nitrogen (N) and phosphorous (P) loadings.

We focused on the following research questions:

What is the effect of changing the lake/marsh ratio, water exchange rate, and nutrient loading on the modeled nutrient concentrations in the open water and the marsh zone?
Which combination of these measures is most effective in reducing modeled nutrient concentrations in the lake surface water?
What is the contribution of burial, denitrification, and settling to modeled nutrient removal, both in summer and winter?

Methods

Model Description

For this research, the existing model PCLake, version 5.08, was used. We applied the implementation of this model in Osiris (Mooij and Boersma 1996), with a user interface through Excel. This interface facilitates simulations with the model through easy data communication and setting conditions for individual simulations. PCLake has been tested and calibrated using data of several lakes in the Netherlands: Lake Loosdrecht (Janse and others 1992), Reeuwijk Lakes (Janse and others 1993), and Lake Zwemlust (Janse and others 1995). In a later stage, a combined calibration was performed on total phosphorous (TP), chlorophyll-*a*, Secchi depth and vegetation coverage for a set of >40 lakes (both turbid and clear) (Janse 2005; see also Aldenberg and others 1995). Data on a number of lakes in other European countries were used for validation (Dagevos and others 2003). For validation, the Bayesian method was used (Aldenberg and others 1995). Although PCLake was calibrated with data of European (in majority Dutch) lakes and is mainly used for Dutch lake calculations, it is probably also applicable to more or less comparable shallow lakes in other countries when input parameters, such as temperature and sediment type, are adjusted. A full description of PCLake, including all processes and equations, is given by Janse (2005). For this study we did not modify PCLake.

PCLake was developed for predicting the critical nutrient loading at which switches between the clear and turbid state occur (Janse 1997). In a later stage, a marsh module was added to model the effect of adjacent zones with emerged vegetation. The initial values that were used for the most important variables concerning vegetation and soil characteristics in the littoral zone were estimated on the basis of values measured in several shallow lakes studies (Table 1). Initial values of the open water section were already described by Janse (2005). Please note that "refractory N" (bound in humic compunds and the like) is not included in the model. The resulting model describes a shallow lake as consisting of two compartments, i.e., the open water (also referred to as the lake) and the marsh zone (Fig. 1a, b, c) (Janse and others 2001).

In order to achieve closed mass balances, all organic components were modeled in multiple units, namely, dry-weight, N and P. The modeled abiotic state variables in the water column are inorganic material, detritus, dissolved phosphorus, ammonium, and nitrate (Fig. 1). The state variables in the sediment are organic and inorganic material, and pore water with dissolved phosphorus, ammonium, and nitrate. The sediments of the open water and marsh zones are always flooded and the depth of the aerobic layer is dependent on the oxygen concentration in the water, the sediment oxygen demand in the soil, and the oxygen diffusion coefficient. The aerobic layer is influencing the rate of denitrification, mineralization, and P adsorption. The sorption of P is reversible, but when P concentration in the pore water exceeds a maximum, the excess of P is irreversibly immobilized. The exchange of nutrients between surface water and pore water is modeled by a diffusion equation. There is also exchange by settling and (in the lake part only) resuspension.

The biotic components in the model are phytoplankton—divided in three functional groups (cyanobacteria, diatoms, and other small algae)—zooplankton, zoobenthos, two types of fish, submerged vegetation (in the lake), and helophytes vegetation (in the marsh zone) (Fig. 1). Algae take up phosphate, ammonium, and nitrate from the surface water to maintain optimal P:biomass and N:biomass ratios, respectively. Ammonium uptake is preferred, but when the ammonium concentration is low, the algae switch to nitrate. Submerged plants (simulating *Elodea*) take up nutrients from both surface water and soil pore water. Like algae, they prefer ammonium. Water plants die off from half September onwards. The biomass left in winter is the starting value for next growing season.

The sediment of the marsh zone is modeled analogous to the sediment of the lake zone. The exchange of water with the nutrients contained therein between the two compartments is based on an exchange coefficient (related to water level fluctuations and dispersive transport) multiplied by the difference in concentration. When the water exchange rate is zero, the compartments, "open water" and "marsh," act as separate compartments without interaction. It is assumed that resuspension in the marsh zone is zero. Phytoplankton growth in the marsh zone is zero because of shadowing by reed, but it can be imported by exchange with the lake water. Reed biomass and nutrient contents are dependent on allocation, production, mortality, respiration, and reallocation. Reed is growing in two stages; the initial stage and the productive stage. Initially, part of the carbon is allocated from the rhizomes to the shoot, together with a proportional amount of N and P. When the stem length is equal to the water level the production of new biomass starts. During this stage carbon is incorporated via photosynthesis. The growth rate is determined by temperature, day length, and nutrient contents of the vegetation (lower growth rate when nutrient contents are low). The absolute amount produced is the product of the growth rate, the

Table 1 Input variables of the littoral zone in PCLake compared to empirical values measured in wetland soils

| Parameter	PCLake	Empirical value	Study
Lake submerged vegetation (g m^{-2})	1 (= no vegetation)		
Marsh helophyte vegetation (aboveground) (g m^{-2})	1500	180–5280	Gopal (1990)
Marsh helophyte vegetation (belowground) (g m^{-2})	7500	700–1000	Vretare and others (2001)
		1500–2500	Van den Wyngaert and others (2003)
Dry weight soil (g g^{-1})	0.3	0.1–0.3	Malmaeus and Rydin (2006)
Lutum (<16 μ) content of sediment (g g^{-1})	0.1		
Organic fraction soil (g g^{-1})	0.1	0.12	Barko and Smart (1981)
		0.02–0.06	Bai and others (2005)
Organic N fraction soil (g g^{-1})	0.025	0.039	Barko and Smart (1981)
Organic P fraction soil (g g^{-1})	0.0025	0.0007	Barko and Smart (1981)
		0.001	Sondergaard and others (2003)
		0.0007	Malmaeus and Rydin (2006)
		0.0028	Bai and others (2005)
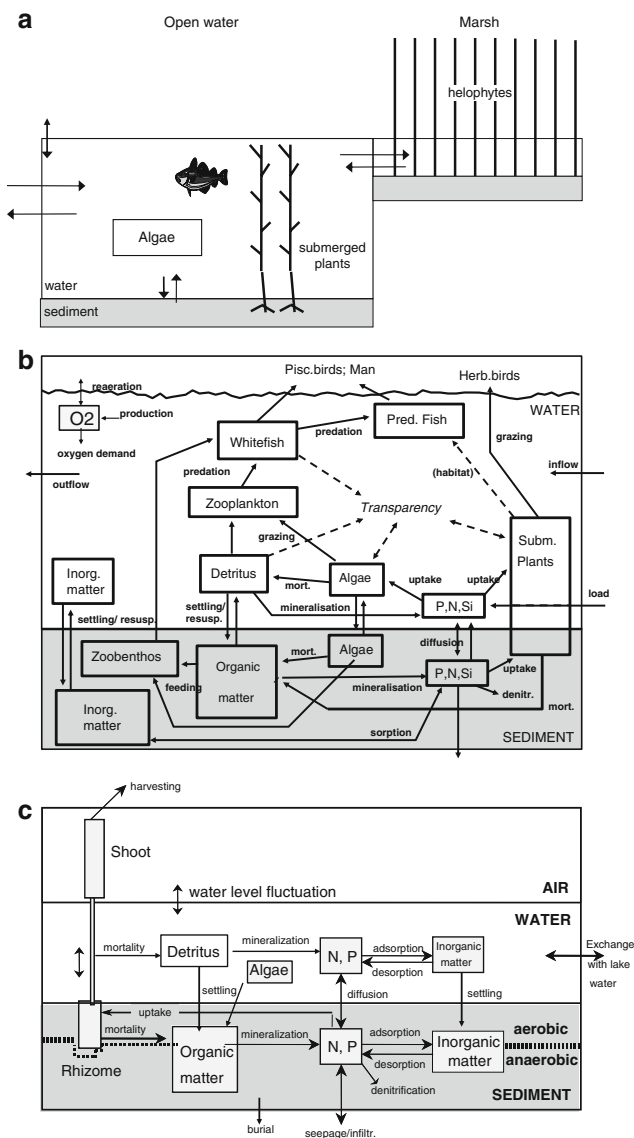
Fig. 1 Schematic picture of the PCLake model. (**a**) general overview; (**b**) open water module; (**c**) marsh module. See text for model explanation. After Janse (2005)

biomass per stem, and the stem density. The amounts of nutrients taken up from the pore water is the product of rhizome biomass and the uptake rate, which is determined by the actual rhizome nutrient content, pore water nutrient concentration, nutrient affinity, and temperature. In autumn, the mortality rate of reed increases. A proportion of the nutrients is retranslocated to the rhizomes. The aboveground biomass which is left after retranslocation is the starting value for the next year.

Lake Characteristics

The main inputs to the model are: water inflow, infiltration or seepage rate, nutrient (N, P) loading, particulate loading, temperature and light, dimensions (lake depth and size),

size of the marsh zone, sediment features, as well as the initial conditions of the state variables. Lake characteristics have been chosen for a hypothetical lake, representative for a small eutrophicated shallow lake in The Netherlands. The lake surface area is 100 ha (small lake) and when a marsh zone is introduced, the area of the total system increases. Water depth of the open water is 3 m and of the marsh zone is 0.3 m. Water enters the system in the lake through an input of 20 mm water day^{-1}. As regards the initial conditions in the lake, the model was started in the "turbid state" with near-absence of submerged vegetation. Initial conditions in the marsh zone comprised a fully developed helophyte vegetation with, on January 1st, 80% of the biomass below-ground. The values of the other input and process parameters are described in Janse (2005).

Scenarios

To address the research questions, three parameters were varied, i.e., (1) marsh surface area, (2) degree of water exchange between the open water and the marsh, and (3) nutrient loading (Table 2). The size of the marsh area relative to that of the open water was varied between 0 (no marsh) and 50% marsh. The exchange coefficient ranged between 0 and 1 m^3 m^{-3} day^{-1}. For nutrient loading, three scenarios were used: present loading, past loading (1975), and 1/10 times the present loading. The present loading represents nutrient loading in lake IJsselmeer and Veluwemeer in 2002 (2 g P m^{-2} y^{-1} and 60 g N m^{-2} y^{-1}) (Fig. 2). For the past loading, values of 1975 were used, resembling the highest loading in lake IJsselmeer of the last decades (6 g P m^{-2} y^{-1} and 100 g N m^{-2} y^{-1}). These loading rates exceed most reported values for restoration of a clear state of shallow lakes (e.g., Jeppesen 1991; Benndorf 1990). The loading rate was assumed to be constant over the year. Because we used empirical data for nutrient loading, the N:P ratio differed between the scenarios. Every combination of parameters values was calculated, resulting in a total of 252 model runs (Table 2).

Output

We focussed on two output variables: total nitrogen (TN) and total phosphorus (TP) (mg l^{-1}) in the water phase. These were calculated as follows:

$$TN = N_{diatoms} + N_{greenalgae} + N_{cyanobacteria} + N_{detritus} + NH_4 + NO_3$$

$$TP = P_{diatoms} + P_{greenalgae} + P_{cyanobacteria} + P_{detritus} + P_{inorganicmaterial} + PO_4$$

The results of the calculations are presented as average values of the TN and TP concentrations in the summer

Table 2 Scenarios varying in marsh surface area, exchange rate, and N/P loading

Parameter	Number of scenarios						
Marsh surface area (%)	21	0	2.5	5	–	45	47.5	50
Exchange rate ($m^3 \ m^{-3} \ day^{-1}$)	4	0	0.25	0.5	1		
Loading ($g \ m^{-2} \ day^{-1}$)	3	N 0.0164	N 0.164	N 0.274			
		P 0.0005	P 0.005	P 0.0164			
		(10% of present)	(present)	(past)			

Total number of model runs: $21 \times 4 \times 3 = 252$

Fig. 2 P and N load in Lake IJsselmeer and in Lake Veluwemeer from 1972–2002 (E. Lammens, D. van der Molen, and R. Rijsdijk personal communication, 2006)
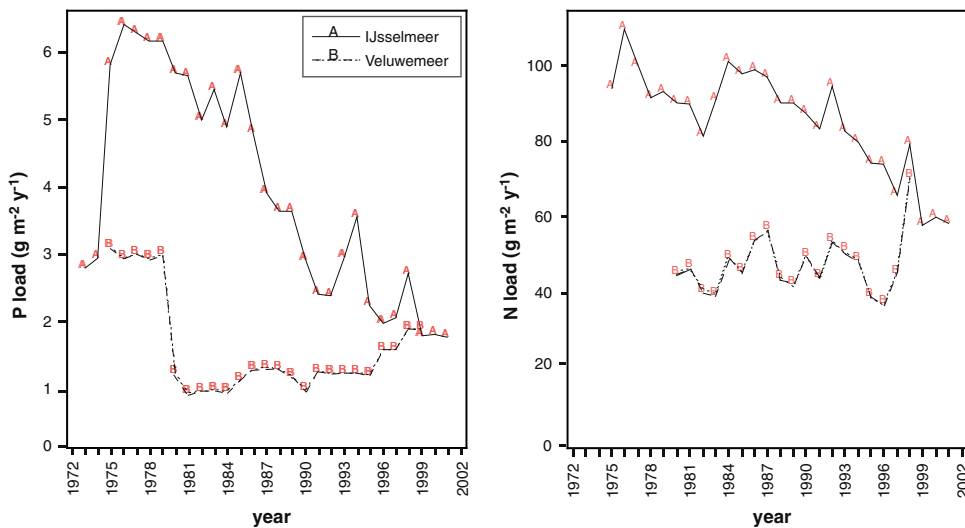
(April 1st to September 30th) of the 15th year after the start of the simulation. Furthermore, when winter averages are shown, these are calculated as the average value of October 1st of the 15th year to March 31st of the 16th year. The 15th year was chosen because reed biomass increased per square meter until year 15, after which equilibrium was established. Also, TN and TP

concentrations in lake and marsh surface water were at equilibrium after 15 years. In Fig. 3 an example is shown for a scenario run in time.

To investigate the importance of different nutrient retention processes, average summer and winter fluxes were calculated. Important for water quality in the lake are the processes which permanently deliver and retain

Fig. 3 Simulation of 4 scenarios (differing in relative marsh area) for TN and TP in the surface water of the open water (left) and in the marsh zone (right) (mg l^{-1}). Summer averages are calculated with data between April 1st and September 30th of the 15th year. N loading was 0.164 g m^{-2} d^{-1} and P loading was 0.005 g m^{-2} d^{-1} (Present loading). Water exchange between open water and marsh was 1 m^3 m^{-3} d^{-1}
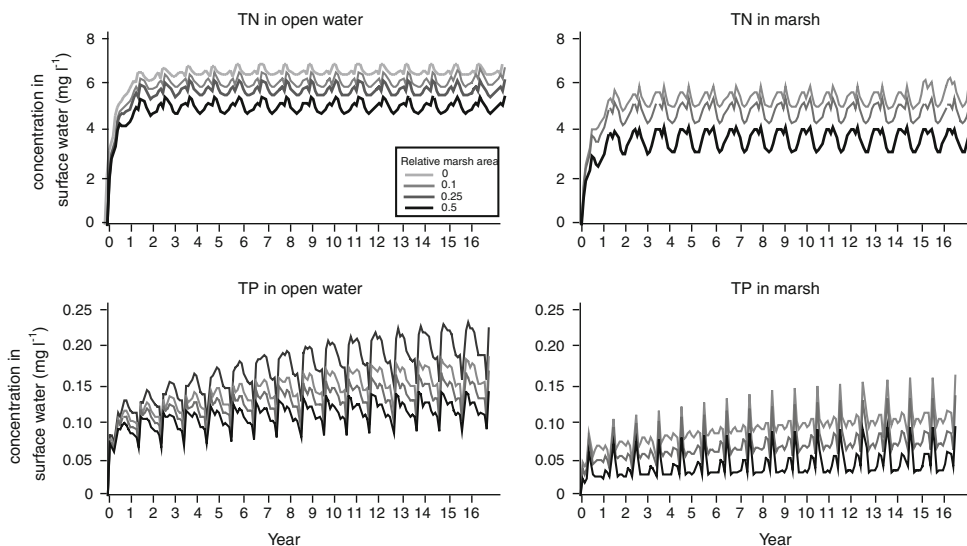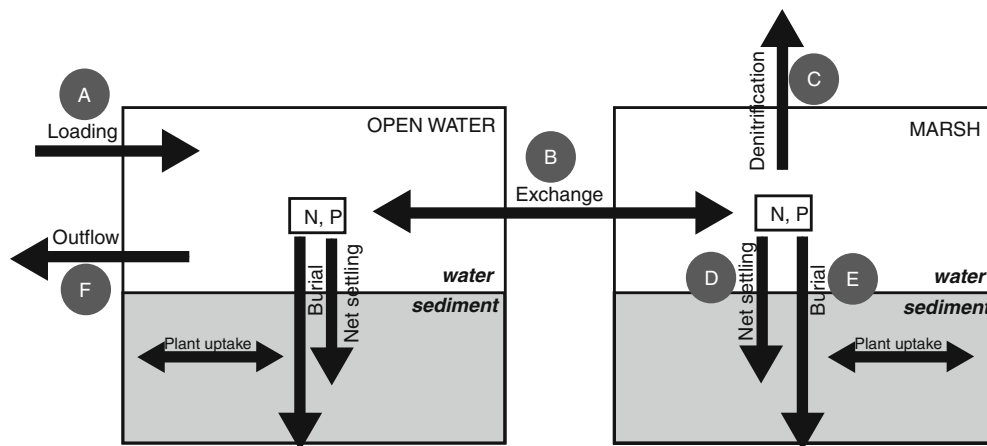
Fig. 4 Scheme of important nutrient retention fluxes of N and P in the lake and the marsh zone. See text for exact calculation of fluxes A–F

nutrients (Fig. 4): (Fig. 4a) the loading of N and P from external sources, (Fig. 4b) the net exchange of nutrients from the lake to the marsh, and (Fig. 4f) the outflow of nutrients. Inside the marsh zone nutrients are permanently removed from the surface water by (Fig. 4c) denitrification, (Fig. 4d) net settling (=settling minus release), and (Fig. 4e) burial (permanent storage of organic matter in deeper soil layers). Flux B is the sum of the exchange of nutrients in phytoplankton, NH_4, NO_3, PO_4, detritus, P adsorbed onto inorganic matter, and zooplankton. Flux D is the sum of settling of phytoplankton, detritus, and adsorbed P, minus the diffusion of dissolved nutrients from sediment to water and the evaporation flux from water to soil (emerged plants evaporate, causing a net flow of water containing nutrients from the soil to the plant roots. This, in turn, causes a net water flow from the surface water to the soil). Flux E is the sum of burial of detritus, humus, NH_4, NO_3, PO_4, and adsorbed P. The nutrients taken up by the vegetation will be returned to the water phase or the sediment after vegetation die-off, so that these fluxes only temporarily remove nutrients. All fluxes are expressed in g m^{-2} d^{-1}.

Results

Water Quality

The model produced predictions for the concentrations of TN and TP in surface water in simulations with different marsh surface area and degree of water exchange for the "present" (Fig. 5a), "past" (Fig. 5b), and "reduced" nutrient loading (Fig. 5c). After 15 years of the present nutrient loading rate, the simulated TN and TP concentrations in the open water were 6.4 mg N l^{-1} and 0.2 mg P l^{-1}, respectively, in the absence of a marsh zone (Fig. 5a). These concentrations steadily decreased with increasing

relative marsh area to 80% and 60% of the original (without marsh) TN and TP concentrations, respectively, at a relative marsh area of 50% and a water exchange rate of 1 m^3 m^{-3} d^{-1}. At this high water exchange rate, nutrient concentrations in the marsh zone were comparable to concentrations in the lake. At a lower exchange rate, less N and P are transported between the open water and the marsh zone, and concentrations in the open water decrease less. Increases of the water exchange rate become less effective the higher this rate already is. The effect of marsh surface area is, as expected, strongest at high rates of water exchange.

The large impact of the much higher nutrient loading in the "past" scenarios is clearly visible in the also much higher TN and TP concentrations (Fig. 5b). Even if the marsh surface area was increased greatly, TN and TP concentrations still would have been higher than present values. The model resulted in similar qualitative responses to changes in marsh area and exchange rate at the "present" and "past" loading scenarios.

In the scenarios with 90% reduced nutrient loading, TN and TP concentrations in the open water in the scenarios without a marsh zone were, predictably, lower. At present nutrient loading, TN and TP concentrations in the lake after 15 years were 6 mg N l^{-1} and 0.2 mg P l^{-1}, respectively (Fig. 5a), while at a ten times lower loading rate these concentrations were approximately also ten times lower (Fig. 5c). In contrast to the past loading rate scenarios, at reduced loading rates (10% of the present rate), TN and TP concentrations respond differently to an increase in relative marsh area than at the present loading rates. The most striking difference is abruptly decreasing concentrations with increasing marsh area at low nutrient loading. As long as marsh area is small, nutrient concentrations in the surface water decrease proportionally with increasing marsh area. However, at an exchange rate of 0.5 m^3 m^{-3} and marsh area covering 25% of the lake, nutrient

Fig. 5 Summer averages of TN (upper graphs) and TP (lower graphs) concentrations (mg l^{-1}) in the open water (left) and the marsh zone (right) at (**a**) present (2002) nutrient loading of 0.164 g N m^{-2} d^{-1} and 0.005 g P m^{-2} d^{-1} and different marsh surface areas and different degrees of water exchange; (**b**) past (1975) nutrient loading of 0.274 g N m^{-2} d^{-1} and 0.0164 g P m^{-2} d^{-1} and different marsh surface areas and different degrees of water exchange; (**c**) 10% of present nutrient loading of 0.0164 g N m^{-2} d^{-1} and 0.0005 g P m^{-2} d^{-1} and different marsh surface areas and different degrees of water exchange
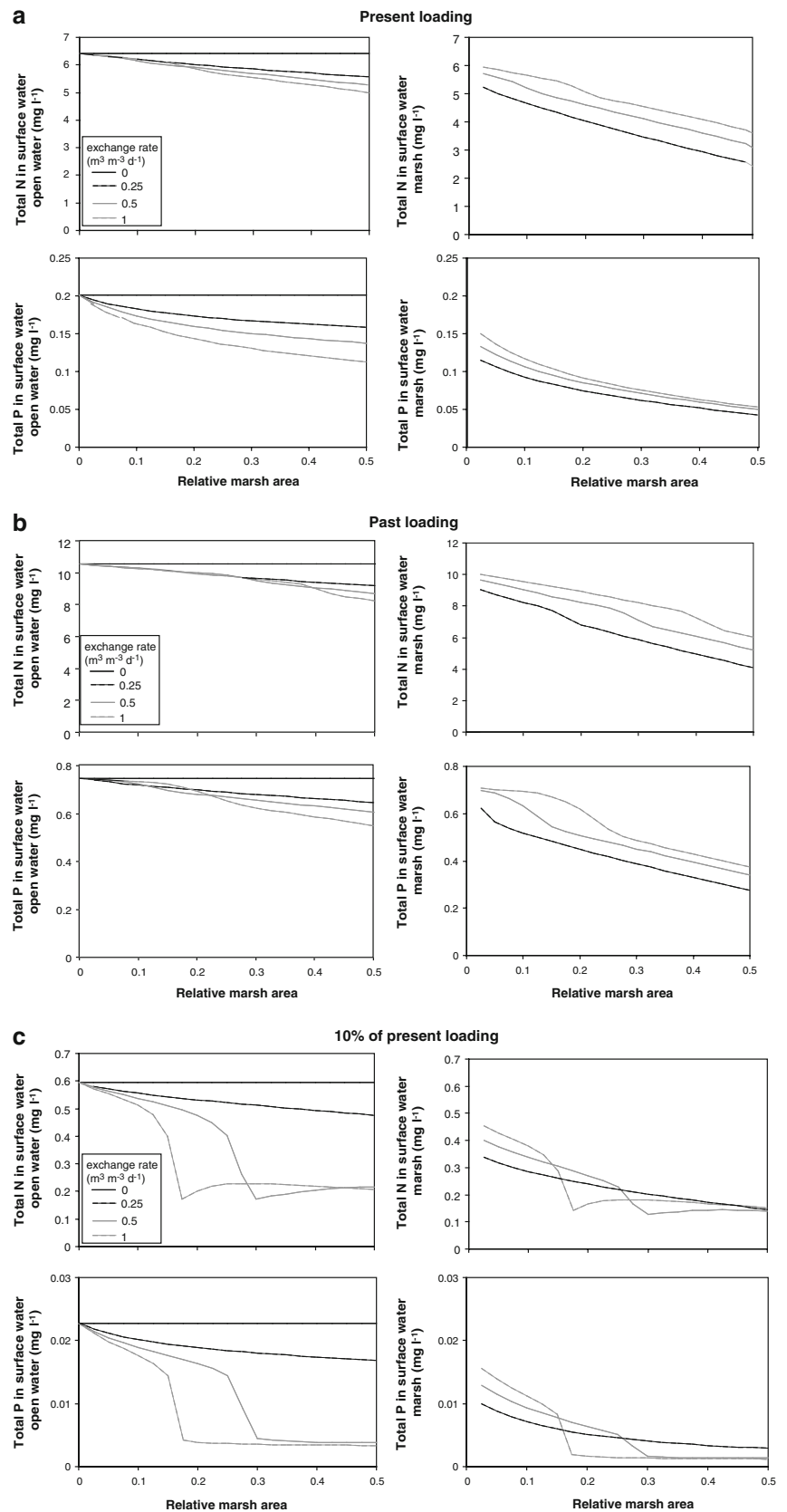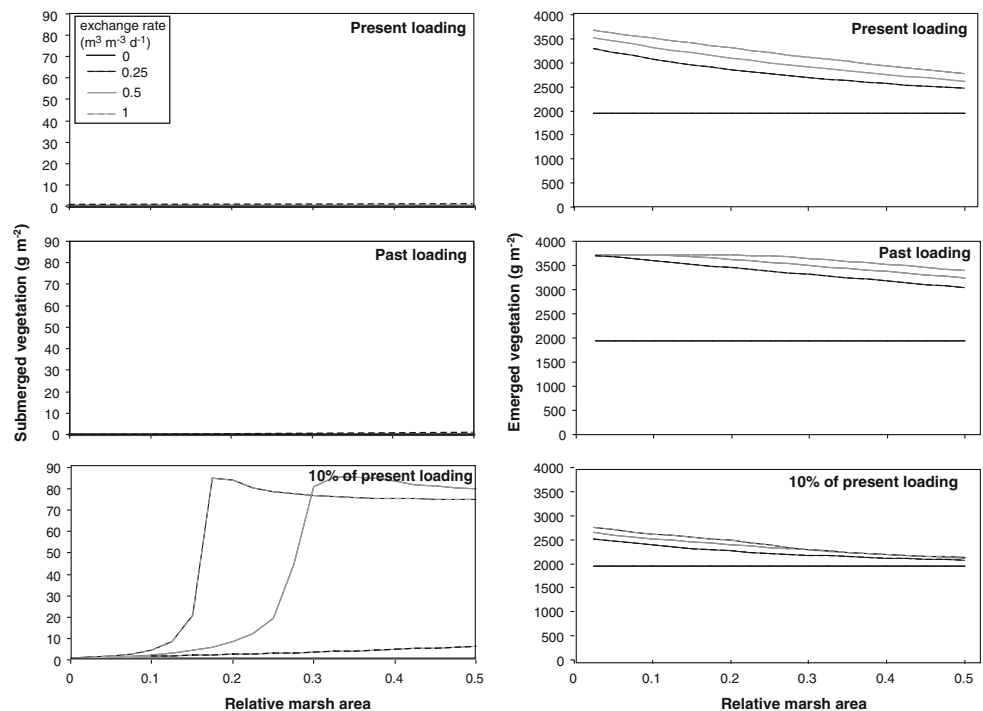
Fig. 6 Summer average of biomass (g m^{-2}) of submerged lake vegetation (left) and emergent marsh vegetation (right) at different loading levels, water exchange rates, and relative marsh areas

concentrations in both the lake and the marsh zone sharply dropped to low values. Further increase of the marsh area had no effect on TN and TP concentrations anymore. At an exchange rate of 1 m^3 m^{-3}, this phenomenon already occurs from marsh areas higher than about 18%. This threshold relationship relates to the appearance of submerged vegetation in the open water (Fig. 6). Because plant uptake in the marsh zone decreased nutrient concentrations in the lake, chlorophyll-*a* levels dropped and light availability at soil surface increased (data not shown). Low nutrient concentrations caused a shift from a turbid to a clear lake system and promoted the establishment of submerged vegetation (Gulati and Van Donk 2002). As a result, the submerged vegetation takes up significant amounts of nutrients. From that moment forward, the efficiency of further marsh zone enlargement is negligible. At higher loading rates, the submerged vegetation was unable to grow and no such shifts are seen for TN and TP in those cases.
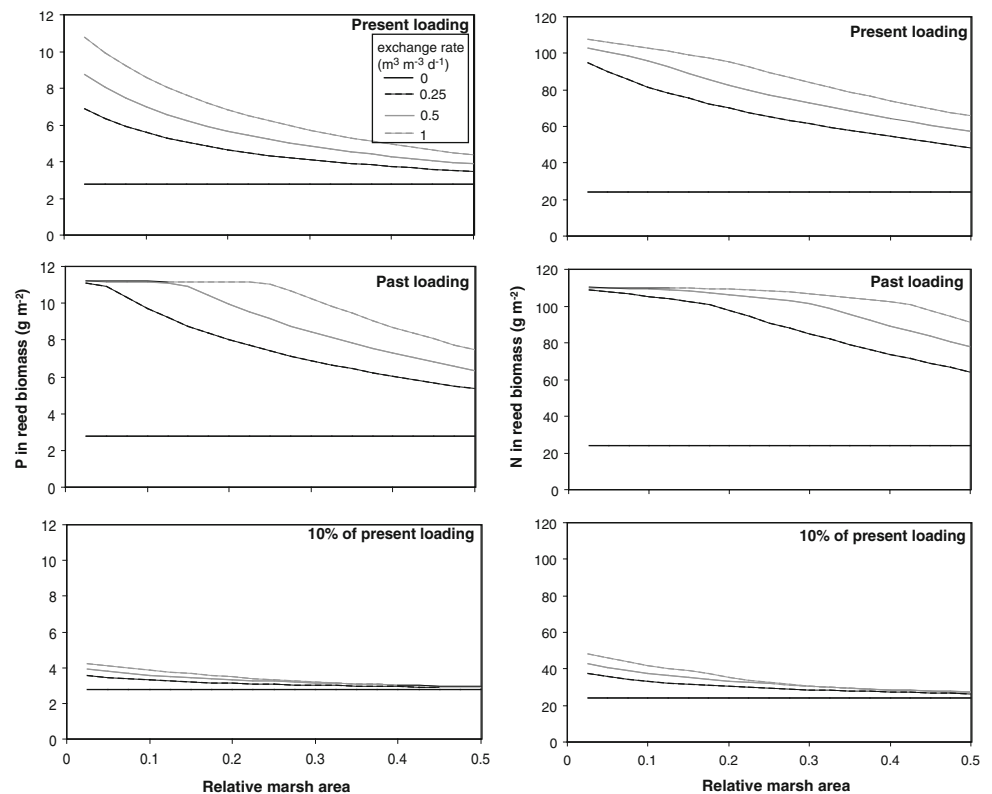
Vegetation

The growth of submerged vegetation in the lake and emergent vegetation in the marsh was strongly dependent on nutrient loading (Fig. 6). At present and past loading rates, no submerged vegetation established due to high phytoplankton densities in the open water, limiting light availability (data not shown). At lower nutrient concentrations, algal densities dropped and the submerged vegetation started to grow to a maximum summer-averaged

biomass of 85 g m^{-2}. Light availability and submerged vegetation biomass were directly related. The growth of the plants was also dependent on the relative marsh to open water area, because high relative marsh areas are able to reduce nutrient concentration in the open water to a level at which submerged vegetation can establish. In the simulations, proportional marsh areas of 0.15 and 0.25 were sufficient to reduce nutrient concentrations to a level enabling submerged vegetation at water exchange rates between open water and marsh of 1 and 0.5 m^3 m^{-3}, respectively. And the same effect will occur at smaller marsh fractions when water exchange is high.

Nutrient loading strongly affected reed biomass (Fig. 6). At high nutrient loading (loading of 1975) biomass of reed was at its maximum of 3800 g m^{-2} compared to 2800 g m^{-2} at 10% of the present loading. With increasing marsh fraction, total reed biomass in the marsh zone increased, but per square meter biomass decreased due to the dilution and intraspecific competition for nutrients. The water exchange rate did not influence marsh reed biomass to a large degree.

In Fig. 7, the nutrients in the reed standing stock are shown for the different nutrient loading rates present. Nitrogen and phosphorus contents in the reed standing stock were influenced by relative marsh area, water exchange rate, and nutrient loading. An increase of nutrient transport from the open water to the marsh zone was reflected in higher reed nutrient contents. Therefore, the contents were positively related to nutrient loading and exchange rate. The contents decreased with relative marsh

Fig. 7 Summer averages of N and P contents (g m^{-2}) in *Phragmites australis* at different loading levels, water exchange rates, and relative marsh area

area due to dilution of available nutrients in the marsh zone. The maximum amounts of nutrients in the standing stock of the reed vegetation were 11.5 g P m^{-2} and 110 g N m^{-2}.

Nutrient Retention Processes

The contribution of the different fluxes is strongly dependent on relative marsh area and on water exchange rate (Fig. 8). Obviously, when the exchange rate is 0, no nutrients are transported from the marsh to the open water and vice versa. In the summer season, the net amount of nutrients transported to the marsh (flux B) varies from 0% to 19% of the input (A) for N and 0–17% for P at an exchange rate of 1 m^3 m^{-3} and dependent on the relative marsh area. The outflow (flux F) of the lake ranges from 40–73% of the input (A). Of the removal processes in the marsh zone, net settling is the most important. Up to 15% of N and 29% of P loading is removed from the water by irreversible settling in the soil (flux D). As a result of an increased nutrient loading, relative burial (flux E) and denitrification (flux C) rates hardly accelerate. For these processes, an increase in relative marsh area and exchange rate is not efficient.

In the winter season, a similar amount of nutrients is transported from the lake to the marsh zone. However, the irreversible removal fluxes in the marsh zone are different from those in the summer season. Denitrification rate in the

marsh zone (flux C) is relatively low, whereas settling (flux D) fluxes still quantitatively contribute to nutrient removal. However, compared to the summer season, burial of nitrogen and phosphorus (flux E) is as important as settling. At most favorable conditions (high exchange rate and high marsh fraction) the burial process in the marsh zone removes 11% of N loading and 37% of P loading in the winter season. Because in winter the burial and settling fluxes are higher than the exchange flux, part of the buried and settled N and P could originate from the marsh area itself (e.g., due to marsh vegetation die-off) in that period.

Discussion

The goal of this modeling study was to analyze the effects of three different management options (enlargement of the littoral zone, increasing water exchange rate between lake and marsh zone, and the reduction of external nutrient loading) for water quality improvement in a shallow lake. We showed that according to the model all three have a positive influence on water quality. Of these measures, reduction in loading rate is the most effective, because this factor has an approximately linear relation with nutrient concentrations in the lake. This result is comparable to the findings of empirical lake TP models (Reckhow and Chapra 1983). The efficacy of water exchange measures is dependent on the present exchange rate. An increase of

Fig. 8 Average nutrient removal fluxes (Flux B–F) as percentage of present nutrient loading (Flux A) in the lake (left figs.) and the marsh zone (right figs.) for N and P in summer and winter. Upper *x*-axis is marsh fraction (% of total area) and lower *x*-axis is exchange rate (m³ m⁻³ d⁻¹). See text and Fig. 4 for further explanation
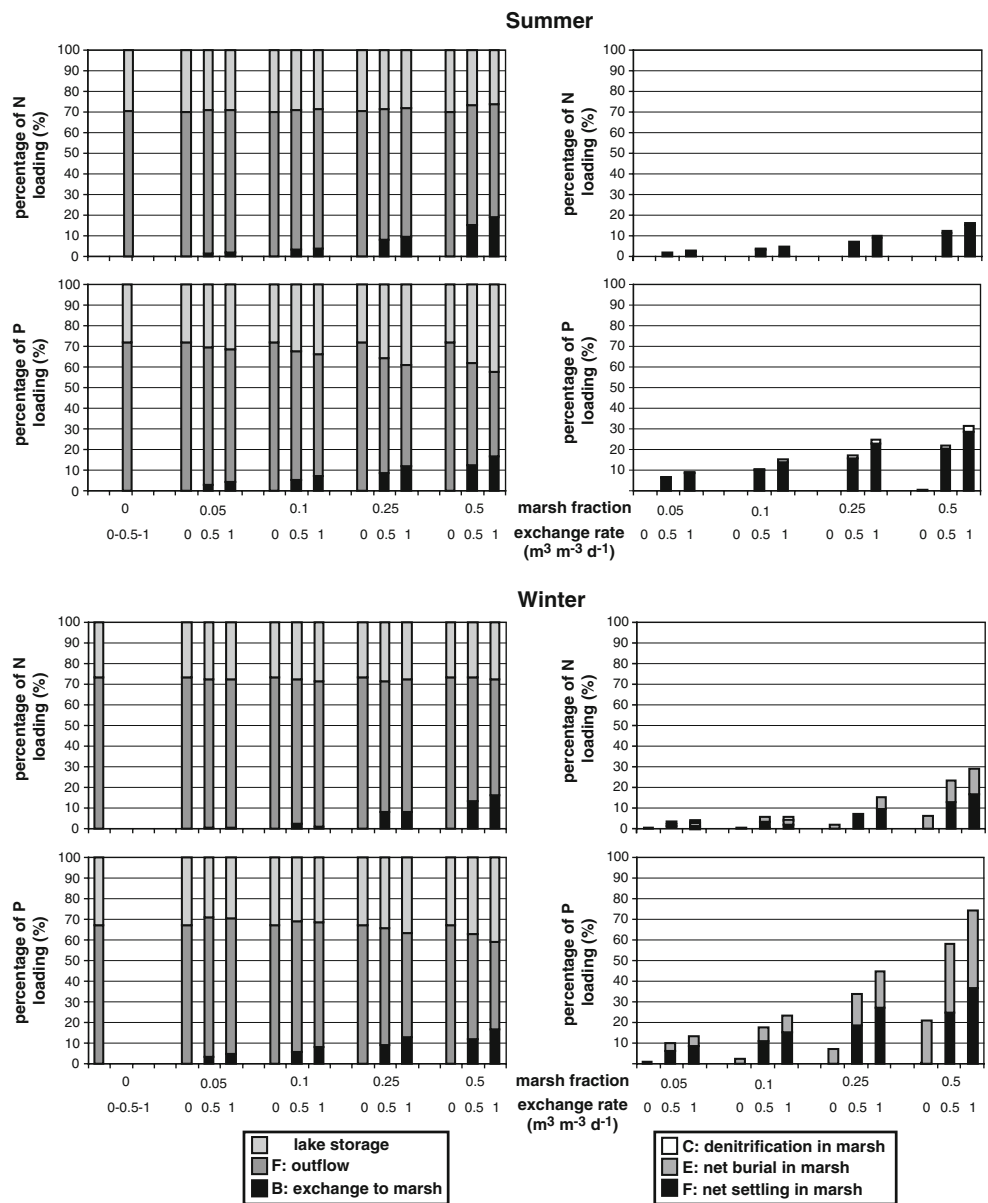
water exchange rate between lake and marsh zone from 0 to 0.5 m³ m⁻³ d⁻¹ is more effective than from 0.5 to 1 m³ m⁻³ d⁻¹. An increase in marsh zone is predicted to potentially decrease nutrient concentrations in the lake due to higher total plant uptake and stimulates the switch to a clear state when nutrient concentrations are low enough so as to allow the development of submerged vegetation in the lake. The simulations show that the optimal method to achieve a clear state is a combination of reduced loading and an increased relative marsh zone.

Model Restrictions

In this study we used PCLake because it allows an analysis of the interaction between open water and marsh and because of its ability to investigate scenarios with different management options. However, there are some restrictions since the model omits some processes that play a role in the natural situation. First, it is known that the growth of reed can be influenced by the seasonal water level regime (Bodensteiner and Gabriel 2003; Coops and others 2004). Water level regimes and hence depth variations could be modeled in PCLake, but the relation between water level, slope, and area of the marsh zone is not part of the model. We have to keep in mind that in reality, large marsh zones only will develop when water levels fluctuate (Coops and others 2004) in contrast to what is modeled in PCLake. Second, natural lakes have a certain littoral slope, which influences the marsh surface area and reed growth (Coops and others 2004; Rea and others 1998). In PCLake, the marsh zone has a horizontal soil surface and no spatial heterogeneity in vegetation, water, and soil is modelled.

Third, depending on their slope in combination with water level regime and/or waves, the upper part of the littoral zones often show wet-dry cycles with various periodicities. PCLake does not model "dry" periods in the littoral zone but rather assumes that this zone is permanently flooded across its full width. Fourth, the exchange rate between lake and marsh zone is modeled in a very simple way, without accounting for influencing factors such as the lake morphometry (e.g., the shoreline to area ratio). Fifth, ecosystem models often contain a great deal of uncertainty caused by the model structure itself or the parameter values. To visualize this problem, Janse (2005) carried out a sensitivity and uncertainty analysis for PCLake, concluding an uncertainty in critical loading at least a factor 2 due to variability in the most important process parameters. Another important conclusion was that the model is very sensitive to lake features, such as water depth, water inflow, and fetch.

These restrictions are important when drawing conclusions on model results. As the model was calibrated with values which are representative for the lakes in the IJsselmeer area, the model is expected to generate outcomes which can be used for scenario evaluation with respect to water quality issues in that area.

Nutrient Retention Processes

In our simulation, around 70% of the nutrients flowing into the lake are flowing out with discharging surface water for all calculated scenarios. When no marsh zone is present, the other 30% of the nutrients will remain in the open water section of the lake system in vegetation, algae, or sediment. At a small marsh surface area (5%), only a small fraction of the nutrient input is transported to the marsh zone, ranging from 0–5%. This fraction increases rapidly to 16–19% at a marsh surface area of 50% and an exchange rate of $1 \ m^3 \ m^{-3} \ d^{-1}$. Obviously, in total, more nutrients are retained in large marsh zones compared to smaller marsh zones. However, the amount of nutrients stored per square meter becomes less at increased marsh sizes. The maximum retention efficiency per unit of marsh area was found at a marsh cover of 5% of the total lake area.

There are several processes responsible for permanent or temporary removal of nutrients. In the marsh zone, nutrients are temporarily incorporated in the reed vegetation during the growing season, but a major proportion will return to the system after reed die-off (Asaeda and others 2002). Other processes which remove nutrients temporarily are diffusion and adsorption.

Permanent reduction of nutrients in the lake system is caused by denitrification, net settling, and burial of organic matter. The denitrification process transforms nitrate into nitrogen gas (Patrick and Reddy 1976) and has been demonstrated to contribute significantly to nutrient retention in wetlands (Saunders and Kalf 2001; Bachand and Horne 2000; Olsen and Anderson 1994). Olde Venterink and others (2003) found a rate of 55 mg N m^{-2} d^{-1} in a flooded reed bed of a Dutch floodplain, while Toet and others (2003) measured that 0.5–26 mg N m^{-2} d^{-1} was removed from sediment in a system of flow trough ditches polishing the effluent of a sewage treatment plant. In our model, values up to 8 mg N m^{-2} d^{-1} were generated, comparable to empirical data of sediment denitrification. In PCLake, denitrification rate in only the sediment was estimated. Toet and others (2003) and Sirivedhin and Gray (2006) showed the importance of denitrification in periphyton with higher values than measured in sediment in constructed wetlands.

Settling and resuspension rates are strongly dependent on vegetation presence and biomass (Li and others 2008). In our study, maximum net settling rates calculated in the model were 0.03 g N m^{-2} d^{-1} and 0.002 g P m^{-2} d^{-1}. Plunkte and Kozerski (2003) measured sedimentation rates in vegetated stands of 64–99 g DW m^{-2} d^{-1}. When using N and P contents in the soil of 5 mg N g^{-1} and 0.5 mg P g^{-1}, this would correspond to a gross settling rate of 0.3–0.5 g N m^{-2} d^{-1} and 0.03–0.05 g P m^{-2} d^{-1}. These results are not comparable to the rates from our study because we consider the *net* flux from water to sediment.

Finally, in PCLake burial is the loss of a small layer of sediment as a compensation for sediment thickening, as the model assumes that only the sediment top (10 cm) layer actively takes part in the nutrient cycling. The contribution of burial to nutrient removal is higher in winter than in summer, since in winter organic matter dies off and is deposited on the soil surface. Brenner and others (2006) also found that aquatic plants promoted burial, since they remove nutrients effectively from lake waters and transport them to the sediment pool. A lower burial rate in summer than in winter is caused by the fact that mineralization rate is positively related to temperature (Aerts and others 2006) and therefore fewer nutrients are left to be buried as organic matter.

Water Quality Related to Management

TN and TP concentrations in the surface water of aquatic ecosystems are often used as an indication of water quality (Tsegaye and others 2006). Sources of surface water nitrogen and phosphorus are inflow from rivers, agricultural runoff (Muscutt and others 1993), internal eutrophication (Smolders and others 2006), and atmospheric deposition (Jassby and others 1994). The Dutch Maximum Admissible Risk levels (MAR) for shallow lakes are 2.2 mg N l^{-1} and 0.150 mg P l^{-1} and target levels are even lower with 1.0 mg N l^{-1} and 0.050 mg P l^{-1} (RIVM

2000). These MAR values correspond to chlorophyll-*a* concentration of maximum 0.1 mg l^{-1}. Nutrient concentrations in the lake under study are far above these levels. It is generally known that at high nutrient concentrations, lakes may switch to a turbid state and submerged macrophytes disappear (Hilt and others 2006; Jackson 2003). A clear water state may return by measures like nutrient reduction (Anderson and others 2005; Köhler and others 2005) and biomanipulation (Kasprzak and others 2002; Tatrai and others 2005). In accordance to many lake studies (e.g., Jackson 2003; Scheffer 2001), in our model we also have shown the negative correlation between nutrient concentrations and submerged vegetation biomass. This correlation is caused by the increase of chlorophyll-*a* concentrations (indicating phytoplankton growth and therefore a reduction of light availability) at high loading rates (Jackson 2003).

Three different management options were chosen to evaluate their effect on water quality. First, the exchange rate of surface water between the marsh zone and the lake was modeled between 0 and 1 m^3 m^{-3} d^{-1}. These values were chosen on the basis of expert judgment, since no references were found on this subject. The exchange rate implicitly contains the hydraulic detention time, which is often mentioned as an important factor in nutrient retention efficiency (Ishida and others 2006; Jing and others 2002). The results show that above 0.5 m^3 m^{-3} d^{-1}, a further increase of the exchange rate has only minor effect.

As a second management option, the range of relative marsh surface area was varied between 0 and 50%. The potential surface area for emergent plant growth in a shallow lake is controlled by factors such as lake morphometry and water level (Coops and others 2004). The actual percentage of that potential area which will be covered, is dependent on (local) environmental conditions, e.g., nutrient availability (Clevering 1998), grazing (Van den Wyngaert and others 2003), and wave action (Coops and Van der Velde 1996). In most lakes, large helophyte zones are not feasible due to other lake functions concerning navigation, recreation and water storage. In addition, in many cases, lake morphometry limits helophyte growth to a narrow band along the lake shore (Duarte and Kalff 1986). Little research has been done on quantifying the surface areas of emergent helophyte fractions in these lakes. More is known about submerged macrophytes. The vegetated fraction of the open water part differs strongly per lake and also between years. Scheffer and others (1992) showed the variability of *Potamogeton pectinatus* abundance, ranging from 10 to 50% in some lakes of the IJsselmeer area between 1969 and 1989. For nutrient retention, not only the percentage of vegetated marsh area is important, but biomass of the vegetation will mainly determine nutrient contents in the standing stock. Biomass

of the reed vegetation in our simulations was with 3500 g m^{-2} rather high in the typical range of reed biomass in lake littoral zones (Gopal 1990). The plant nutrient concentrations of 28.5 mg N and 2.3 mg P per gram dry weight were also high, but still in the range of field measurements for *Phragmites australis* (Gusewell and Koerselman 2002).

The third management option under investigation was changing nutrient loading, for which empirical data for past, present, and future situations in the IJsselmeer area were chosen.

As expected, all evaluated measures influenced the TN and TP concentrations, but were to a quite different extent. By increasing the relative marsh surface area from 0 to 50% and optimizing water exchange to 1 m^3 m^{-3} d^{-1}, TN concentration decreased by 20% from 6.4 to 5 mg N l^{-1} and TP concentration decreased by 45% from 0.20 to 0.11 mg P l^{-1}. After this measure, simulated TN concentration still did not meet the MAR level, while simulated TP concentration did. The reduction of external nutrient loading has a stronger effect on simulated water quality than the other measures. The scenarios showed that at a 90% reduction of the present loading, TN and TP concentrations also decreased by 90%, implying a near-linear relationship. Interpolating, the MAR level can be achieved at a loading reduction of 65% for N and 25% for P, even without extra measures (i.e. without marsh zone). Although TN and TP concentrations were reduced below the MAR level, at either 50% relative marsh area or a nutrient loading reduction of 90%, the lake was still in a turbid state without submerged vegetation.

When nutrient concentrations in the open water reached levels below approximately 0.4 mg N l^{-1} and 0.015 mg P l^{-1}, light availability was high enough for submerged vegetation to grow exponentially. At the present loading rate, biomass of the submerged vegetation (modeled according to *Elodea* characteristics) was negligible, but the model calculated a maximum biomass of 90 g m^{-2} at reduced loading levels in combination with a marsh surface area of 15% or higher. This maximum biomass is comparable to empirical results of the study of Nagasaka (2004), in which *Elodea nuttallii* biomass ranged between 20 and 150 g m^{-2} in a shallow littoral zone. Immediately after significant growth of the submerged vegetation, in the model chlorophyll-*a* levels dropped from 0.057 mg l^{-1} to 0.002 mg l^{-1}, due to a positive feedback loop of improved light conditions, improved plant growth, reduced nutrient levels, decreased algal biomass, improved light conditions and so on. The level of 0.057 mg l^{-1} is in the range of levels found in many turbid shallow lakes in Europe (Jeppesen and others 2005; Noges and others 2003). The low value of 0.002 mg l^{-1} is, like in our simulations, often corresponding with a clear state (Jackson 2003). This "threshold" P concentration is low compared to values derived from data on some natural shallow lakes (e.g., Jeppesen and others 1991;
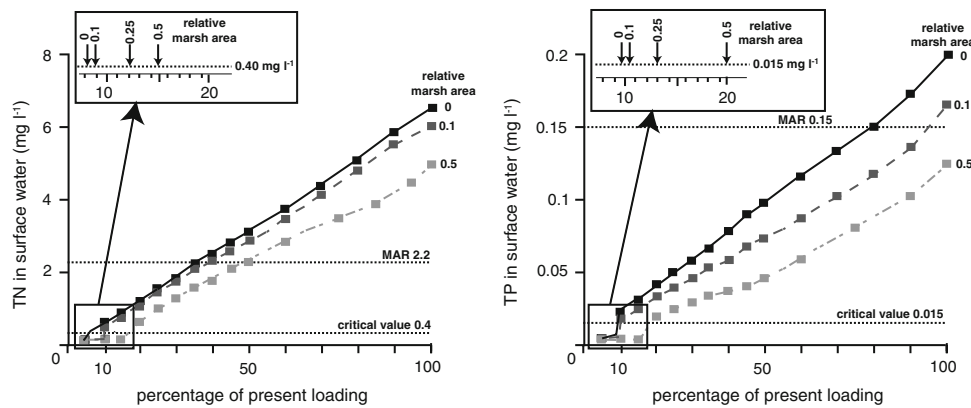
Fig. 9 Effect of nutrient loading reduction (*x*-axis) and relative marsh area (different lines) on TN (left) and TP (right) concentrations in the surface water in the open water. The lower dotted line represents the critical concentrations for the switch from turbid to a clear lake and the upper dotted line represents the MAR

concentrations. Magnified is the most left part of the dotted line. The arrows indicate at which percentage of the present loading four different marsh areas cross the critical value. Exchange rate between open water and marsh zone is 1 m^3 m^{-3} d^{-1}

Jeppesen 1997; Hosper 1997). Earlier studies with PCLake revealed that the calculated threshold values highly depend on lake features, especially water depth (Janse 2005). At a lake depth of 3 m, as used in this study, the calculated threshold is considerably lower than at, for instance, 2 m. The model might also underestimate the threshold value due to the absence of depth variation in the model, in contrast to most natural lakes. Since we coupled N and P loading, in this study we did not measure effects of a reduction of N or P separately. Many authors (e.g., Jeppesen and others 2005), found that especially P has a strong influence on phytoplankton growth in freshwaters.

Management Implications

Three different management options have been discussed in the previous paragraphs. This evaluation can be useful for lake managers to get insight into the processes playing a role in water quality improvement. We used a hypothetical turbid lake of 100 ha to simulate a small eutrophic lake. Calculations with PCLake on larger lakes (Lake IJsselmeer is 115,000 ha and Lake Veluwemeer 3000 ha) show the same response of management on surface water nutrient concentrations as shown in this study. However, it takes more time before the switch from a turbid lake to a clear lake takes place. In large lakes, wind fetch is generally larger, resulting in more waves and consequently a lower settling rate. Submerged vegetation will not establish within decades. Since wind fetch has a strong influence on settling rates and individual lakes differ in shape, depth, presence of islands, etc., the effects of management should be evaluated for every single lake, with model settings adjusted to specific features of the lake.

We showed that reduction in nutrient loading or increasing the marsh area alone is not sufficient to bring the turbid lake into a clear state. Our analysis with the ecosystem model PCLake showed that the switch to a clear lake only occurs when N and P concentrations decrease to below the critical values (0.4 mg N l^{-1} and 0.015 mg P l^{-1}, for the lake settings used). Different combinations of nutrient load reduction and littoral marsh construction may reduce nutrient concentrations to the critical value (Fig. 9). It is obvious that there would be hardly any difference in the costs involved in reducing present nutrient loading by 85% (with 50% marsh) and reducing it by 90–92% (without marsh) to reach the critical nutrient concentration levels. While a reduction of the external loading is absolutely necessary to create and/or maintain a clear lake state, a littoral zone only plays a significant role when the relative surface area is very large. For optimal functioning of marsh zones in improving water quality, enhancement of the water exchange rate is important. The exchange rate can be optimized by, for example, creating marsh zones in parts of the lakes where water flow is highest or by realizing a bendy shoreline. From the modeling study it becomes clear that littoral zones alone are not sufficient for water quality improvement at a whole lake scale, but that in combination with other measures, they can cause the desired switch from a turbid to a clear lake system.

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