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# Seasonal response of nutrients to reduced phosphorus loading in 12 Danish lakes

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## SUMMARY

1. Concentrations of phosphorus, nitrogen and silica and alkalinity were monitored in eight shallow and four deep Danish lakes for 13 years following a phosphorus loading reduction. The aim was to elucidate the seasonal changes in nutrient concentrations during recovery. Samples were taken biweekly during summer and monthly during winter.
2. Overall, the most substantive changes in lake water concentrations were seen in the early phase of recovery. However, phosphorus continued to decline during summer as long as 10 years after the loading reduction, indicating a significant, albeit slow, decline in internal loading.
3. Shallow and deep lakes responded differently to reduced loading. In shallow lakes the internal phosphorus release declined significantly in spring, early summer and autumn, and only non-significantly so in July and August. In contrast, in deep lakes the largest reduction occurred from May to August. This difference may reflect the much stronger benthic pelagic-coupling and the lack of stratification in shallow lakes.
4. Nitrogen only showed minor changes during the recovery period, while alkalinity increased in late summer, probably conditioned by the reduced primary production, as also indicated by the lower pH. Silica tended to decline in winter and spring during the study period, probably reflecting a reduced release of silica from the sediment because of enhanced uptake by benthic diatoms following the improved water transparency.
5. These results clearly indicate that internal loading of phosphorus can delay lake recovery for many years after phosphorus loading reduction, and that lake morphometry (i.e. deep versus shallow basins) influences the patterns of change in nutrient concentrations on both a seasonal and interannual basis.

*Keywords*: alkalinity, internal loading, lake recovery, nitrogen, silica, Water Framework Directive

## Introduction

After decades with continually increasing nutrient loading, many lakes now receive less nutrients because of large investments in improving wastewater treatment combined with the implementation of other measures to reduce, in particular, the phosphorus input (Güde, Rossknecht & Wagner, 1998; Jeppesen *et al.*, 1999; Gulati & van Donk, 2002). In spite of these efforts, many lakes are still eutrophic and exhibit an unsatisfactory water quality. The reason may be delayed recovery caused by, for example, a fish community dominated by zooplanktivorous species (Benndorf, 1990; Jeppesen *et al.*, 1990; Hansson *et al.*, 1998), but also continued release of phosphorus from the sediment may play a role (Marsden, 1989; Jeppesen *et al.*, 1991; Granéli, 1999; Søndergaard, Jensen & Jeppesen, 2001). In eutrophic shallow lakes the influence of internal loading varies considerably over the season, phosphorus concentrations in summer often being much higher than in winter because of

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net release of phosphorus from the sediment (Osborne & Phillips, 1978; Søndergaard, Jensen & Jeppesen, 1999; Willander & Persson, 2001). Thus, summer phosphorus concentrations are greatly controlled by internal processes (Ramm & Scheps, 1997; Kozerski & Kleeberg, 1998; Søndergaard *et al*., 2001).

Concentrations of nutrients other than phosphorus influencing lake water quality may vary after reduced phosphorus loading. In some lakes silica is believed to play an important role in regulating planktonic communities because of its often critical role for diatom growth (Chen *et al*., 2002). In Lake Michigan, for example, a reduction in phosphorus loading led to increased Si concentrations, affecting both diatom biomass and species composition (Barbiero *et al*., 2002). In some lakes or during part of the season the nitrogen : phosphorus (N : P) ratio and nitrogen limitation may change owing to the fact that phosphorus concentrations have often been reduced to levels lower than those of nitrogen, which may influence both phytoplankton and zooplankton communities (Stemberger & Miller, 1999; Smith, 2001; Elser *et al*., 2000; Kilinc & Moss, 2002). Particularly in lakes with a long hydraulic retention time, the internal recycling of nutrients may significantly affect seasonal phytoplankton growth (Schelske, 1985; Kilinc & Moss, 2002). In contrast, lakes having a short retention time may respond rapidly, although rapid responses may not occur when past loading was high (Jeppesen *et al*., 1991).

Most long-term studies provide only little information on the seasonal response pattern of nutrients after a reduction of phosphorus loading. Such information is important, however, for predicting the expected recovery pattern of lakes and is thus vital to lake managers.

In this study we analyse how the seasonality in nutrient concentrations in a series of Danish lakes responds to reduced phosphorus loading with the aim of assessing the seasonal recovery pattern. The analysis is based on eight shallow and four summer-stratified lakes to which the external loading of phosphorus has been reduced significantly. A companion paper elucidates the biological changes occurring in the eight shallow lakes (Jeppesen *et al*., 2005).

## Methods

Seasonal changes in nutrient concentrations following a significant reduction of phosphorus loading were followed for 13 years (1989–2001) in 12 Danish lakes. Eight of the lakes are shallow and non-stratified with a mean depth below 5 m, and four lakes are deeper and dimictic with a mean depth above 8 m (Table 1). The eight shallow lakes are Ørn, Bryrup, Søgård in Jutland; Gundsømagle, Arresø, Damhus, Bagsværd on Zealand; and Vesterborg on Lolland. The four deep lakes are Ravn in Jutland and Fure, Tissø and Tystrup (no loading data available for the last) on Zealand. All lakes are relatively small and have a low hydraulic retention time (Table 1). Submerged macrophytes are either absent or coverage is <10% of the lake area in all lakes, except for one shallow lake where coverage occasionally exceeds 50%. All lakes are included in the 'Danish Nationwide Monitoring Programme on Lakes', which ensures the use of comparable sampling techniques and analyses (see Kronvang *et al*., 1993).

**Table 1** Morphometric characteristics and hydraulic retention time ($t_w$) of eight shallow and four deep Danish lakes

| Parameter | Lake type | Mean | Median | Min | Max |
|---|---|---|---|---|---|
| Mean depth (m) | Shallow | 2.4 | 1.7 | 1.2 | 4.6 |
| | Deep | 12.4 | 12.5 | 8.2 | 16.5 |
| Max depth (m) | Shallow | 4.8 | 3.1 | 1.9 | 10.5 |
| | Deep | 26.5 | 27.4 | 13.5 | 37.7 |
| Area (ha) | Shallow | 539 | 40 | 21 | 3987 |
| | Deep | 704 | 701 | 182 | 1233 |
| $t_w$ (years) | Shallow | 0.65 | 0.12 | 0.05 | 2.2 |
| | Deep | 2.5 | 1.6 | 0.24 | 6.6 |

Water was sampled biweekly during summer and once a month during winter as a pooled sample from the photic zone. The sampling was conducted by the local county authorities according to the standardised sampling protocols of the monitoring programme (Kronvang *et al*., 1993).

Total phosphorus (TP) and total nitrogen (TN) were analysed after persulphate digestion as dissolved inorganic phosphate. Dissolved inorganic phosphate ($PO_4$), nitrate + nitrate ($NO_3$) and ammonia ($NH_4$) were measured on filtered (Whatman GF/C, average pore size 1.2 μm) samples and analysed using standard procedures (Søndergaard, Jeppesen & Jensen, 2000). Silica ($SiO_2$) was analysed spectrophotometrically after heteropolymolybdate formation and total alkalinity (TA) was determined by titration with 0.1 M HCl. Chlorophyll *a* (chl *a*) was determined spectrophotometrically after ethanol extraction on GF/C filters. Data on pH are unavailable for 4 years in two lakes. Nutrient

**Table 2** Changes in mean TP and TN loadings and concentrations in the inlets of eight shallow and four deep Danish lakes during three periods following reduction of phosphorus loading

| | TP inlet (mg $L^{-1}$) | | TP load (mg $m^{-2}$ $day^{-1}$) | | TN inlet (mg $L^{-1}$) | | TN load (mg $m^{-2}$ $day^{-1}$) | |
|---|---|---|---|---|---|---|---|---|
| Period | Shallow | Deep | Shallow | Deep | Shallow | Deep | Shallow | Deep |
| 1989–92 | 0.556 | 0.273 | 22.8 | 9.3 | 7.65 | 8 | 521 | 347 |
| 1993–97 | 0.173 | 0.145 | 11.2 | 5.2 | 5.75 | 5.98 | 522 | 295 |
| 1998–2001 | 0.126 | 0.117 | 12.4 | 4.9 | 5.31 | 5.93 | 537 | 297 |

loading data were obtained as described in Søndergaard *et al.* (2002).

The data were divided into three sampling periods representing comparable temperature conditions without significant differences (Jeppesen *et al.*, 2005) and three stages of recovery; period 1: 1989–92 when TP loading was still relatively high but on the decline; period 2: 1993–97 when TP loading had reached a plateau; and period 3: 1998–2001 when only minor changes occurred in phosphorus and nitrogen loading relative to the previous periods (Table 2). These data are presented in box plots showing minimum and maximum values and, 25 and 75% quartiles for each month of the three periods for both shallow and deep lakes. Statistical tests were performed on log-transformed data on medians for each month of the 13 years using the general linear models procedure of SAS version 8 (proc GLM, SAS, 1989).

## Results

### *Nutrient loading*

During the study period, which included the late phase of external loading reduction and the early recovery phase (Jeppesen, Jensen & Søndergaard, 2002; Søndergaard *et al.*, 2002), mean phosphorus concentrations in the inlets to the 12 lakes decreased from 0.56 to 0.13 mg P $L^{-1}$ in the shallow lakes and from 0.27 to 0.12 mg P $L^{-1}$ in the deep lakes (Table 2). The largest reduction occurred in the early 1990s. Mean inlet concentrations of TN only decreased from 7.7 to 5.3 mg N $L^{-1}$ in the shallow lakes and from 8.0 to 5.9 mg N $L^{-1}$ in the deep lakes.

From period 1 (1989–92) to 3 (1998–2001), TP inlet concentrations and loading to both shallow and deep lakes declined throughout the year, least noticeably in autumn and most markedly in lakes with a high TP loading (75% quartiles; Fig. 1; Table 2). The TN reduction was much less pronounced, and only in deep lakes and for median loadings did a substantive reduction occur during part of the summer. Secchi depth was almost unchanged in both shallow and deep lakes during the study period, whereas chlorophyll in shallow lakes exhibited reduced values during the whole season. Reductions were only significant, however, in spring and early summer (Fig. 2; Table 3).

### *Response in 12 lakes with reduced TP loading*

Lake water TP was reduced throughout the year; in the deep lakes primarily and significantly so from May to August. In the shallow lakes, the period was markedly longer, from March to November with the exception of July and August (Fig. 3; Table 3). $PO_4$ exhibited a similar pattern and was lower in months with reduced TP. The $PO_4$ reduction appeared to occur progressively throughout the investigation period.

In general, nitrogen concentrations during the season changed little from period 1 to 3, but an overall trend of reduced TN and $NO_3$ could be detected, whereas $NH_4$ did not change in either shallow or deep lakes (Fig. 3; Table 3). The most pronounced change was a significant decrease in TN in the shallow lakes between May and August.

In both deep and shallow lakes, pH tended to decrease from period 1 to 3 in early summer to late autumn, the trend being particularly evident in the shallow lakes (Fig. 4). The decrease was most pronounced from the first to the second period. TA increased from period 1 to 3 during summer in both lake types, most markedly from April to July in the shallow lakes and in October in the deep lakes (Table 3). The pattern of silica was less clear, but pointed towards reduced concentrations in both shallow and deep lakes during the recovery period. Most obvious was a significant reduction in June in

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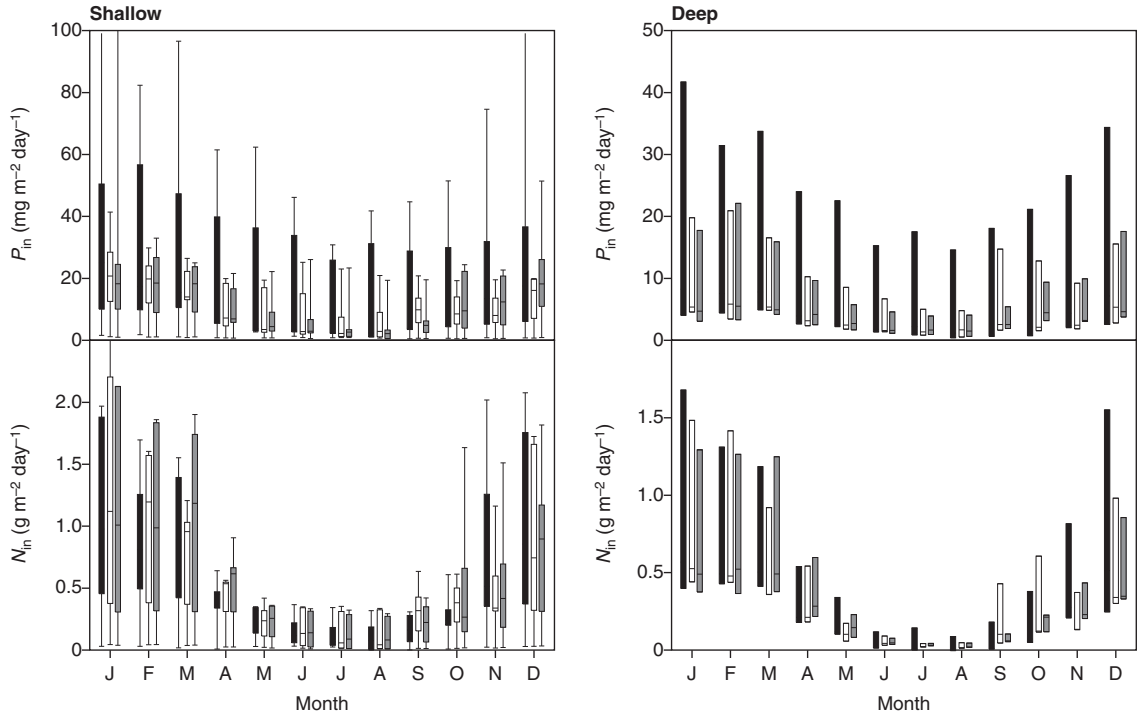



**Fig. 1** Seasonal changes in total phosphorus (TP) and total nitrogen (TN) loading in eight shallow and three deep Danish lakes during three periods following reduction of phosphorus loading: 1989–92 (left black bars), 1993–97 (middle white bars) and 1998–2001 (right grey bars).

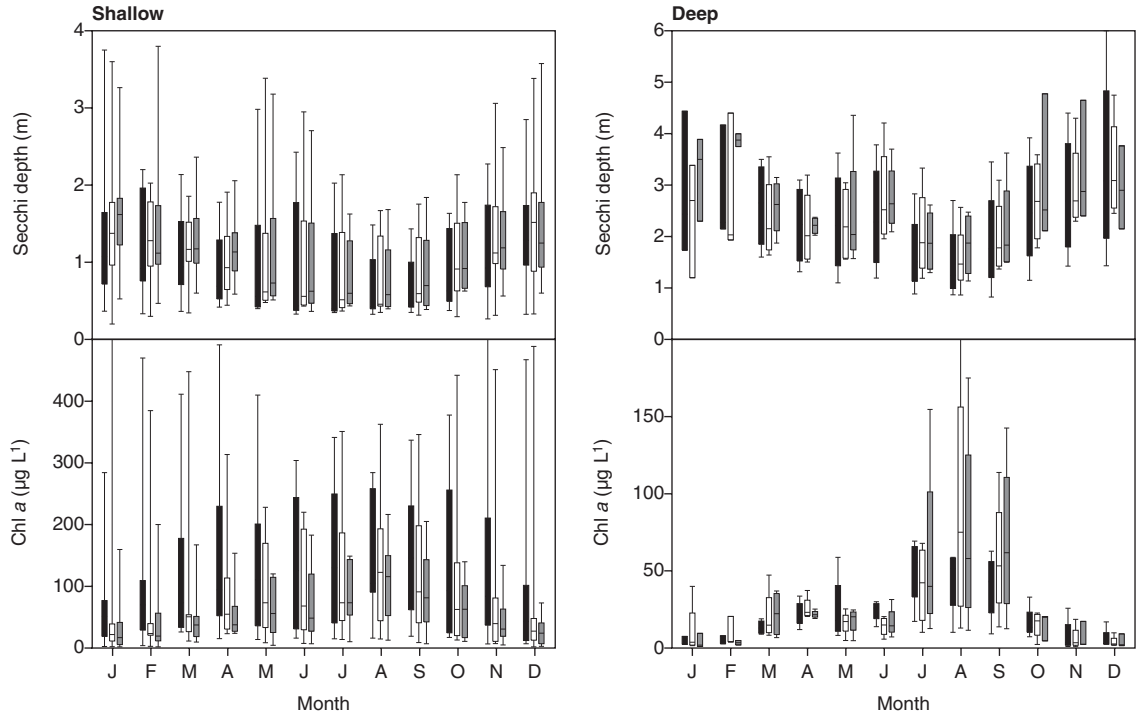



**Fig. 2** Seasonal changes in Secchi depth and chlorophyll *a* concentrations in eight shallow and four deep Danish lakes during three periods following reduction of phosphorus loading: 1989–92 (left black bars), 1993–97 (middle white bars) and 1998–2001 (right grey bars).

**Table 3** Results of statistical analyses of seasonal changes (1989–2001) in median values determined in four deep and eight shallow Danish lakes with reduced TP loading

| | | Month | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Variable | Lake type | J | F | M | A | M | J | J | A | S | O | N | D |
| Secchi | Shallow | ++ | | | +++ | | | | | ++ | ++ | | |
| | Deep | | | | | | | | | | | +++ | |
| Chl | Shallow | | | - | - - - | - - | | | | - - | | | |
| | Deep | | | | | | | | | | | - - - | |
| TP | Shallow | - - - | | - - - | - - - | - - - | - - - | | | - - - | - - - | - - | |
| | Deep | | | | | - | - | - - | | | | | |
| $PO_4$ | Shallow | | | | - - - | - - - | - - - | | - | - - - | - - - | - | |
| | Deep | | | | | - - - | | - - | - - | | | | |
| TN | Shallow | | - | - | - | - - | - - - | - - | | - | | | |
| | Deep | | | | | | | | - | | | | |
| $NO_3$ | Shallow | | | - | | | | - - | | | | | |
| | Deep | | | | | | | | | | | | |
| $NH_4$ | Shallow | + | | | | | | - - | | | | | |
| | Deep | | | | | | | | | | | | |
| TA | Shallow | | | | +++ | ++ | | + | | | | | |
| | Deep | | | | | | + | | | + | +++ | | |
| Si | Shallow | | | | | | - - - | | - | | - - | | |
| | Deep | | | | | | | | | | | | |
| TN:TP | Shallow | ++ | + | +++ | +++ | +++ | +++ | +++ | +++ | +++ | +++ | +++ | + |
| | Deep | | | | | | | | | | | | |
| Chl:TP | Shallow | | | | | | | ++ | ++ | | + | | |
| | Deep | | | | | +++ | | | ++ | | | | |
| % $PO_4$ | Shallow | | - - | | | - - - | - - - | - - | - - - | - - - | - - | - | - - |
| | Deep | | ++ | | | - - - | | | - - | | +++ | | ++ |

+: increase, $P < 0.1$; ++: increase, $P < 0.05$; +++: increase, $P < 0.01$; -: decrease, $P < 0.1$; - -: decrease, $P < 0.05$; - - -: decrease $P < 0.01$; empty cell, $P > 0.1$.

the shallow lakes, and a trend of lower concentrations during winter and spring. Si tended to decrease from period 1 to 3 in the deep lakes from March to July, particularly from the first to the second period. During autumn, Si tended to increase in the deep lakes, but only significantly so in September.

No substantive changes occurred in the TN : TP ratio in the deep lakes, while it increased during most of the season in the shallow lakes (Fig. 5), most significantly between March and November. The ratio between chlorophyll and TP varied considerably, but overall tended to increase in both shallow and deep lakes, particularly in July and August in the deep lakes. The proportion of TP present as $PO_4$ decreased during most of the season in both shallow and deep lakes, most markedly in the shallow lakes.

## Discussion

As was expected, the decreased phosphorus loading led to reduced TP concentrations in the lakes. However, as shown previously in mass balance studies of Danish shallow lakes (Søndergaard *et al*., 1999), the reduction was usually lower than expected from simple empirical equations (Organisation for Economic Co-operation and Development (OECD) 1982) because of the internal loading of phosphorus. The importance of sediment phosphorus release for lake water concentrations is emphasised by the highly increased concentrations during summer as discussed previously for 15 shallow eutrophic Danish lakes including the shallow lakes in this study (Søndergaard *et al*., 2002). A similar response has been observed in many other eutrophic lakes after a loading reduction (Marsden, 1989; Sas, 1989; Rossi & Premazzi, 1991; Scharf, 1999). In shallow lakes, the duration of negative TP retention, as well as occurrence of maximum concentrations of TP, has been shown to gradually decrease after a loading reduction (Søndergaard *et al*., 2002). These trends are confirmed by our study, which additionally demonstrates that the decline in TP was smaller in July and August than in the other summer

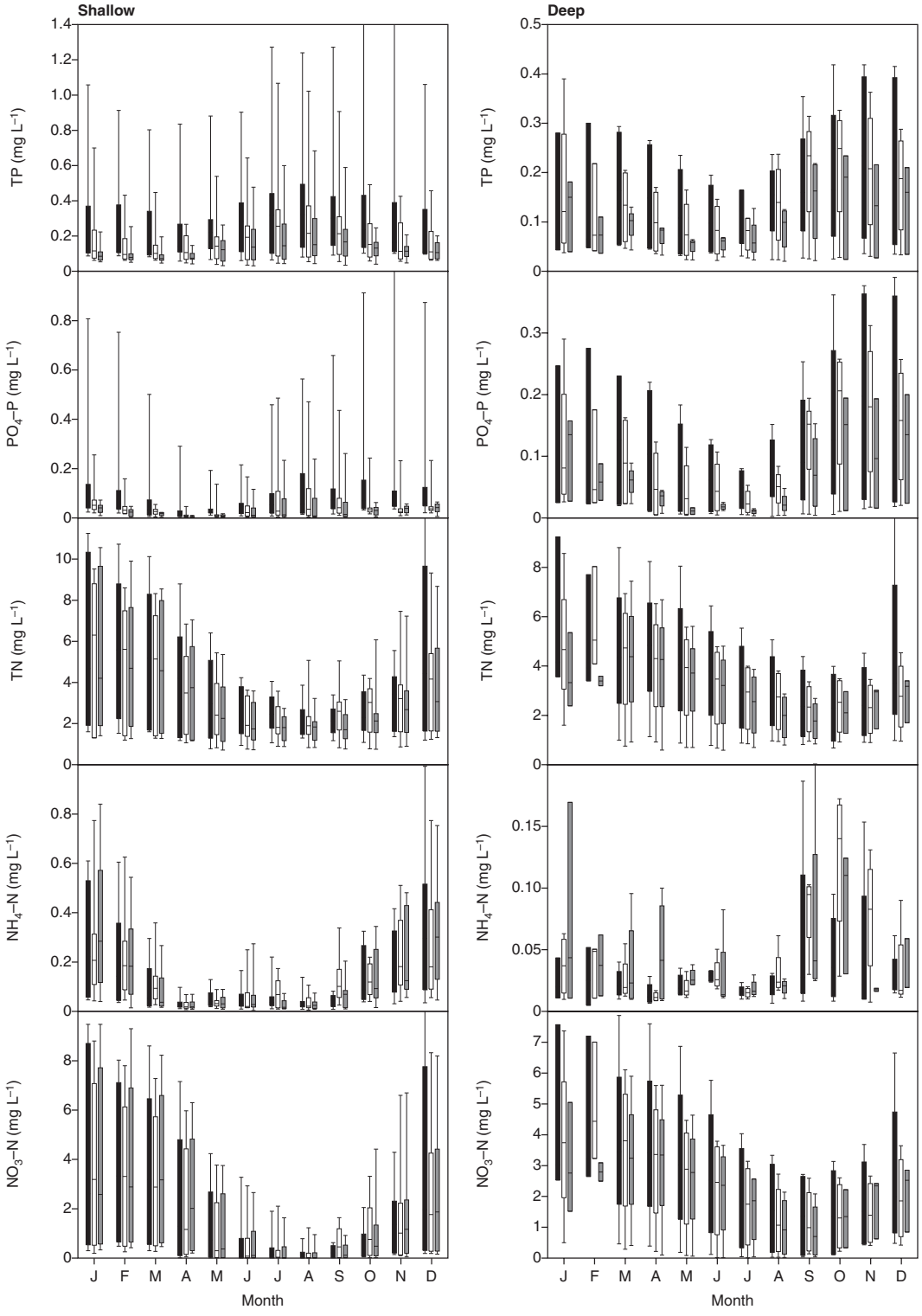



**Fig. 3** Seasonal changes in phosphorus and nitrogen concentrations in eight shallow and four deep lakes during three periods following reduction of phosphorus loading: 1989–92 (left black bars), 1993–97 (middle white bars) and 1998–2001 (right grey bars).

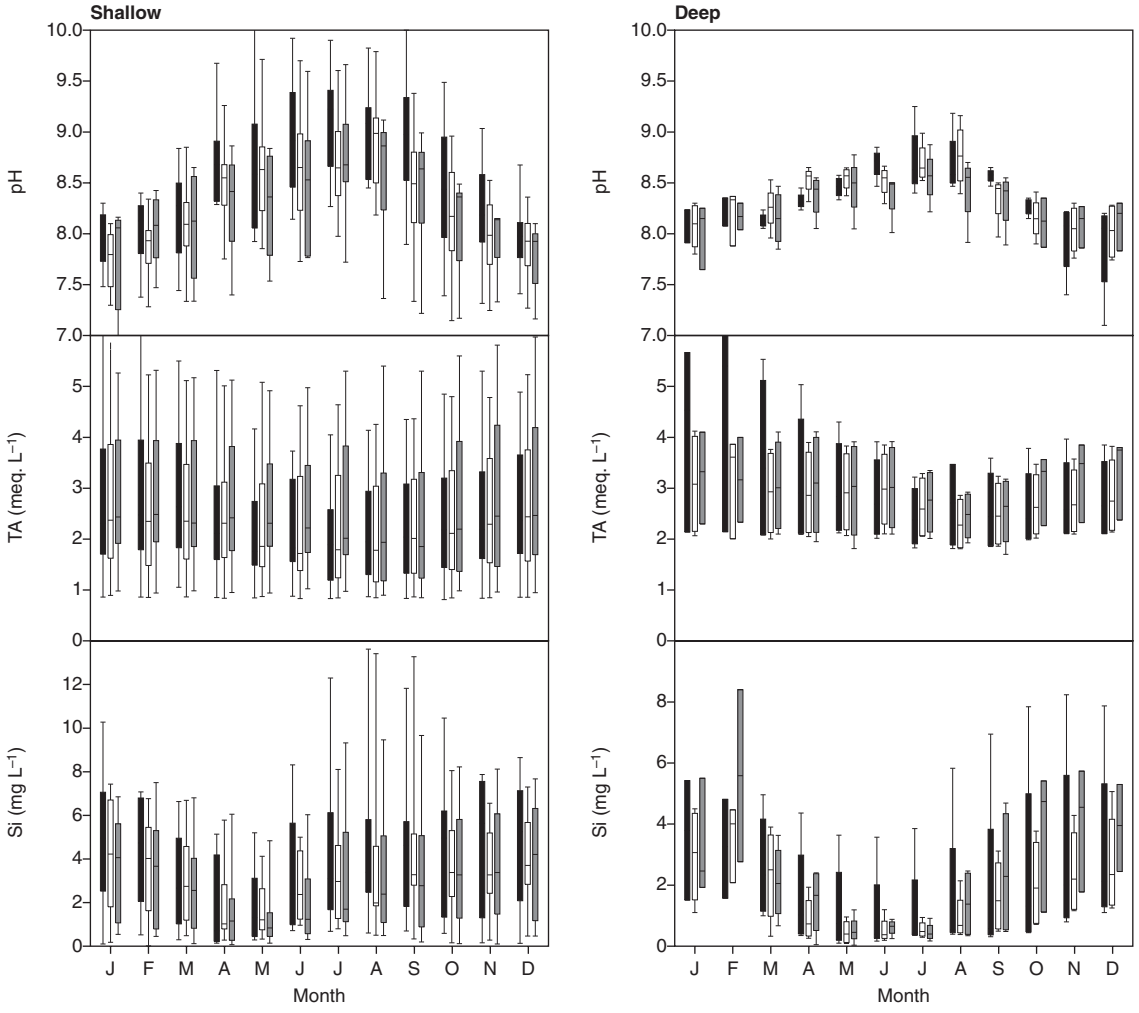



**Fig. 4** Seasonal changes in pH, total alkalinity (TA) and silica concentrations in eight shallow and four deep Danish lakes during three periods following reduction of phosphorus loading: 1989–92 (left black bars), 1993–97 (middle white bars) and 1998–2001 (right grey bars).

months (Fig. 3). Similarly, in Barton Broad, U.K., Phillips *et al.* (2005) observed a fast response of TP in spring and early summer, but a delayed response for 15 years during late summer after a loading reduction. These results also lend support to those of Köhler, Bernhardt & Hoeg (2000) and Köhler *et al.* (2005) who noted a rapid spring decline of TP in Lake Müggelsee, Germany, following reduced nutrient input.

A likely explanation of the seasonal pattern of phosphorus concentrations during recovery from excessive nutrient loading could be that phosphorus released from the sediment in spring and early summer originates from a phosphorus pool accumulated during the previous winter. In contrast, phosphorus released in late summer derives from deeper parts of the sediment and was thus accumulated a longer time ago (Søndergaard *et al.*, 1999). Phosphorus moving upwards from deeper parts of the sediment during winter may more easily be trapped in the oxidised surface layers of the sediment. Furthermore, the reduced phosphorus loading and the resultant lower phytoplankton biomass during spring would lead to lower organic input to the sediment, lower oxygen demand for decomposition and higher P sorption capacity in the sediment.

Si-induced desorption of phosphorus from Fe, Al and Mn oxides has also been suggested to be involved in the phosphorus mobility from the sediment (Krivtsov, Sigee & Bellinger, 2001). Dissolution of diatoms in the sediment surface layer occurring within a few weeks after their sedimentation might produce Si pulses high enough to influence the mobility of

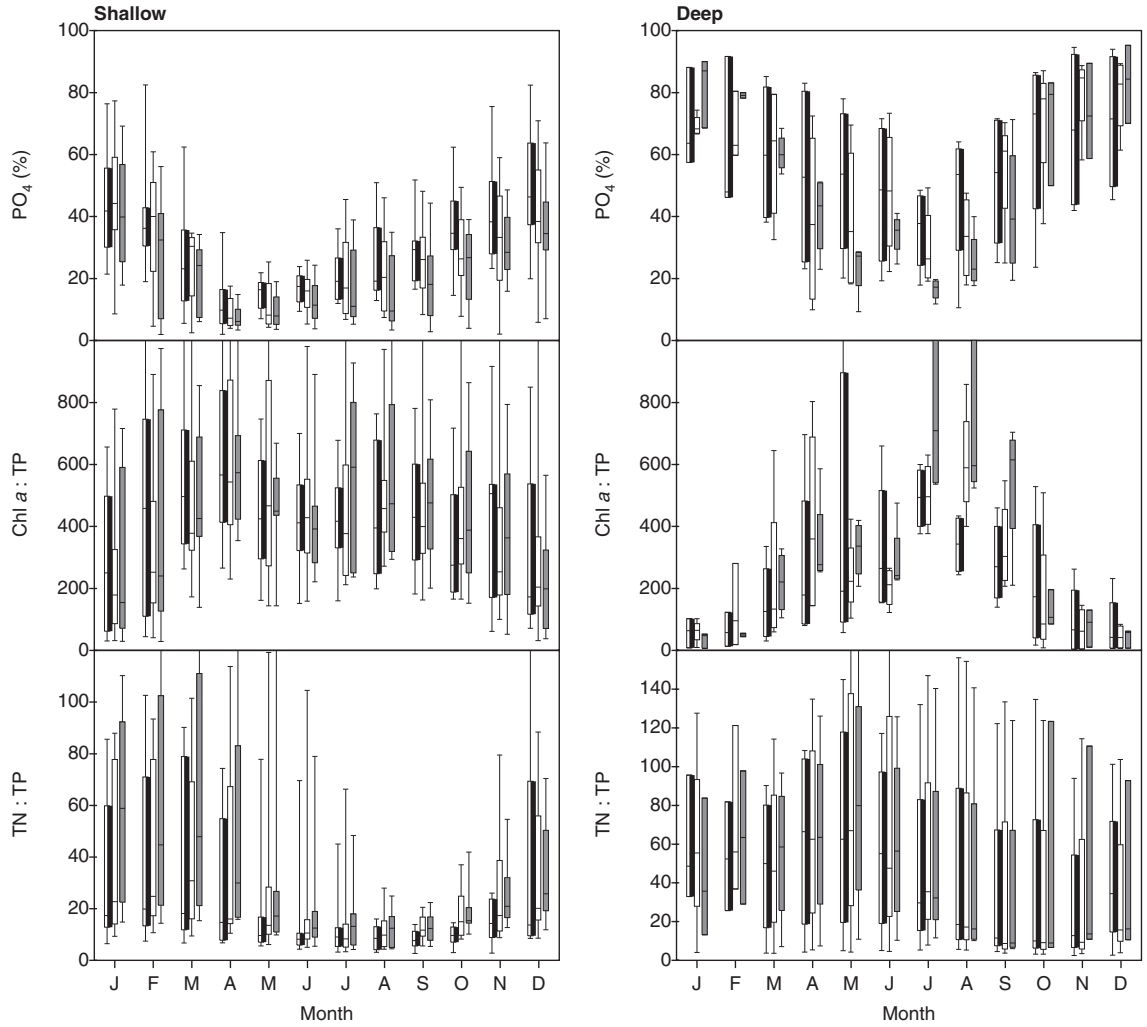



**Fig. 5** Seasonal changes in the $PO_4$:TP ratio, chl *a*:TP ratio and TN:TP ratio in eight shallow and four deep Danish lakes during three periods following reduction of phosphorus loading: 1989–92 (left black bars), 1993–97 (middle white bars) and 1998–2001 (right grey bars).

phosphorus (Tallberg, 1999). Reduced diatom sedimentation during spring might then also reduce phosphorus release, as recorded in this study; however, we cannot assess the quantitative importance of this mechanism. Seasonality in phosphorus release may also be related to differences in nitrate availability (Jensen & Andersen, 1992), as nitrate varies considerably over the season, or to reduced influence of elevated pH resulting from photosynthetic activity during the study period (Søndergaard, 1988; Welch & Cooke, 1995).

Some deep lakes appear to show a phosphorus response pattern similar to that of shallow lakes. Thus, during recovery of the over 100 m deep Lake Lucerne, periodic decreases in phosphorus occurred in spring and autumn and midsummer replenishment did not stop until more than 10 years after the occurrence of maximum concentrations (Bührer & Ambühl, 2001). In the deep lakes of the present study, however, a reduction in phosphorus concentrations apparently occurred during late summer as recovery proceeded. The reason could be thermal stratification impeding the close coupling of sediment and surface water characteristic of shallow lakes.

One of the important mechanisms of shallow lakes in general, as opposed to deep lakes, is that even relatively small improvements in turbidity may expose large bottom areas to sufficient light to induce

benthic primary production by macrophytes, filamentous or epipelic algae, and, with it, oxidation of the sediment surface (van Luijn *et al.*, 1995; Woodruff *et al.*, 1999; Phillips *et al.*, 2005). The resultant effects on internal loading in lakes shifting from the turbid to the clearwater state have been recorded for biomanipulated lakes (Søndergaard *et al.*, 2002). Correspondingly, Liboriussen & Jeppesen (2003) observed very similar levels of primary production in a clear and a turbid eutrophic lake, but in the clear lake most of the production took place at the sediment surface, while in the turbid lake it was almost completely pelagic. Lower TP concentrations in the clear state may also have been induced by a higher redox potential and sorption capacity in the surface sediment when sedimentation of organic matter declines.

During recovery following reduction in phosphorus loading, the chl *a* : TP ratio did not show any clear pattern in our study lakes, except for an increase during late summer in the shallow and particularly in the deep lakes (Fig. 5). This reflects an increase in chl *a* concentrations despite the reduced TP, indicating, in agreement with the relatively high $PO_4$ concentrations, that phytoplankton biomass was not limited by phosphorus at that time of the year. The chl *a* : TP values should be interpreted with caution, however, as some phytoplankton groups, such as filamentous cyanobacteria, may have high C : P ratios. The chl *a* : TP ratio may thus be higher in lakes dominated by these phytoplankters (Gulati & van Donk, 2002). Cyanobacteria abundance did not increase in our study lakes, however (Jeppesen *et al.*, 2005 and unpublished).

Nitrogen concentrations changed much less conspicuously than phosphorus concentrations during the recovery phase (Fig. 3). This reflects primarily the lower decrease in nitrogen loading. However, despite only insignificant changes in TN loading, a clear trend towards lower in-lake TN concentrations occurred in both deep and shallow lakes, especially during summer. As inorganic nitrogen in the shallow lakes did not show a similar decline, the decreasing TN concentrations are probably attributable to a reduction in the particulate fraction as well as to the overall reduced phytoplankton biomass. Based on data from 695 Danish lakes, the relationship between particulate nitrogen ($N_{part}$) defined as TN – $NO_3$ – $NH_4$ and chl *a* can be described as chl $a = 8.9 + 36.2N_{part}$ ($P < 0.0001$, $r^2 = 0.21$), which means that the observed reduction of 1–1.5 mg TN $L^{-1}$ during summer more or less reflects the simultaneous reduction in chl *a* of 60–70 μg $L^{-1}$. In the deep lakes nitrate tended to decrease during most of the year, as reflected by the reduced TN values (Fig. 3). During the first part of the investigation period, the reduction may be attributed to the reduced loading, the continuing reduction suggesting a higher loss through denitrification. As total concentrations do not necessarily represent biologically available forms, predictions of limiting nutrients based on TN : TP ratios may overestimate the importance of phosphorus (Schelske, Aldridge & Kenney 1999). Nevertheless, the increasing ratio suggests that, as recovery proceeds, nitrogen is less likely to become a limiting nutrient in Danish lakes.

In both shallow and deep lakes, silica generally followed the classical pattern with decreasing concentrations in late winter and spring and increasing concentrations in summer and autumn, reflecting the spring uptake of silica by diatoms followed by release from the sediment later in the season (Tessenow, 1966; Bührer & Ambühl, 2001). Only from May to July did Si in the deep lakes reach levels close to the limit of 0.5 mg Si $L^{-1}$, where silica depletion is normally expected to affect the diatoms (Stoermer & Smol, 1999). The tendency to lower Si concentrations from April to June in both deep and shallow lakes during recovery opposes the trend seen in Lake Michigan where Si concentrations increased after reduced phosphorus loading, this being attributed to a simultaneous reduction in diatom biomass and reduced Si uptake (Barbiero *et al.*, 2002). This is not likely to occur in the Danish lakes as diatom biomass decreased during the period (Jeppesen *et al.*, 2005). More probable explanations are instead reduced Si release from the sediment coupled to either a slower decomposition of diatoms when sedimentation of organic matter is reduced or to increased benthic uptake of Si with improved light conditions. The latter mechanism has also been suggested to account for the decreased spring Si concentrations in Barton Broad (Phillips *et al.*, 2005), although the pelagic diatoms were reduced to a minimum in that lake.

In summary, reduced TP loading caused marked changes in TP, TN and Si in both shallow and deep lakes. Phosphorus, in particular, appeared to be influenced by internal processes and in shallow lakes especially spring and early summer concentrations declined progressively, possibly because of increased benthic-pelagic coupling as lake water became

clearer. Nitrogen concentrations primarily decreased because of lower phytoplankton biomass, while Si decreased probably because of lower release from the sediment.

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# Water Framework Directive: ecological classification of Danish lakes

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**Summary**

**1.** The European Water Framework Directive (WFD) requires that all European waterbodies are assigned to one of five ecological classes, based primarily on biological indicators, and that minimum good ecological quality is obtained by 2015. However, the directive provides only general guidance regarding indicator definitions and determination of boundaries between classes.

**2.** We used chemical and biological data from 709 Danish lakes to investigate whether and how lake types respond differently to eutrophication. In the absence of well-defined reference conditions, lakes were grouped according to alkalinity and water depth, and the responses to eutrophication were ordered along a total phosphorus (TP) gradient to test the applicability of pre-defined boundaries.

**3.** As a preliminary classification we suggest a TP-based classification into high, good, moderate, bad and poor ecological quality using 0–25, 25–50, 50–100, 100–200 and > 200 μg P $L^{-1}$ boundaries for shallow lakes, and 0–12·5, 12·5–25, 25–50, 50–100 and > 100 μg P $L^{-1}$ boundaries for deep lakes. Within each TP category, median values are used to define preliminary boundaries for the biological indicators.

**4.** Most indicators responded strongly to increasing TP, but there were only minor differences between low and high alkalinity lakes and modest variations between deep and shallow lakes. The variability of indicators within a given TP range was, however, high, and for most indicators there was a considerable overlap between adjacent TP categories. Cyanophyte biomass, submerged macrophyte coverage, fish numbers and chlorophyll *a* were among the 'best' indicators, but their ability to separate different TP classes varied with TP.

**5.** When using multiple indicators the risk that one or more indicators will indicate different ecological classes is high because of a high variability of all indicators within a specific TP class, and the 'one out – all out' principle in relation to indicators does not seem feasible. Alternatively a certain compliance level or a 'mean value' of the indicators can be used to define ecological classes. A precise ecological quality ratio (EQR) using values between 0 and 1 can be calculated based on the extent to which the total number of indicators meets the boundary conditions, as demonstrated from three Danish lakes.

**6.** *Synthesis and applications*. The analysis of Danish lakes has identified a number of useful indicators for lake quality and has suggested a method for calculating an ecological quality ratio. However, it also demonstrates that the implementation of the Water Framework Directive faces several challenges: gradual rather than stepwise changes for all indicators, large variability of indicators within lake classes, and problems using the one out – all out principle for lake classification.

*Key-words*: ecological quality ratio, eutrophication, indicators, phosphorus, recovery

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## Introduction

The European Water Framework Directive (WFD) was adopted in December 2000 to protect and improve the quality of all surface water resources (European Union 2000). Its main target is to achieve as a minimum 'good ecological status' in all waterbodies by the year 2015. The WFD operates with five different ecological classes assessed by using a wide array of biotic variables, including phytoplankton, macrophytes, invertebrates and fish. However, the directive is not very specific and provides only general guidance on how to define the proposed ecological classes (Wallin, Wiederholm & Johnson 2003). One of the major and more practical challenges for implementation of the directive is therefore how to define and determine the ecological status of a specific waterbody.

According to the WFD the ecological state of a waterbody should be defined relative to its deviation from the reference condition, i.e. the expected ecological quality in the absence of anthropogenic influence. Reference conditions and ecological classifications need to be specified to individual lake types, as different lakes do not necessarily respond similarly to a stress factor such as, for instance, eutrophication. Reference conditions can be determined using different approaches: palaeolimnological analyses, identification of the characteristics of unimpacted sites, historical data, modelling, expert judgement, or a combination of these (Laird & Cumming 2001; Gassner, Tischler & Wanzenböck 2003; Nielsen *et al*. 2003). However, defining reference conditions is problematic given the often limited availability of data and high natural variability. Moreover, it is debatable how far back in time we should go to find minimally impacted conditions, as lakes often undergo gradual change over time, as demonstrated by palaeolimnological studies (Bradshaw 2001; Johansson *et al*. 2005). Recent studies indicate that it may be extremely difficult to find minimally impacted lakes to act as reference sites (Bennion, Fluin & Simpson 2004).

To overcome this issue of reference conditions, we selected total phosphorus (TP) as the key variable for lake water quality. While this neglects the philosophy of using the reference state to define a present ecological state and increases the risk of circular conclusions, it provides an opportunity to evaluate the classification of lakes relative to TP, which in turn might help the implementation of the WFD. Clearly, the classification of lakes in the WFD must eventually be based on biological indicators, but TP is the main environmental stressor and the primary determining factor for numerous biological variables, and it is also used in present-day lake classification (Vollenweider & Kerekes 1982; Wetzel 2001). In our analyses we have ordered along a TP gradient a number of pre-selected ecological variables often used in lake monitoring, in order to trace their potential applicability for ecological classification. We used multivariate analyses to test the applicability of the selected indicators, acknowledging that categorization of lakes according to a rigid classification scheme is problematic because changes of biological indicators along a phosphorus gradient often occur gradually rather than in a stepwise fashion (Jeppesen *et al*. 2000).

The selection of the ecological indicators was based on the response of the variables to eutrophication, but at this large scale it was constrained by data availability; for example we have no good data on benthos. Species richness and biodiversity change along a phosphorus gradient (Jeppesen *et al*. 2000), but were not included because the diversity of many biological variables relevant for WFD are sensitive to lake size (Dodson, Arnott & Cottingham 2000; Oertli *et al*. 2002; Søndergaard, Jeppesen & Jensen 2005). Data were grouped according to alkalinity and depth, two of the main factors used in lake typology (European Union 2000; Rioual 2002; Ruoppa & Karttunen 2002). Hydromorphology and variables such as lake area and salinity, which also influence the structure and function of lakes (Jeppesen *et al*. 1994; Moss 1994; Søndergaard, Jeppesen & Jensen 2005) were omitted from the present study because of scarcity of data.

Our aims were to: (i) identify potential good indicators and analyse their distribution along a phosphorus gradient for different lake types; (ii) analyse potential boundaries between the WFD's five ecological classes and develop a method to calculate an ecological quality ratio (EQR); (iii) elucidate potential problems in the implementation of the WFD and contribute to the ongoing and future intercalibration exercise aimed at establishing a common implementation strategy.

## Materials and methods

### STUDY SITES

A total of 709 lakes was included in the analyses. Chemical data were available for most lakes, whereas biological data were more scarce. Although very small lakes are the most prominent lake type in Denmark, we only included lakes > 1 ha. This is because very small lakes and ponds generally respond differently to eutrophication than larger lakes (Søndergaard, Jeppesen & Jensen 2005). The lakes covered a large morphological gradient, but were dominated by relatively small and shallow lakes (Table 1). Chemically, most lakes were alkaline and eutrophic, with high nutrient concentrations. All Danish lakes are lowland lakes situated at an altitude < 200 m a.s.l.

The lakes were grouped according to total alkalinity (TA), using a boundary of 0·2 meq $L^{-1}$ to distinguish between 'low' and 'high' alkalinity lakes and also to separate isoetids from other submerged macrophyte communities. A mean depth of 3 m was used to separate shallow from deeper lakes, and this ensured that all shallow lakes included in the analyses were unstratified and that most of the deep lakes were temporarily or permanently stratified during summer (Søndergaard, Jensen & Jeppesen 2003). About 90% of Danish lakes

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**Table 1.** Characteristics of the lakes included in the analyses. *n*, number of lakes

| Variable | *n* | Mean | Minimum | 25% | Median | 75% | Maximum |
|---|---|---|---|---|---|---|---|
| Mean depth (m) | 627 | 2·0 | 0·5 | 1·0 | 1·0 | 2·0 | 16·5 |
| Area (ha) | 709 | 48·7 | 1·0 | 1·1 | 3·0 | 17·4 | 4196 |
| Alkalinity (meq $L^{-1}$) | 607 | 1·97 | −0·11 | 0·74 | 1·94 | 2·89 | 7·20 |
| TP (mg $L^{-1}$) | 692 | 0·266 | 0·005 | 0·058 | 0·138 | 0·278 | 3·70 |
| TN (mg $L^{-1}$) | 690 | 2·37 | 0·2 | 1·17 | 1·92 | 2·99 | 16·8 |
| Secchi depth (m) | 650 | 1·28 | 0·19 | 0·64 | 0·99 | 1·56 | 10·2 |
| Chlorophyll *a* (μg $L^{-1}$) | 661 | 66 | 1·3 | 13 | 36 | 77 | 1420 |

**Table 2.** Suggested indicator boundaries in lakes with mean depth < 3 m or mean depth ≥ 3 m and TA > 0·2 meq $L^{-1}$. The boundaries are based on median values (with few exceptions). No data is indicated by –. Note that piscivores include all potential piscivores (perch, pike and pike-perch) irrespective of size). D, deep; S, shallow; DW, dry weight; ind, individuals

| | High | | Good | | Moderate | | Poor | | Bad | |
|---|---|---|---|---|---|---|---|---|---|---|
| Indicator/class | D | S | D | S | D | S | D | S | D | S |
| TP (μg P $L^{-1}$) | < 12.5 | < 25 | < 25 | < 50 | < 50 | < 100 | < 100 | < 200 | > 100 | > 200 |
| TN (mg N $L^{-1}$) | – | < 1·0 | < 1·0 | < 1·0 | < 1·0 | < 1·4 | < 1·4 | < 2·0 | < 2·2 | < 2·9 |
| SS (mg DW $L^{-1}$) | – | < 3·0 | < 2·5 | < 4·0 | < 4·2 | < 7·0 | < 7·0 | < 13 | < 8·6 | < 20 |
| Secchi (m) | > 5·1 | > 2·1 | > 3·9 | > 1·7 | > 2·5 | > 1·0 | > 1·8 | > 0·9 | > 1·3 | > 0·7 |
| CHLA (μg $L^{-1}$) | – | < 6·0 | < 6·5 | < 12 | < 12 | < 22 | < 27 | < 57 | < 56 | < 82 |
| Total phytoplankton ($mm^3$ $L^{-1}$) | – | < 0·68 | < 2·3 | < 1·4 | < 2·3 | < 3·3 | < 6·7 | < 15·3 | < 9·1 | < 18·0 |
| Chrysophytes ($mm^3$ $L^{-1}$) | – | > 0·27 | > 0·17 | > 0·27 | > 0·07 | > 0·01 | ≥ 0 | ≥ 0 | ≥ 0 | ≥ 0 |
| Diatoms ($mm^3$ $L^{-1}$) | – | < 0·04 | < 0·23 | < 0·12 | < 0·36 | < 0·32 | < 0·90 | < 2·9 | < 0·90* | < 2·9† |
| Chlorophytes ($mm^3$ $L^{-1}$) | – | < 0·03 | < 0·09 | < 0·12 | < 0·09 | < 0·23 | < 0·17 | < 2·2 | < 0·17‡ | < 2·9 |
| Cyanophytes ($mm^3$ $L^{-1}$) | – | < 0·01 | < 0·09 | < 0·01 | < 0·20 | < 0·69 | < 1·9 | < 3·4 | < 1·9§ | < 6·0 |
| Total zooplankton (μg DW $L^{-1}$) | – | < 164 | < 227 | < 164¶ | < 280 | < 342 | < 436 | < 487 | < 615 | < 1024 |
| Cyclopoids (μg DW $L^{-1}$) | – | < 7 | < 47 | < 25 | < 67 | < 60 | < 78 | < 98 | < 88 | < 237 |
| Cladocerans (μg DW $ind^{-1}$) | – | > 3·0 | – | > 2·6** | – | > 2·6 | – | > 1·6 | – | > 1·1 |
| Calanoids (μg dw $ind^{-1}$) | – | – | – | < 1·1 | – | < 1·7 | – | < 2·3 | – | < 2·3 |
| Zooplankton : phytoplankton (DW : DW) | – | > 0·41 | > 0·48 | > 0·27 | > 0·40 | > 0·19 | > 0·21 | > 0·13 | > 0·16 | > 0·11 |
| Fish numbers (CPUE) | – | < 20 | < 62 | < 43 | < 93 | < 96 | < 134 | < 151 | < 149 | < 201 |
| Fish weight (CPUE, kg) | – | < 2·7 | < 3 | < 4·7 | < 4·5 | < 4·7 | < 5·4 | < 6·2 | < 7·2 | < 10·3 |
| Piscivore (weight percentage) | – | (100) | > 58 | > 64 | > 42 | > 42 | > 35 | > 21 | > 26 | > 10 |
| Piscivore (number percentage) | – | (100) | > 61 | > 56 | > 58 | > 46 | > 57 | > 36 | > 45 | > 10 |
| Piscivore weight (g $ind^{-1}$) | – | > 111 | > 56 | > 84 | > 56 | > 42 | > 40 | > 36 | > 40†† | – |
| Macrophyte max depth (m) | – | 5·0 | > 5·0 | 3·4 | – | 1·3 | – | – | – | – |
| Macrophyte coverage (%) | – | 58 | – | 41 | – | 4 | – | – | – | – |

Medians: *0·78; †2·2; ‡0·12; §1·2; ¶143; **2·2; ††43.

> 5 ha have an alkalinity above 0·2 meq $L^{-1}$ and about 70% have a mean water depth below 3 m (Søndergaard, Jeppesen & Jensen 2003). Thus, the most detailed data comprised shallow and relatively alkaline lakes. Brackish lakes with a conductivity > 80 mS $m^{-1}$ or a chloride concentration > 140 mg Cl $L^{-1}$ were not included.

To elucidate changes over time in EQR, we selected three shallow high-alkalinity lakes monitored since 1989. Lake Arreskov (area 317 ha, mean depth 1·9 m) is eutrophic, with highly fluctuating environmental conditions [summer mean TP from 57 to 231 μg P $L^{-1}$, chlorophyll *a* (CHLA) from 12 to 146 μg $L^{-1}$, Secchi depth from 0·3 to 2·5 m]. Lake Soby (area 83 ha, mean depth 2·8 m) is mesotrophic, with more stable conditions (summer mean TP from 16 to 27 μg P $L^{-1}$, CHLA from 4 to 17 μg $L^{-1}$, Secchi depth from 2·4 to 4·0 m). Lake Damhus (area 46 ha, mean depth 1·6 m) is under recovery from eutrophication (summer mean TP from 43 to 123 μg P $L^{-1}$, CHLA from 6 to 39 μg $L^{-1}$, Secchi depth from 1·4 to 1·9 m). Data on submerged macrophytes were only available from 1993 to 2002. The fish community was investigated three to seven times during 1989–2002 and data from each year were obtained through interpolation or for a few lake years through extrapolation.

## SELECTED INDICATORS AND ECOLOGICAL CLASSIFICATION

For shallow lakes we used five TP categories as a guide for the ecological indicators: 0–25 μg P $L^{-1}$ for high ecological quality, 25–50 μg P $L^{-1}$ for good quality, 50–100 μg P $L^{-1}$ for moderate quality, 100–200 μg P $L^{-1}$ for poor quality and > 200 μg P $L^{-1}$ for bad quality (Table 2). This choice was based on previous findings from Danish lakes revealing that marked changes occur for a number

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**Table 3.** Potential class boundaries to be used in the ecological classification using TP (μg P $L^{-1}$), chlorophyll *a* (μg $L^{-1}$) and Secchi depth (m) by 1, Borja *et al*. (2004); 2, Premazzi *et al*. (2003); 3, Moss *et al*. (2003) and this study. *Reference 2 states > 10 in the paper, but this is probably an error

| Waterbody type | Parameter | Study | High | Good | Moderate | Poor | Bad |
|---|---|---|---|---|---|---|---|
| Transitional | CHLA | 1 | < 4 | < 10 | < 20 | < 30 | > 30 |
| Deep lakes | TP | 2 | < 10 | < 25 | < 50 | < 100 | > 100 |
| | | This study | < 12·5 | < 25 | < 50 | < 100 | > 100 |
| | CHLA | 2 | < 3 | < 6 | < 10* | < 25 | > 25 |
| | | This study | – | < 5·9 | < 11·5 | < 28 | > 28 |
| | Secchi | 2 | > 5 | < 5 | < 2 | < 1·5 | < 1 |
| | | This study | – | > 3·9 | > 2·5 | > 1·8 | < 1·8 |
| Shallow lakes | TP | 3 | < 15 | < 30 | < 50 | < 75 | > 75 |
| | | This study | < 25 | < 50 | < 100 | < 200 | > 200 |
| | CHLA | 3 | < 10 | < 20 | < 30 | < 50 | > 50 |
| | | This study | < 5 | < 11·0 | < 21 | < 55 | > 55 |
| | Secchi | 3 | > 3 | > 3 | > 2 | > 1·0 | < 0·9 |
| | | This study | > 2 | > 1·5 | > 1·0 | > 0·8 | < 0·8 |

biological variables along a TP gradient, particularly from 0 to 100 μg P $L^{-1}$ (Jeppesen *et al*. 2000). For deep lakes, which may develop cyanobacteria blooms and respond markedly to increasing TP at levels below 25 μg P $L^{-1}$ (Sas 1989; Dokulil & Teubner 2003; Jeppesen *et al*. 2005), we categorized high ecological quality lakes with a TP between 0 and 12·5 μg P $L^{-1}$, and good ecological quality between 12·5 and 25 μg P $L^{-1}$. TP values between 25 and 50 represented moderate ecological quality, 50–100 μg P $L^{-1}$ poor quality and > 100 μg P $L^{-1}$ bad quality. The suggested TP boundaries correspond closely to the values suggested for deep lakes by Premazzi *et al*. (2003), while those of shallow lakes are somewhat higher than those proposed by Moss *et al*. (2003) (Table 3).

Twenty-two indicators were pre-selected based on data availability and on their known and presumed marked response to eutrophication (Table 2). They represented five main indicator groups (phytoplankton, zooplankton, fish, macrophytes and chemistry) and five indicators were selected from each group, except for macrophytes where only two were available. For each of the indicators we used their median values within each TP class in deep and shallow lakes to define ecological boundaries (Table 2). Median values were preferred over means to reduce the influence of extreme values. Use of 25% or 75% fractiles would lead to higher divergence from expected TP levels than use of median values (Søndergaard, Jeppesen & Jensen 2003). It must be emphasized, however, that we have only few data for deep, unimpacted lakes in Denmark and we cannot therefore suggest boundaries for biological indicators between the high and good ecological status for deep lakes. The suggested boundaries for CHLA and Secchi depth were generally comparable to those proposed by others (Table 3). EQR ratios between 0 and 1 were calculated as the mean value of the 22 indicators based on the boundaries shown in Table 2 and by assigning high, good, moderate, poor and bad quality values of 1, 0·75, 0·50, 0·25 and 0 before calculating the mean value.

### SAMPLING AND ANALYSES

Data were mainly collated by the local counties, using standard sampling techniques and analyses, as part of national and regional monitoring programmes (Kronvang *et al*. 1993). Most lakes > 5 ha were sampled monthly or more frequently during summer at a mid-lake station, while lakes between 1 and 5 ha were often sampled on a single or a few occasions during summer. Mean summer values (1 May–1 October) were calculated for each year and multi-year data were averaged to obtain one value for each lake. For phytoplankton and zooplankton, however, data from individual years on 27–37 lakes monitored biweekly during summer from 1989 to 2002 were used to ensure sufficient data, including a total of 495 lake years (summer mean values). Many of these lakes were in recovery after a reduction of the external phosphorus loading, and different years represented different phosphorus levels (Jeppesen *et al*. 1999; Søndergaard *et al*. 2002). For low-alkalinity lakes, phyto- and zooplankton data were only available for lakes with TP < 100 μg P $L^{-1}$.

TP, total nitrogen (TN), TA, suspended solids (SS) and CHLA were analysed according to standard procedures (Jespersen & Christoffersen 1987; Søndergaard, Kristensen & Jeppesen 1992). Quantitative measurements of the fish stock were expressed as catch per unit effort (CPUE) of biomass or numbers based on catches in 42-m long multiple mesh-sized gill nets with 14 different mesh sizes ranging from 6·25 mm to 75 mm (Mortensen *et al*. 1990; Jeppesen *et al*. 2004). In lakes > 5 ha, six to 24 nets were used, while fewer nets were used in lakes < 5 ha. The nets were set in late afternoon and retrieved after 18 h. Because of the low number of fish data sets these were not divided into high–low alkalinity and shallow–deep lakes. Phytoplankton and zooplankton were fixed in Lugol's iodine and identified to genus or sometimes species level. Phytoplankton biovolume was calculated by fitting each species/genus to simple geometric forms. Zooplankton biomass was calculated on the basis of

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published length–weight relationships (Dumont, van De Velde & Dumont 1975; Bottrell *et al*. 1976). Submerged macrophytes were investigated during maximum abundance in July or August, and relative abundance (% coverage of lake area) and species numbers were recorded (Jeppesen *et al*. 2000). Coverage was grouped into the following categories: 0–1%, 1–5%, 5–25%, 25–50%, 50–75% and 75–100% of total lake area.

### DATA ANALYSES AND STATISTICS

Multivariate statistics were conducted for four indicator groups (phytoplankton, zooplankton, fish and macrophytes) and five–six environmental variables [TP, TN, TA, pH, mean depth and Secchi depth (macrophytes only)]. Each lake was only represented by measurements for a single year, the years being selected randomly if data were available from several years. All response and environmental variables except pH were $\log(x + 1)$ transformed. Initial exploratory analyses of each indicator group were performed using detrended correspondence analysis (DCA) to determine whether linear or unimodal statistical techniques would be most appropriate for the modelling of responses (Birks 1995; ter Braak 1995). Unconstrained ordination (principal components analysis, PCA) was applied separately to each indicator group to estimate the explanatory power of the best possible environmental variable (eigenvalue of the PCA axis one), and PCA of the environmental data was performed to explore redundancy (collinearity) within the variables. Constrained ordination (redundancy analysis, RDA) using forward selection (FS) was applied to examine the relationship between the indicator groups and the environmental variables. Significance of the forward-selected variables was tested by Monte Carlo permutations (499 iterations) and adjusted by Bonferroni-corrected type 1 error according to ter Braak & Smilauer (2002). A series of RDA ordinations specifying only one environmental variable at a time and the remaining as covariables was run to estimate the contribution of explanatory power to the variance by each single variable. All ordinations were performed using CANOCO version 4.5 (ter Braak & Smilauer 2002). *t*-tests were used to identify differences between lake types within individual TP categories.

## Results

We present data on changes in the pre-selected indicators along a TP gradient, and their variability within each of the pre-selected TP classes. We focused mainly on deep vs. shallow alkaline lakes as data from lakes with low alkalinity were scarce. We also analysed the response strength and the overlap between adjacent TP classes, and used multivariate techniques to evaluate the potentials of the indicators.

### PHYSICAL AND CHEMICAL VARIABLES

CHLA, TN, SS and Secchi depth responded markedly to changes in TP (Fig. 1). The relative changes in medians across the TP gradient were most prominent for chlorophyll, which increased by a factor of 11–19, and for suspended solids, which increased by a factor of 4–6. All variables showed a considerable range within each TP category, and in most cases there was a considerable overlap between the 25–75 percentile for adjacent TP categories.

Generally, there were only minor differences between low- and high-alkalinity lakes, whereas the differences between shallow and deep lakes were more pronounced. For instance, Secchi depth was significantly ($P < 0{\cdot}003$) higher in deep than in shallow lakes within all TP categories, and CHLA and SS were significantly ($P < 0{\cdot}001$) higher in shallow than in deep lakes at TP > 100 μg P $L^{-1}$. Secchi depth tended to be higher in lakes with TA > 0·2 than in lakes with TA < 0·2 meq $L^{-1}$, but only significantly so ($P = 0{\cdot}02$) in lakes with TP between 100 and 200 μg P $L^{-1}$.

### SUBMERGED MACROPHYTES

Both macrophyte coverage and their maximum depth distribution decreased with increasing TP. In shallow lakes macrophyte coverage changed most markedly from the 25–50 to the 50–100 μg TP $L^{-1}$ category, where median coverage decreased from 41% to 4% (Fig. 2). In lakes with a mean depth above 3 m, median coverage never exceeded 11%. Maximum depth of submerged macrophytes tended to be lower in lakes with low alkalinity than in high-alkalinity lakes (Table 2), but none of the five TP categories differed significantly ($P > 0{\cdot}1$). Maximum depth distribution also tended to be higher in deeper (> 3 m) than in the shallow lakes, but only significantly ($P = 0{\cdot}046$) so in lakes with a TP of 100–200 μg P $L^{-1}$. The maximum depth distribution in shallow lakes should, however, be interpreted with care as actual lake depth may limit the value in some of the lakes. Isoetids were only important in lakes with TA < 0·2 meq $L^{-1}$, but their proportional contribution to macrophyte cover decreased steadily from a median of 59% in lakes with TP < 25 μg P $L^{-1}$ to 0% in lakes with TP > 100 μg P $L^{-1}$ (Fig. 2).

### PHYTOPLANKTON

Total phytoplankton, cyanophyte and chlorophyte biovolume increased with increasing TP, particularly above 50 μg P $L^{-1}$ (Fig. 2). Total biovolume did not differ significantly between low- and high-alkalinity lakes. However, in lakes with low TA biovolume was significantly lower ($P < 0{\cdot}01$) for diatoms at 25–50 and 50–100 μg P $L^{-1}$. Chlorophyte biovolume was significantly ($P = 0{\cdot}04$) higher in low- than high-alkalinity lakes at TP below 25 μg P $L^{-1}$. Total phytoplankton biovolume only differed significantly ($P \leq 0{\cdot}02$) between deep and shallow lakes at TP above 100 μg P $L^{-1}$. Chlorophyte biovolume was significantly ($P < 0{\cdot}05$) lower in deep than in shallow lakes within the TP categories 0–25, 25–50 and 100–200 μg P $L^{-1}$.

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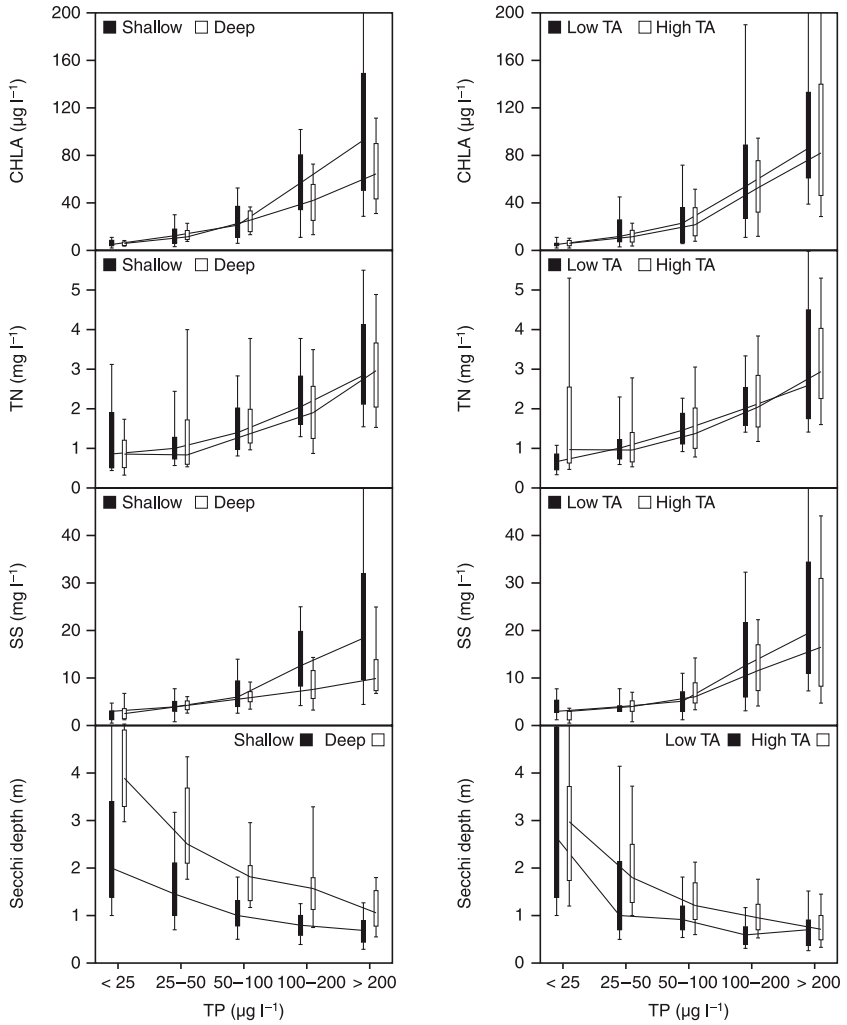



**Fig. 1.** Box plots of CHLA and chemical variables along a TP gradient. Right column: lakes with TA ≤ 0·2 meq $L^{-1}$ (low TA) and with TA > 0·2 meq $L^{-1}$ (high TA). Left column: lakes with mean depth < 3 m (shallow) and mean depth ≥ 3 m (deep). Number of lakes, 451–631. For each box 10% (bottom end of line), 25% (bottom edge of box), median (connected by lines), 75% (top edge of box) and 90% (top end of line) percentiles are shown.

### ZOOPLANKTON

Total zooplankton biomass increased with increasing TP in both shallow and deep lakes, but the variability within each TP category was high (Fig. 2). The ratio between zooplankton and phytoplankton biomass decreased steadily across the TP gradient.

### FISH

Total fish biomass (CPUE) increased three-fold and the number of fish rose eight-fold along the TP gradient (Fig. 3). With increasing TP, the percentage of potential piscivores (defined as all perch *Perca fluviatilis* L., pike *Esox lucius* L. and pike-perch *Sander lucioperca* L.) decreased both in weight and numbers.

### RESPONSE STRENGTH AND OVERLAP OF INDICATORS

To evaluate the potential usefulness of the pre-selected indicators we calculated the ratio between median values of the high–good and good–moderate classes to express the relative change of the indicators from one TP class to the next (Fig. 4). The highest ratio (greatest change) was found for some of the phytoplankton and zooplankton indicators, but the ratio varied considerably. For example, macrophyte coverage changed only negligibly between high–good classes but markedly between good–moderate classes. We also calculated the overlap between adjacent TP classes, and this was relatively high for most of the indicators (Fig. 4). For some indicators the overlap between high and good classes was more than 80%. The 'best' indicators, with the lowest overlap, were suspended solids and macrophyte maximum depth.

### ORDINATIONS

To evaluate further the relationship between the pre-selected indicators, environmental variables and the class variables mean depth and alkalinity, we conducted multivariate analyses. The gradient lengths of the first DCA axis of the four indicator groups were all less than

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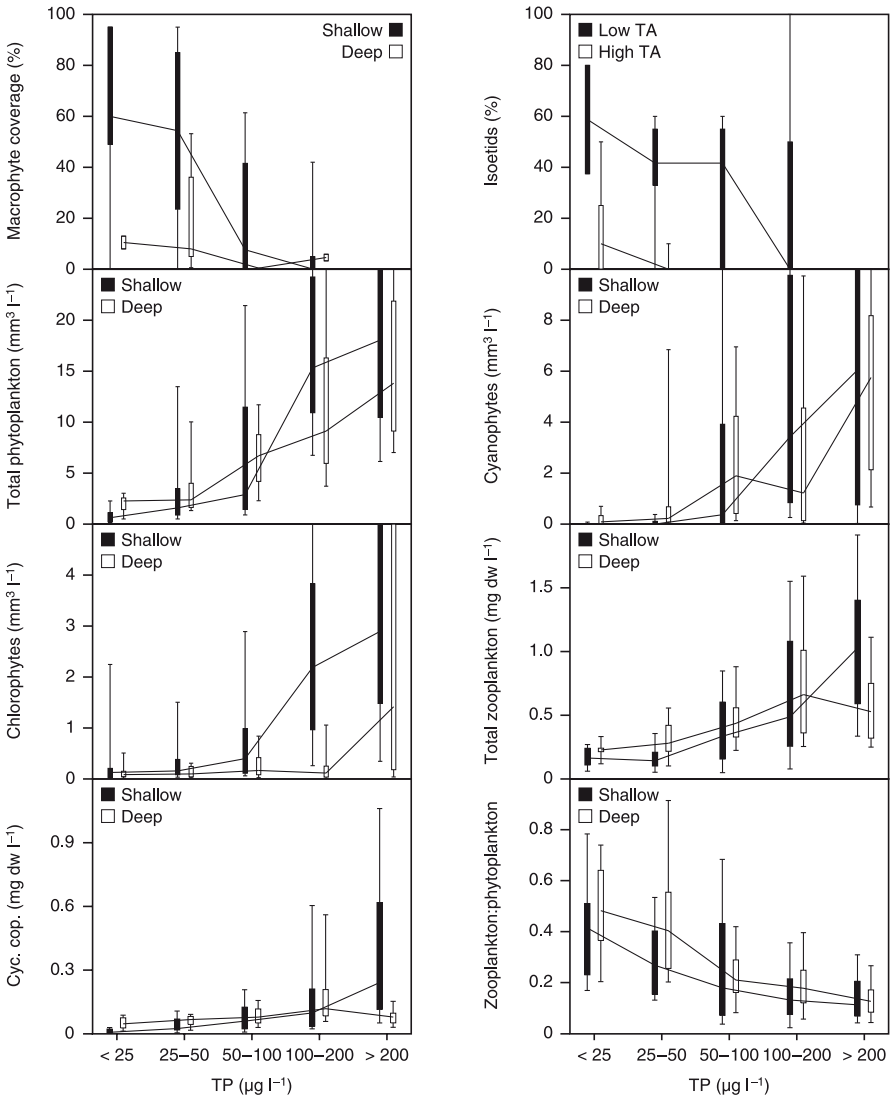



**Fig. 2.** Box plots of data on submerged macrophytes (number of lakes 30–66), phytoplankton (number of lake years 495) and zooplankton (number of lake years 495) along a TP gradient and at differing alkalinity and lake depth. For each box 10% (bottom end of line), 25% (bottom edge of box), median (connected by lines), 75% (top edge of box) and 90% (top end of line) percentiles are shown. Cyc. cop., cyclopoid copepods.

**Table 4.** Results of multivariate statistics (DCA, PCA, RDA and partial RDA) for the four indicator groups. Z, mean depth

| | Macrophytes | Fish | Zooplankton | Phytoplankton |
|---|---|---|---|---|
| Number of lakes | 37 | 71 | 82 | 67 |
| DCA: gradient length axis 1 (SD) | 0·935 | 1·332 | 0·934 | 1·551 |
| PCA: variation explained axis 1 (%) | 91·8 | 65·7 | 74·7 | 51·4 |
| RDA: variation explained by all environmental variables (%) | 61·0 | 37·6 | 30·3 | 20·5 |
| RDA: Bonferroni-adjusted FS variables | TA, Z | TP, pH | TA, TN | TP |
| Partial RDA: variation explained by interactions (%) | 23·2 | 19·8 | 19·0 | 7·2 |

2 SD (Table 4), indicating monotonic responses to underlying ecological gradients and the need for linear methods such as PCA and RDA (ter Braak 1995). The RDA ordinations showed positive relationships between pH, TA, TN and TP (Fig. 5). Macrophyte maximum depth distribution and coverage, piscivorous fish abundance (weight, numbers, mean weight), cladoceran specimen biomass and zooplankton : phytoplankton ratio, as well as the biovolume of chrysophytes, were all negatively related to TP, TN, pH and TA, while total fish abundance (weight, numbers), the biomass of total zooplankton, cyclopoid and calanoid copepods, and the biovolume of total phytoplankton, chlorophytes, diatoms and cyanophytes, showed opposite trends.

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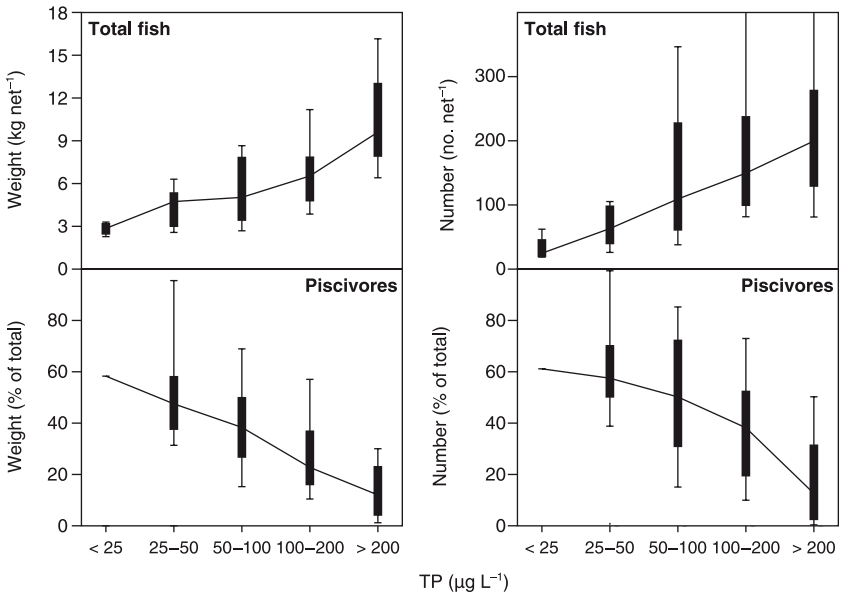



**Fig. 3.** Box plot of fish data (number of lakes 71) along a TP gradient. For each box 10% (bottom end of line), 25% (bottom edge of box), median (connected by lines), 75% (top edge of box) and 90% (top end of line) percentiles are shown. Piscivores are all potential piscivores, i.e. all sizes of perch, pike and pike-perch.

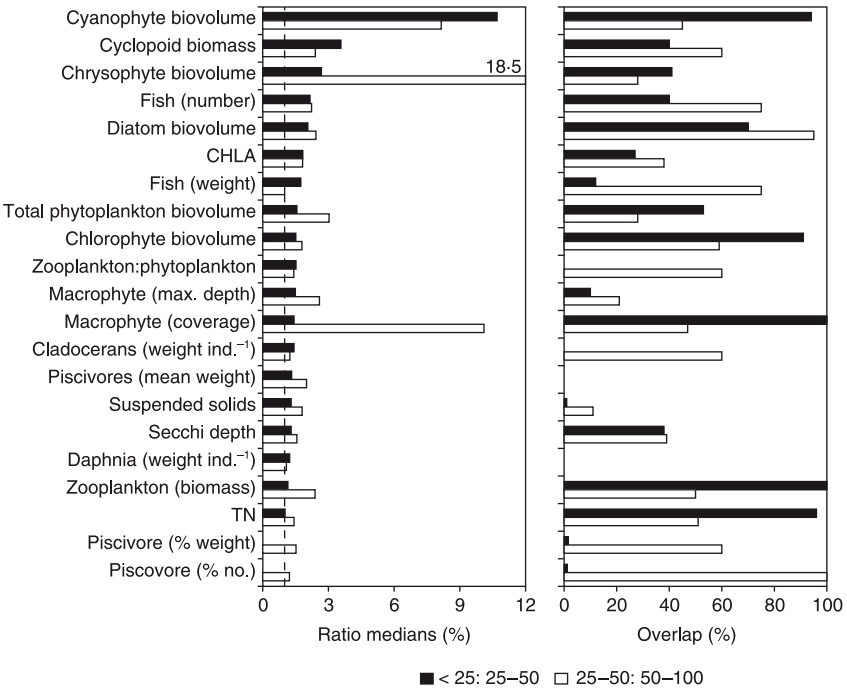



**Fig. 4.** Left panel: the ratio between the median value of each of the 21 indicators for the 0–25 : 25–50 (shaded columns) and 25–50 : 50–100 (open columns) µg P L$^{-1}$ TP classes in shallow lakes with TA > 0·2 meq L$^{-1}$. For macrophytes and fish, all lake types were included. Piscivores are all potential piscivores, i.e. all sizes of perch, pike and pike-perch. If the value of an indicator decreased with increasing TP (e.g. Secchi depth), the ratio was reversed to give values above one. The dashed line shows a ratio of 1. Right panel: overlap of 19 indicators between adjacent TP classes using the interquartile range (25–75% percentile) and expressed as percentage overlap with the next TP class, i.e. the percentage proportion of the interquartile range included in the next TP category. For example, if the 25–75% percentile of CHLA ranges from 7·2 to 17·0 µg L$^{-1}$ at 25–50 µg P L$^{-1}$, and from 13·3 to 35·8 µg L$^{-1}$ at 50–100 µg P L$^{-1}$, the overlap is (17·0 – 13·3)/(17·0 – 7·2) = 37·8%.

Between one-half and two-thirds of the variation in the response variables could be explained by the environmental variables (Table 4), although high inter-correlation (interaction) among the variables made it difficult to identify the most important variable. In general, TP, TN, TA and pH correlated with RDA axis one and explained most of the species variance, while mean depth was mainly confined to RDA axis two, except for the phytoplankton group (Fig. 5). Similar strong inter-correlations among the variables of TP, TN, TA and pH

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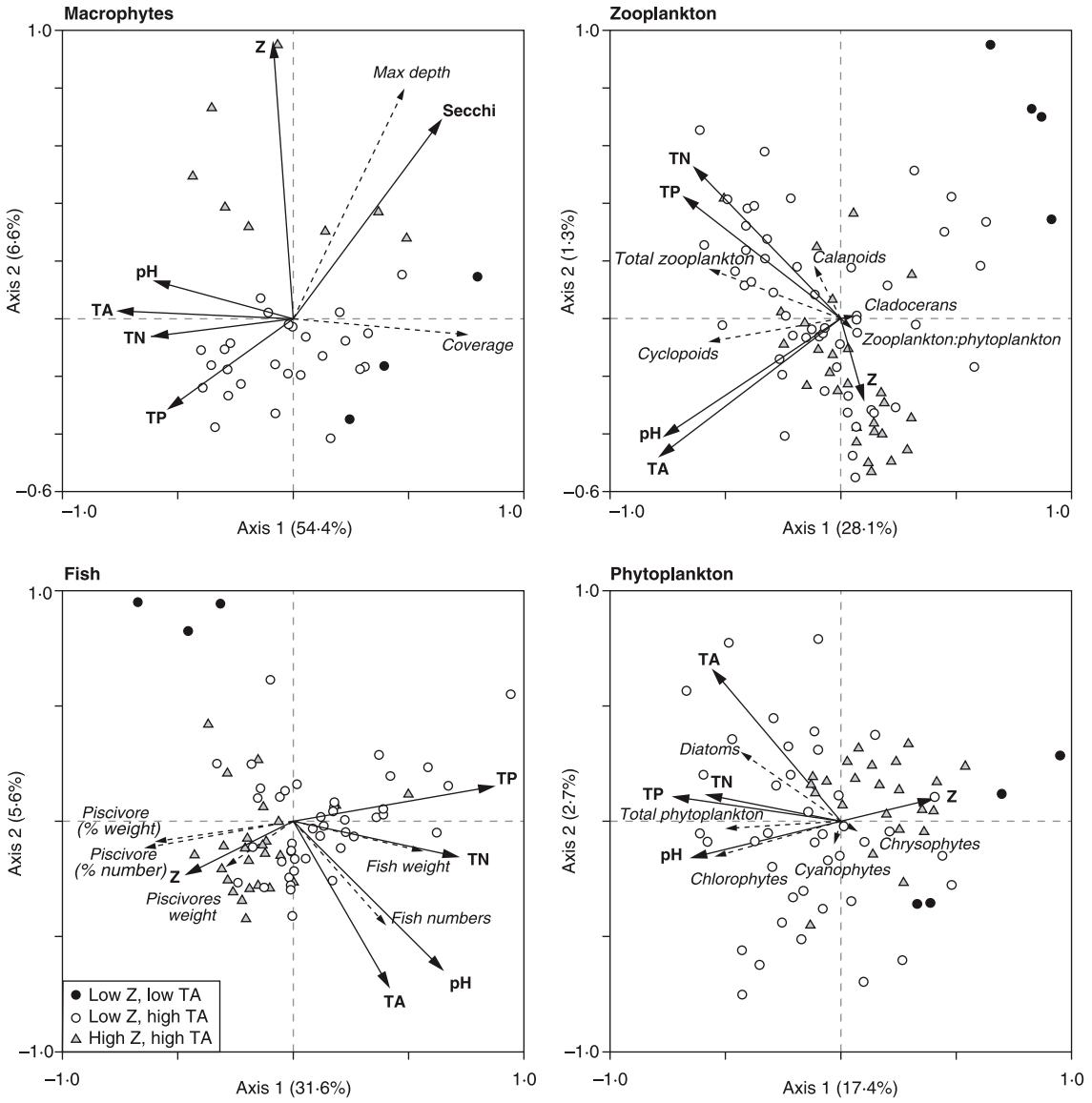



**Fig. 5.** RDA ordination plots of submerged macrophytes, fish, zooplankton and phytoplankton and environmental variables (as listed in Table 2). Environmental variables are represented by solid arrows and species variables by dotted arrows. Percentages of variance explained are given for RDA axes 1 and 2. Low alkalinity (TA) defined as < 0·2 meq $L^{-1}$, high TA as > 0·2 meq $L^{-1}$, low mean depth (Z) as < 3 m and high Z > 3 m.

were also identified by the PCA ordinations based solely on the environmental variables, with high correlation between the vectors of TP, TN, TA and pH and axis one (explaining 82–86% of variation), whereas the vector of mean depth and axis two was strongly correlated (explaining 8–11% of variation, data not shown), again except for the phytoplankton group, which showed almost opposite trends. Bonferroni-adjusted forward selection identified mainly TA, TP, TN and pH to be of significance for the observed patterns of the four indicator groups, while mean depth was only found to be significant for the macrophyte group (Table 2). Among the lakes with high alkalinity, the RDA ordinations showed deep lakes to be mainly associated with low nutrient conditions, except for the macrophyte group, whereas both low and high nutrient regimes were found for shallow lakes (Fig. 5). Thus, there was no evidence of distinct clusters relative to nutrient levels for shallow lakes with high alkalinity.

## EQR CALCULATIONS

An EQR was calculated for three lakes using 14 years of data from 1989 to 2002, and the classification compared with changes recorded in TP, CHLA and Secchi depth (Fig. 6). In the examples, calculated EQR values for the three lakes ranged between 0·14 and 0·94, representing all five ecological classes. Even in the relatively nutrient-poor Lake Soby, with no evidence of changes in the external nutrient loading, EQR varied between 0·76 and 0·94, corresponding with high or good ecological quality. Overall, the response in EQR followed the changes in TP, CHLA and Secchi depth relatively closely.

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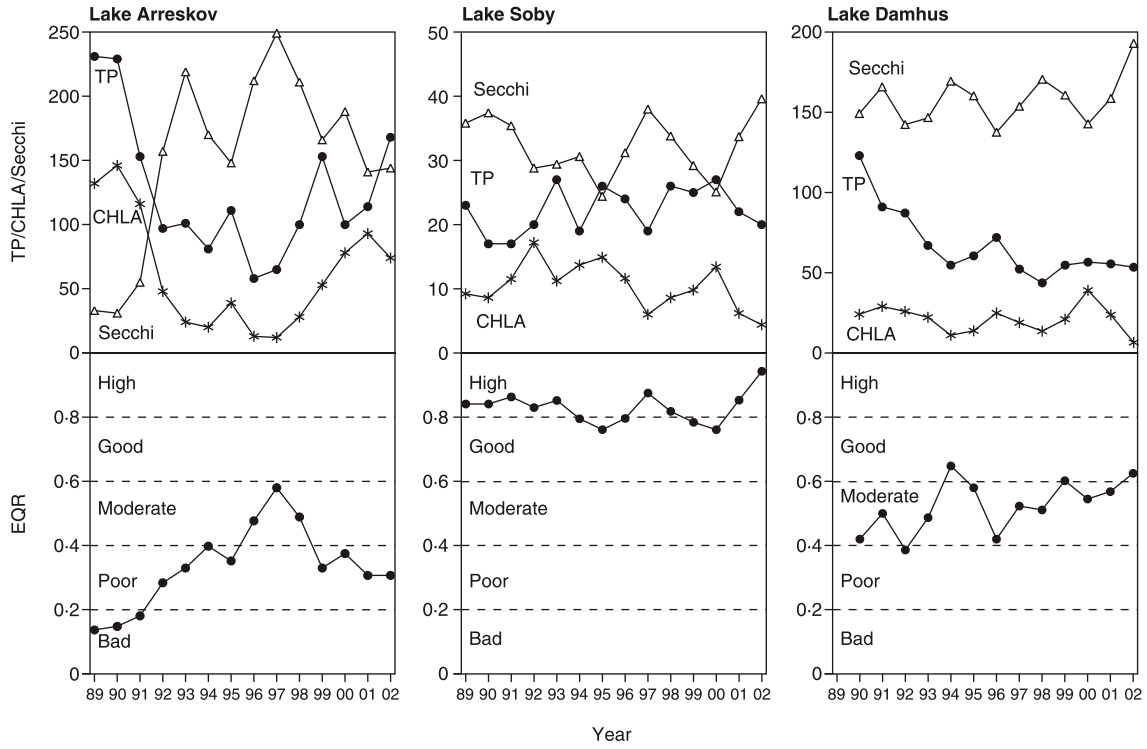



**Fig. 6.** Examples of EQR (lower panels) in three Danish lakes from 1989 to 2002 using summer mean values of 22 indicators (five from each group of indicators: phytoplankton, zooplankton, fish and chemistry, and two from submerged macrophytes) relative to the boundaries given in Table 3. The upper panels show summer mean values of TP (μg P $L^{-1}$), CHLA (μg $L^{-1}$) and Secchi depth (in cm for Lake Arreskov and Lake Damhus and in dm for Lake Soby).

## Discussion

All the selected indicators responded markedly to changes in TP and thus have potential for the classification of lakes relative to eutrophication. The strength of the response varied considerably, however, and for many indicators the difference between high–good and good–moderate were modest. For example, TN did not differ between the two lowest TP classes, probably because some of the Danish low TP lakes receive nitrogen from agricultural areas, thus this boundary should be further elaborated with more data from undisturbed sites. The need to consider TN in the classification of lakes is emphasized by the important role that nitrogen seems to play for the abundance of submerged macrophytes (Moss 2001; Gonzales *et al*. 2005) and the crucial role that submerged macrophytes play in maintaining clear water conditions in shallow lakes (Moss 1990; Scheffer *et al*. 1993; Jeppesen *et al*. 1997; Van Donk & Van de Bund 2002; Jackson 2003). The most promising indicators, identified from the relative change between TP classes, were chrysophyte biovolume, macrophyte coverage, cyanophyte biovolume, macrophyte maximum depth and zooplankton biomass for separating the 25–50 and 50–100 μg P $L^{-1}$ classes, and cyanophyte biovolume, cyclopoid biovolume, chrysophyte biomass, fish numbers and CHLA for separating the 0–25 and 25–50 μg P $L^{-1}$ classes. Indicators that increased with decreasing TP, such as chrysophyte biovolume or macrophyte coverage, should be examined more carefully at low TP, in case their response to TP is unimodal.

High- and low-alkalinity lakes responded relatively similarly to TP. There was, however, a tendency for lower Secchi depth in low-alkalinity lakes, probably because of their often higher humic content (Søndergaard, Jeppesen & Jensen 2005). The clearest effect of alkalinity was found for the relative number of isoetid macrophyte species, which was, as expected, much greater in low-alkalinity lakes as a result of their ability to exploit sediment carbon dioxide (Steeman-Nielsen 1947; Vestergaard & Sand-Jensen 2000). As expected, eutrophication led to strong dominance of elodeids at the expense of isoetids (Moss *et al*. 2003), and elodeid/isoetid abundance thus seems to be a good indicator of eutrophication in low-alkaline lakes. It must be emphasized, however, that lakes included in our study are in the upper end of the low alkalinity gradient of north European lakes. A different response might be found in more softwater lakes. Larger differences between shallow and deep lakes emerged, such as in Secchi depth. This may reflect the higher influence of sediment resuspension in shallow lakes (Nagid, Canfield & Hoyer 2001; Jackson 2003; Jeppesen *et al*. 2003), or at low TP it may reflect the higher coverage of submerged macrophytes, diminishing sediment resuspension (James, Barko & Butler 2004). Correspondingly, significant differences in CHLA between deep and shallow lakes were only observed in lakes with TP above 100 μg $L^{-1}$. Overall, however, in

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the different lake types used in this study most indicators had a relatively similar response to eutrophication, which suggests that the number of lake types can be restricted.

The substantial overlap between adjacent TP classes of almost all indicators constitutes a major problem in defining and separating ecological classes. This reflects the continuum response of biological indicators to increasing TP and also the natural variability (e.g. seasonal and interannual) occurring within a certain nutrient range for many biological variables. For example, CHLA is known to vary relative to algal biomass depending on species, radiation intensity and nutrient availability (Vollenweider & Kerekes 1982; Fennel & Boss 2003), and submerged macrophyte coverage can vary from year to year independently of nutrient loading (Søndergaard *et al*. 1997; Lauridsen *et al*. 2003), with cascading effects on the entire lake ecosystem (Moss 1990; Scheffer *et al*. 1993; Körner 2001; Bayley & Prather 2003). The problem of indicator overlap raises the risk of classifying a lake into the 'wrong' class. Some of the indicators have a 100% overlap between the two lowest TP classes and are therefore obviously poor indicators at low TP concentrations (e.g. macrophyte coverage and zooplankton biomass). One solution is to use fewer ecological classes, for example three comprising high–good, moderate and poor–bad; however, this is (so far) outside the scope of the WFD. It could also be argued that use of a TP gradient is irrelevant as the WFD's ecological classification should be based on biological indicators. Our analyses indicate, however, that, irrespective of the boundaries chosen for a biological indicator, natural variability is high and some lakes will inevitably be assigned to a 'wrong' class.

In an analysis of 66 shallow lakes from 10 European countries using 28 indicators, Moss *et al*. (2003) found that an 80% compliance level was more appropriate than the use of either a 100% or 50% level, i.e. the 'one out – all out' principle, as discussed by the Common Implementation Strategy for the WFD (European Communities 2003), will not be feasible. Our data offer a similar conclusion, but none of the 50–100% compliance levels yielded results comparable with those expected from TP concentrations, particularly when defining the high ecological class and separating the poor and bad ecological classes (Table 5). When using only the three key indicators CHLA, Secchi depth, and fish numbers, a 100% compliance level was still unable to determine the 'right' classes. The best fit was achieved when using a mean value of all the indicators.

A major problem in using multiple indicators for defining ecological classes is the correlation between indicators. For example, TP and TN are closely correlated, as are the numbers of fish and zooplankton biomass. When tracking the same stressor, such as eutrophication, this cannot be avoided, but it should be taken into account in the selection of indicators. When using multiple indicators and a certain compliance level to define ecological class, the weighting of the different indicators should be considered. If, for instance, five indicators are used for phytoplankton and only two for submerged macrophytes, the result will be biased. In our analyses we weighted all indicators equally, but further elaboration is needed and, for example, equal weighting of groups of indicators (phytoplankton, macrophytes, fish, etc.) may be considered. Cost-effectiveness should be considered in relation to the number of indicators used as well. There may also be problems with the timing of sampling, as TP and different indicators may not change synchronously. For example, delayed establishment of macrophyte coverage is often seen with decreasing TP and turbidity because of a lack of seed banks or other factors such as waterfowl grazing (Søndergaard *et al*. 1996; Marklund *et al*. 2002; Lauridsen *et al*. 2003), and a marked delay in response of phytoplankton biovolume and fish communities to reduced TP has also been reported (Hosper 1998; Jeppesen *et al*. 2005).

**Table 5.** 'Expected' ecological classes of 54 lake years (17 shallow lakes) based on the TP concentrations shown in Table 2 (0–25, 25–50, 50–100, 100–200 and > 200 μg P $L^{-1}$) and calculated ecological classes using different methods: 1–6, six levels of compliance (50–100%) of up to 22 indicators; 7, 100% compliance of three selected indicators (chlorophyll *a*, Secchi depth and total number of fish); 8, mean of up to 22 indicators. The latter was calculated as the nearest ecological class when the mean values of all the individual indicators were used by defining values of 1, 2, 3, 4 or 5 for the high, good, moderate, poor and bad ecological classes

| Method | High | Good | Moderate | Poor | Bad |
|---|---|---|---|---|---|
| Expected | 2 | 2 | 8 | 19 | 23 |
| 1 100% (22 indicators) | 0 | 0 | 0 | 1 | 53 |
| 2 90% (22 indicators) | 0 | 0 | 3 | 2 | 49 |
| 3 80% (22 indicators) | 0 | 1 | 3 | 6 | 44 |
| 4 70% (22 indicators) | 0 | 3 | 3 | 5 | 43 |
| 5 60% (22 indicators) | 0 | 3 | 6 | 8 | 37 |
| 6 50% (22 indicators) | 1 | 2 | 6 | 13 | 32 |
| 7 100% (3 indicators) | 0 | 3 | 2 | 2 | 44 |
| 8 Mean (22 indicators) | 0 | 3 | 7 | 25 | 19 |

The WFD stipulates that the ecological quality of a lake is defined by an EQR using values between 0 and 1, where 1 represents the highest ecological quality. However, as emphasized by Moss *et al*. (2003), ecosystems such as lakes do not readily conform to a single formula. This is demonstrated by the high variability of all indictors within a particular TP class. The suggested EQR is fairly robust, as illustrated by the example from the three Danish lakes, as the ratio responds and tracks the changes seen in TP, CHLA and Secchi depth. However, it also demonstrates a shift between two neighbouring quality classes in lakes over time without marked environmental changes. This raises a question regarding whether the classification of lakes should be based on measurements from a single year or whether sampling should compensate for natural interannual variations.

Overall, WFD is a potentially powerful tool to ensure high quality of our aquatic environment. A number of

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indicators respond significantly to TP and are thus potentially useful for classification of lakes. The method suggested for calculating EQR needs further elaboration before being applicable on a European scale, but examples show that a 'reasonable' EQR can be calculated. However, the analyses also reveal a number of difficulties arising from the implementation of WFD, in particular the fact that all indicators responded to eutrophication in a continuous rather than a discrete stepwise manner. This complicates the establishment of well-defined boundaries between quality classes and challenges the idea of using multiple biological indicators, as they may indicate different ecological classes. Another significant problem is how well a rather limited sampling programme based on one or a few annual samplings provides an adequate and correct definition of the ecological class. We used summer averages of typically 5–10 samples and found high variability. Further studies on how to track efficiently the importance of seasonal changes is needed. Nevertheless, the successful implementation of the WFD requires a common understanding of how to interpret the quality of lakes independently of political and local interests, while simultaneously disregarding the fact that in some areas people's view of lakes may have been inured to high levels of eutrophication for a long time (Moss *et al*. 2003).

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# Pond or lake: does it make any difference?

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With 7 figures and 5 tables

**Abstract:** To investigate the importance of lake size, we analysed the chemical and biological characteristics of nearly 800 Danish lakes ranging from 0.01 to 4200 ha. Most of the lakes were shallow (median depth = 1.5 m) and eutrophic (lake water mean total phosphorus = 0.26 mg P $l^{-1}$ and mean chlorophyll-a = 60 µg $l^{-1}$). Phosphorus and nitrogen concentrations were unaffected by lake size, but positively related to agricultural exploitation. Lakes < 1 ha showed a higher variability in phosphorus concentrations, but had a lower chlorophyll yield per unit of both nitrogen and phosphorus, which is indicative of less importance of nutrients in small lakes. Fish were absent in most lakes smaller than 0.1 ha and mean fish biomass was markedly lower in lakes < 1 ha than in lakes > 1 ha. The absence of fish did, however, not result in higher abundance of *Daphnia*, suggesting a higher impact by invertebrate predators in small lakes. Taxon richness of both zoo- and phytoplankton was weakly related to lake size, whereas the number of submerged macrophyte and fish species increased steadily with lake size. Also species richness of macrophytes increased with increasing alkalinity. The low impact of lake size on the species richness of several taxonomic groups suggests that ponds and small lakes are important biodiversity components in the agricultural landscape.

**Key words:** catchment, phosphorus, phytoplankton, fish, zooplankton, macrophytes, biodiversity.

## Introduction

Research into the role of environmental factors such as nutrient loading and other anthropogenic stresses for lake water quality has focused traditionally on relatively large lakes (WETZEL 2001). Small lakes or ponds covering only few hectares or less have received less attention despite their numerical prevalence

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and rich biological diversity (Biggs et al. 1999, Oertli et al. 2002, Williams et al. 2003). In Denmark, for example, lake research has concentrated on lakes > 5 ha, although these constitute only 0.5 % of the 120,000 Danish lakes > 0.01 ha. The existing pond studies have mainly been directed towards specific taxa, such as amphibians and odonates (Oertli et al. 2002), macrophytes (Palmer et al. 1992, Van Geest et al. 2003) or the nutrient retention capacity of wetland ponds (Johnston 1991, Nairn & Mitsch 2000). Thus, while a broad ecological understanding has been gained in recent decades of human influence on large lakes, the overall ecological function of ponds has been less well elucidated (Palik et al. 2001, Tessier & Woodruff 2002). There is no doubt, however, that data from pond systems including interactions between the pelagic and the littoral zone and the benthic-pelagic coupling may provide useful insight into the function of particularly shallow and relatively small lakes. Still, comparative ecological studies along a gradient in lake size are few (e. g. Tonn & Magnusson 1982, Wellborn et al. 1996, Tessier & Woodruff 2002), which renders it difficult to determine the extent to which existing knowledge from large lakes may be applied to small lakes or ponds and vice versa.

The discrimination between large lakes and small lakes or ponds is difficult to establish as the lake size gradient comprises an environmental continuum without any clear delimitation (Wellborn et al. 1996). However, several factors suggest that the two lake types differ. Small lakes and ponds: 1) have closer contact with the adjacent terrestrial environment and a relatively greater littoral zone (Palik et al. 2001) and thus higher terrestrial-aquatic interchange of both organisms and matter; 2) are potentially more isolated from other wetlands and have a more insular nature compared with the often large catchments and riverine inflows of large lakes; 3) exhibit a potential lack of fish owing to winter fish kill and summer dry out. Fish may potentially have strong cascading effects on multiple levels in both larger and small lakes (Wellborn et al. 1996, Jeppesen et al. 1997); 4) exhibit an increased importance of invertebrate predators taking over the role of fish when absent (Yan et al. 1991, Hobæk et al. 2002); 5) have a shallow and wind-protected morphometry, implying that submerged and floating-leaved macrophytes potentially cover large parts of or even the whole lake area under favourable conditions; 6) have relatively stagnant water favouring certain species of flora and fauna and often also a relatively more heterogeneous environment, the overall biological diversity measured per unit of area thus being higher in small lakes and ponds (Gee et al. 1997, Oertli et al. 2002); and 7) have a relatively low water volume and low input of water resulting in enhanced benthic-pelagic coupling and greater impact by the sediment on the water's content of nutrients (Tessier & Woodruff 2002). High benthic-pelagic coupling may explain why phytoplankton is less often limited by phosphorus in small lakes (Barica 1974, Lim et al. 2001, Waiser 2001).

The need for acquiring knowledge has become increasingly obvious following the initiation of a multitude of restoration projects in wetlands and ponds in recent decades with the aim to mitigate the loss of wetlands and to protect flora and fauna, including waterfowl populations (Zedler 2000, Angeler et al. 2003). In Denmark, after a century with dramatically decreasing numbers of particularly small lakes following land reclamation from intensified agriculture and a growing population, about 700 small lakes and ponds are now created yearly (Danish Forest and Nature Agency 2002). In this study we have collated existing morphological, physical, chemical and biological data from 796 Danish lakes of different sizes, ranging from 0.012 ha to 4200 ha. The aim was to elucidate the overall changes in chemical conditions and biological structure along a gradient of lake size.

## Methods

### Study lakes

Survey data from a total of 796 lakes distributed all over Denmark were included in the analyses. Chemical data were available from more lakes than biological data. Only six lakes were > 1000 ha, while 56 were between 100 and 1000 ha, 169 between 10 and 100 ha, 478 between 1 and 10 ha, 55 between 0.1 and 1 ha and 32 were < 0.1 ha. A majority of the lakes were shallow (median mean depth = 1.5 m) and only 10 % had a mean depth > 6 m. Most lakes were situated in intensively agri-cultivated areas with considerable anthropogenic impact.

### Sampling and analyses

Data were collated mainly by the local counties using standard sampling techniques and analyses (Kronvang et al. 1993). For lakes smaller than 5 ha, sampling was typically conducted once or only a few times during the summer season, while most larger lakes were sampled once monthly or more frequently during summer. In the latter case, mean summer values (1 May–1 October) were calculated for each year and in case of data from several years, these were averaged to obtain one value for each lake.

Water for chemistry and plankton analyses was collected as surface samples from a mid-lake station. Water used for chemical analyses of dissolved forms was filtered on Whatman GF/C-filters. Chemical parameters and chlorophyll-a (CHLA) were analysed according to standard procedures (Jespersen & Christoffersen 1987, Søndergaard et al. 1992). Organic-bound nitrogen (Org-N) was calculated as the difference between total nitrogen (TN) and the inorganic nitrogen fractions [ammonia ($NH_4$) and nitrate + nitrite ($NO_3$)]. Quantitative measurements of the fish stock were expressed as CPUE (catch per unit effort) using standardised 42 m long multiple mesh-sized gill nets with 14 different mesh sizes, ranging from 6.25 mm to 75 mm (Mortensen et al. 1991, Jeppesen et al. 2004). The nets were typically set in late afternoon and retrieved the following morning after 18 hours, except for lakes < 0.5 ha, where the nets were

retrieved after one hour to avoid a significant reduction in the fish stock. In lakes > 5 ha, 6–24 nets were used, while one or a few nets were used in lakes < 5 ha. Zooplankton and phytoplankton were fixed in Lugol's iodine and identified down to the lowest feasible taxonomic level, usually genus or sometimes species level. Presence/absence, and in some lakes also relative abundance (%-coverage of lake area), of submerged macrophytes were recorded during maximum abundance in July or August. Coverage was grouped into the following categories: 0–1, 1–5, 5–25, 25–50, 50–75 and 75–100 % of total lake area.

### GIS and statistics

GIS (Geographical Information System) data were used to categorize lakes of different sizes and to relate lake data with land use characteristics in the nearest surroundings. As an example representing a typical geological and land use landscape of Denmark, this analysis was conducted on data from the island of Funen only (2,985 $km^2$ representing 7 % of Denmark), including approximately 11,000 lakes ranging in size from 0.01 to 317 ha. GIS data were used to define land use within a distance of 25, 50, 100 and 500 m of lake shores, and the results were subsequently correlated with water quality variables. Land use was classified according to five major categories: residential, cultivated, pasture/forest (natural grassland, forest, heath), wetlands (bogs, meadows, etc.) and "others".

Statistical analyses were performed using SAS (SAS Institute 1989). Canonical correspondence analyses (CCA) on submerged macrophyte communities and environmental variables were performed using SAS statistics and conducted according to TER BRAAK & SMILAUER (1998). We only conducted CCA analysis on submerged macrophytes, however, as complete data sets for other biological variables (zooplankton, fish and phytoplankton) were too scarce, particularly for small-sized lakes. We also performed linear regression using proc GLM or proc REG and proc NLIN for non-linear regression. Regression analyses were performed on log-transformed data.

## Results

### Physical and chemical variables

Mean depth in lakes increased significantly with increasing lake area, from about 1 m in the smallest lakes to 3 m in the largest lakes (Fig. 1, Table 1). Water colour decreased steadily from a median value of 150 mg Pt $l^{-1}$ in lakes < 0.1 ha to about 20 mg Pt $l^{-1}$ in lakes > 100 ha. pH increased by about one unit from the smallest to the largest lakes, while silica and total alkalinity only varied slightly along the size gradient. Both Secchi depth and suspended solid concentrations increased significantly with increasing lake size, but the correlation was weak.

A majority of the lakes were eutrophic and had a total phosphorus concentration (TP) above 0.1 mg P $l^{-1}$ (Fig. 2). Neither TP, soluble reactive phospho-

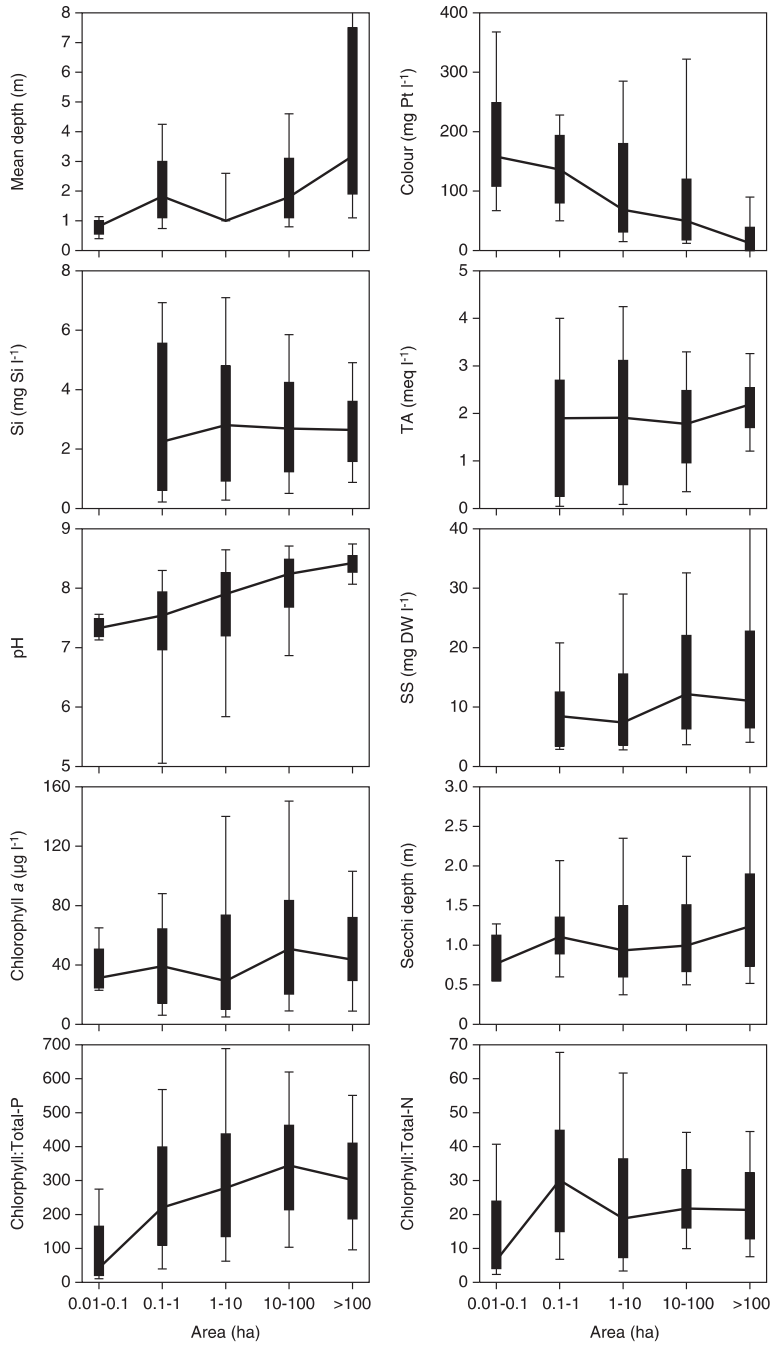



**Fig. 1.** Physical and chemical variables along a lake size gradient shown as box-plots. Each box shows 25 and 75 % percentiles, the horizontal line the mean value, and the top and bottom of the thin line depict the 90 and 10 % percentiles.

**Table 1.** Linear regression between physical-chemical characteristics and lake area and depth (on log transformed data). The right part of the table shows a multiple regression including both area and depth (proc REG, forward selection, entry level = 0.1). N = number of lakes, p = level of significance and $r^2$ = coefficient of determination with + or −, indicating a positive or negative relationship. For the multiple regression $R^2$_a and $R^2$_d are the partial correlation coefficients for area and depth, respectively.

| | Area | | | Depth | | | Area & depth | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| Variable | N | p | $r^2$ | N | p | $r^2$ | N | p | $R^2$_a | $R^2$_d |
| Mean depth | 698 | <0.001 | +0.25 | – | – | – | – | – | – | – |
| Colour | 187 | <0.001 | −0.10 | 187 | 0.001 | −0.10 | 171 | 0.001 | 0.13 | – |
| TP | 777 | 0.48 | – | 683 | 0.007 | −0.01 | 682 | 0.007 | 0.01 | 0.01 |
| SRP | 579 | 0.63 | – | 488 | 0.38 | – | 487 | >0.1 | – | – |
| SRP/TP | 576 | 0.07 | – | 486 | 0.001 | +0.04 | 485 | 0.001 | – | 0.04 |
| Total N | 775 | 0.04 | +0.01 | 681 | 0.02 | −0.01 | 680 | 0.02 | 0.02 | 0.01 |
| $NO_3$ | 602 | <0.001 | +0.13 | 510 | 0.001 | +0.04 | 508 | 0.001 | 0.14 | – |
| $NH_4$-N | 571 | 0.04 | +0.01 | 479 | 0.33 | – | 478 | >0.1 | – | – |
| Si | 454 | 0.07 | – | 393 | 0.009 | −0.02 | 392 | 0.009 | 0.03 | 0.02 |
| TA | 641 | 0.03 | +0.01 | 556 | 0.20 | – | 556 | 0.027 | 0.01 | – |
| pH | 714 | <0.001 | +0.05 | 627 | 0.007 | +0.01 | 626 | 0.001 | 0.07 | – |
| Susp. solids | 566 | <0.001 | +0.05 | 513 | 0.049 | −0.01 | 512 | 0.001 | 0.05 | 0.06 |
| CHLA | 702 | <0.001 | +0.02 | 627 | 0.11 | – | 626 | 0.001 | 0.02 | 0.03 |
| CHLA : TP | 699 | 0.001 | +0.01 | 624 | 0.51 | – | 623 | 0.002 | 0.02 | – |
| CHLA : TN | 699 | 0.01 | +0.01 | 624 | 0.56 | – | 623 | 0.010 | 0.01 | 0.01 |
| Secchi depth | 689 | 0.004 | +0.01 | 615 | 0.001 | +0.16 | 614 | 0.001 | 0.01 | 0.16 |

rus (SRP) nor the SRP : TP ratio changed significantly with lake size, but the upper 10 and 25 % fractiles of TP were higher in the smallest lakes. TN increased slightly with lake size, while nitrate + nitrite concentrations ($NO_3$) differed, particularly between lakes < 10 ha and lakes > 10 ha, mean $NO_3$ being 0.57 mg N $l^{-1}$ (SD = 1.2, n = 384) and 1.26 (SD = 3.2, n = 220), respectively. Ammonia increased slightly with lake size, but very high concentrations were found mostly in lakes < 1 ha. Correspondingly, Org-N was particularly low in lakes < 0.1 ha. The TN : TP ratio did not vary along a size gradient.

In a multiple regression including both depth and area, some of the chemical variables like suspended solids, Secchi depth, TP, TN and silica (Si) correlated significantly to both area and depth; the correlation was generally weak, however (Table 1). For Secchi depth the correlation was much stronger to depth (partial $R^2$ = 0.16) than to area (partial $R^2$ = 0.01). Some of the variables, colour, $NO_3$, pH, total alkalinity (TA) and the CHLA : TP ratio related significantly only to area.

In the multiple regression using TP, TN, area and depth as explainable variables, particularly suspended solids and CHLA were highly correlated ($r^2$ = 0.41–0.43) to TP (Table 2). The correlation to Secchi depth was negative, but this result should be interpreted with caution as Secchi depth cannot be

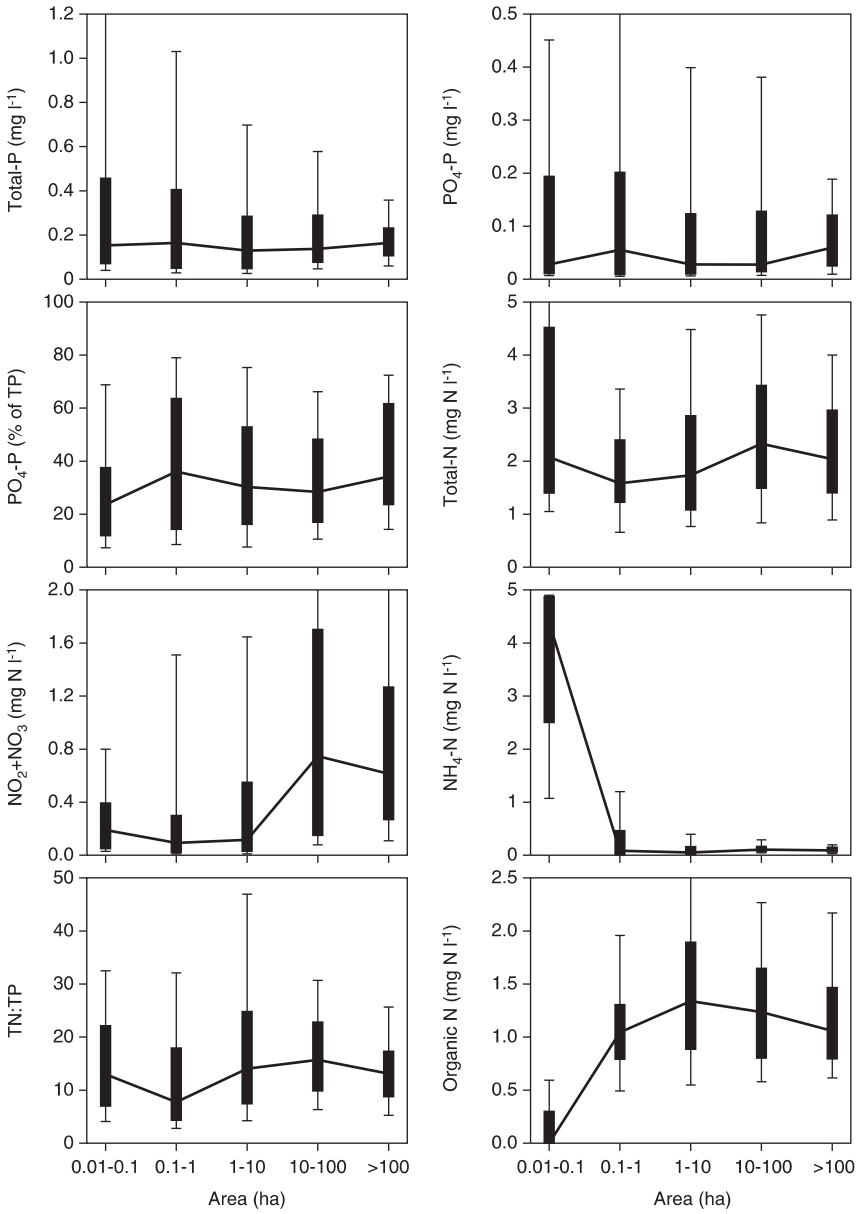



**Fig. 2.** Nitrogen and phosphorus concentrations along a lake size gradient shown as box-plots. See also legend to Fig. 1.

measured correctly in many of the shallow lakes (> maximum depth). TN, area and depth were significantly related to CHLA and suspended solids also, but the correlation was generally weak. Inclusion of water colour reduced the

**Table 2.** Multiple regression between CHLA, suspended solids, pH, Secchi depth and TP, TN, lake area and lake depth (on log transformed data, proc REG, forward selection, entry level = 0.1). N = number of lakes, p = level of significance and $R^2$ = multiple coefficient of determination and $r^2$ = coefficient of determination (for the model or as the partial correlation coefficient) with + or −, indicating a positive or negative relationship.

| Variable | N | Model | TP | | TN | | Area | | Depth | |
|---|---|---|---|---|---|---|---|---|---|---|
| | | $R^2$ | p | $r^2$ | p | $r^2$ | p | $r^2$ | p | $r^2$ |
| CHLA | 623 | 0.47 | 0.001 | +0.43 | 0.001 | +0.02 | 0.001 | +0.01 | 0.028 | −0.00 |
| Susp. solids | 512 | 0.48 | 0.001 | +0.41 | 0.001 | +0.04 | 0.001 | +0.02 | 0.001 | −0.02 |
| pH | 611 | 0.13 | 0.001 | +0.07 | >0.1 | – | 0.001 | +0.06 | >0.1 | – |
| Secchi depth | 596 | 0.46 | 0.001 | −0.31 | 0.001 | −0.02 | >0.1 | – | 0.001 | +0.12 |

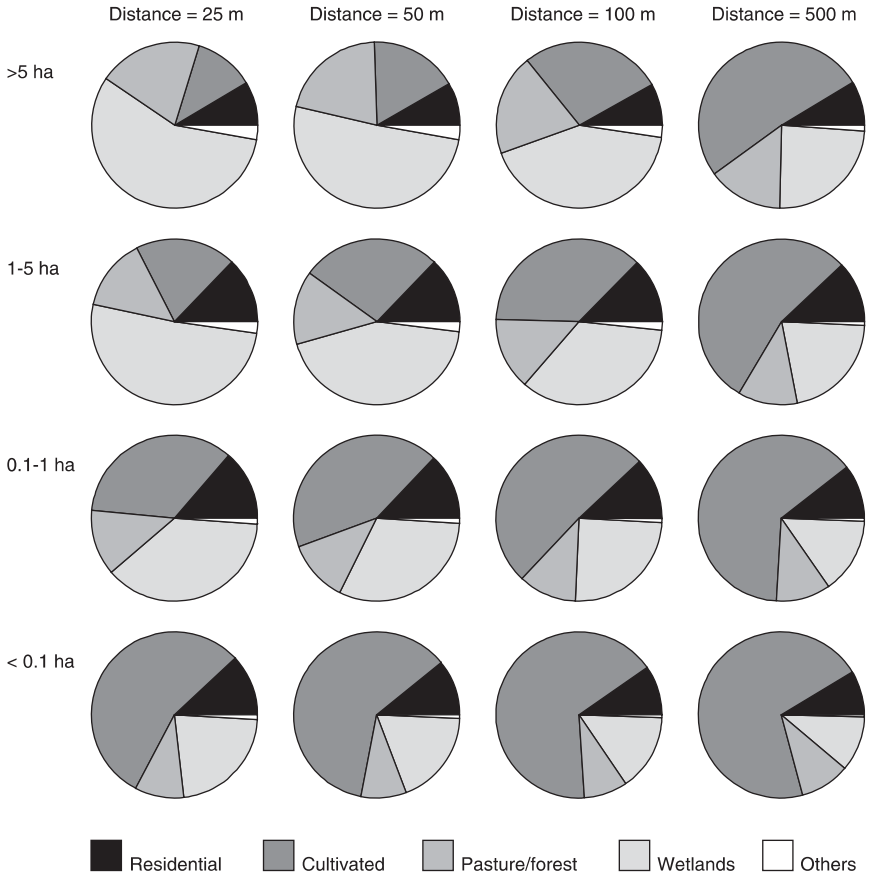



**Fig. 3.** Land use in catchments of all lakes and ponds on the island of Funen (total number of lakes = ca. 11,000). The lakes are divided into 5 size classes and the calculations performed for 4 zones surrounding the lakes (25, 50, 100 and 500 m). Data from the AIS system (NIELSEN et al. 2000).

**Table 3.** Correlation analyses (Spearman) between land use (within 25 m of the lake) and lake water measurements from lakes on the island of Funen. r = correlation coefficient with + or –, indicating a positive or negative relationship and p = level of significance.

| Variable | TP | | TN | | CHLA | | Secchi | | Alkalinity | | pH | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | p | r | p | r | p | r | p | r | p | r | p | r |
| Residential | 0.005 | +0.11 | +0.21 | – | +0.76 | – | +0.11 | – | <0.001 | +0.32 | <0.001 | +0.36 |
| Cultivated | <0.001 | +0.16 | +0.002 | +0.12 | +0.04 | +0.08 | +0.79 | – | <0.001 | +0.36 | <0.001 | +0.32 |
| Wetlands | <0.001 | +0.22 | <0.001 | +0.20 | <0.001 | +0.20 | <0.001 | −0.26 | +0.44 | – | +0.44 | – |
| Pasture/forest | <0.001 | −0.26 | <0.001 | −0.22 | <0.001 | −0.17 | <0.001 | 0.27 | <0.001 | −0.33 | <0.001 | −0.36 |

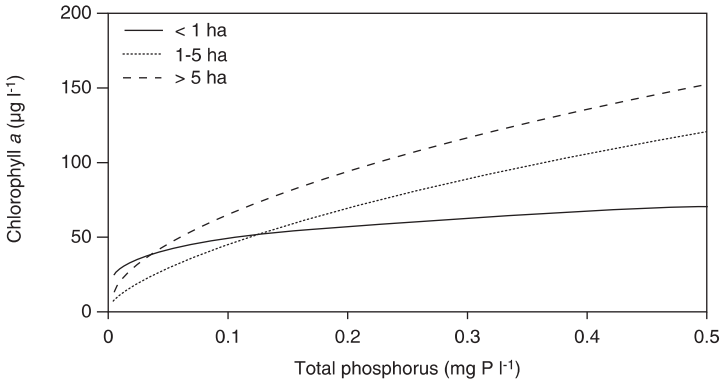



**Fig. 4.** The relationship between total phosphorus and chlorophyll-a in lakes of different areas. Parameter estimates (proc nlin, SAS) using CHLA = a $x$TP$^b$: lake area < 1 ha (n = 41): a = 57, b = 0.18; lake area = 1–5 ha (n = 363): a = 211, b = 0.64; lake area > 5 ha (n = 236): a = 125, b = 0.41.

number of lakes to 129, but CHLA was not significantly (p > 0.1) related to colour in a multiple regression including also TP, TN, area and water depth.

### Land use around the lakes and its relation to nutrient concentrations

Cultivated areas and wetlands were the most common land types surrounding the lakes and ponds (Fig. 3), and the proportion of cultivated land increased from 10–50 % for lakes > 5 ha to between 55 % and 70 % for lakes < 0.1 ha. For all size classes, the cultivated share increased with increased size of the surrounding zone. This pattern was most clear for lakes > 1 ha, and for lakes > 5 ha the percentage of cultivated land increased from 10 % if a 50 m zone around the lake was included, to 50 % using a 500 m zone. Corresponding to the increased share of agricultural areas, particularly the size of wetland area decreased. In a single factor analysis TP, TN, CHLA, total alkalinity (TA) and pH were significantly and positively related to the agricultural exploitation of the catchment zone, but negatively so to the share of pasture/forest (Table 3).

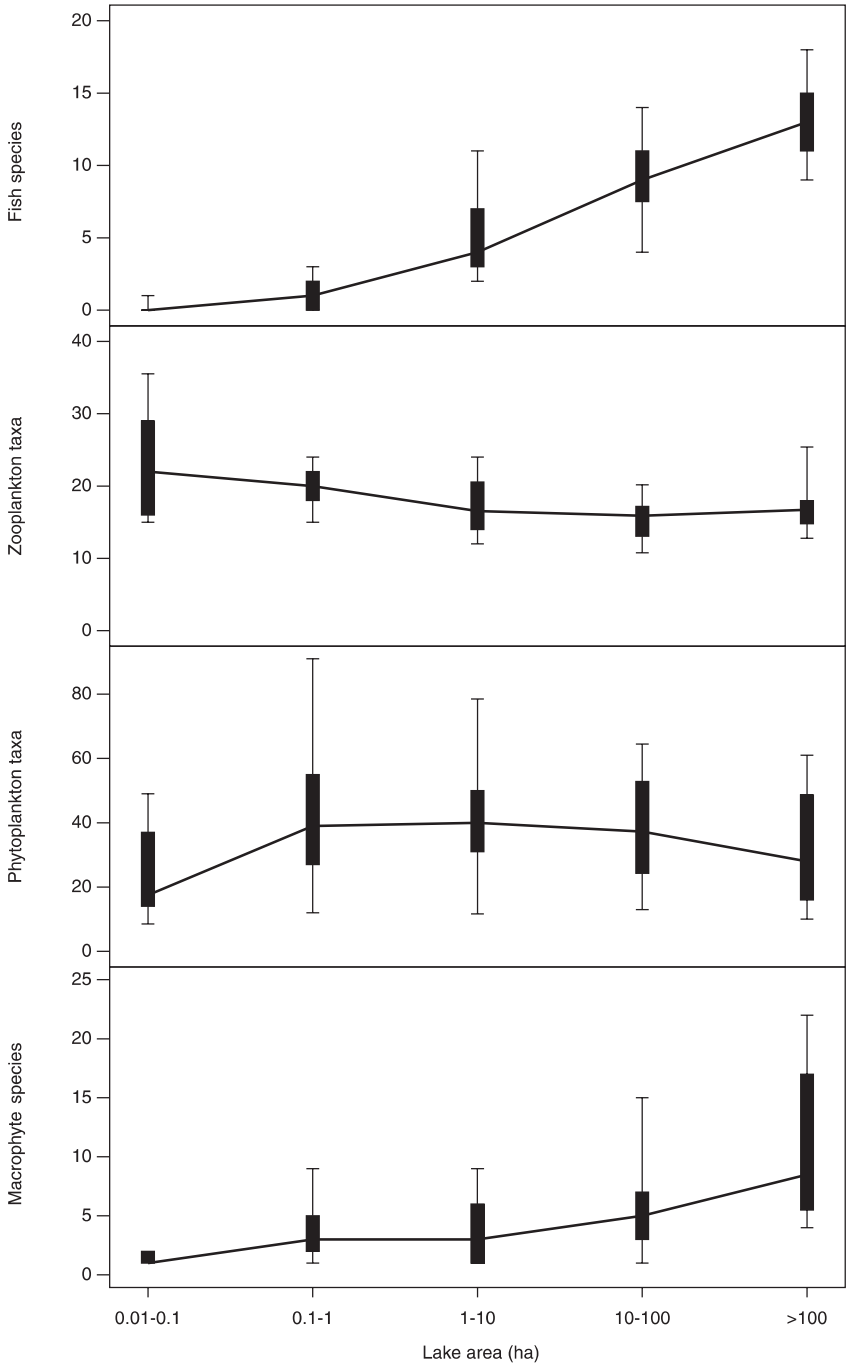



**Fig. 5.** Species and taxa numbers of fish, zooplankton, phytoplankton and submerged macrophytes along a lake size gradient shown as box-plots. See also legend to Fig. 1.

### Phytoplankton

CHLA varied considerably, but increased significantly with lake size. The CHLA : TP and CHLA : TN ratios both correlated significantly and positively with lake area, and the ratios were markedly lower in lakes < 0.1 ha than in larger lakes (Fig. 1). The overall lower CHLA in the smaller lakes was also reflected in a non-linear model relating CHLA to TN and TP (Fig. 4).

The number of phytoplankton taxa recorded ranged between 20 and 40 in most lakes (Fig. 5). Highest numbers occurred in lakes between 1 and 100 ha, with median numbers about 40, but only 20 in lakes < 0.1 ha and 30 in lakes above 100 ha. The number of taxa was significantly but weakly ($p < 0.007$, $R^2 = 0.06$, $n = 363$) unimodally related to both area and TP.

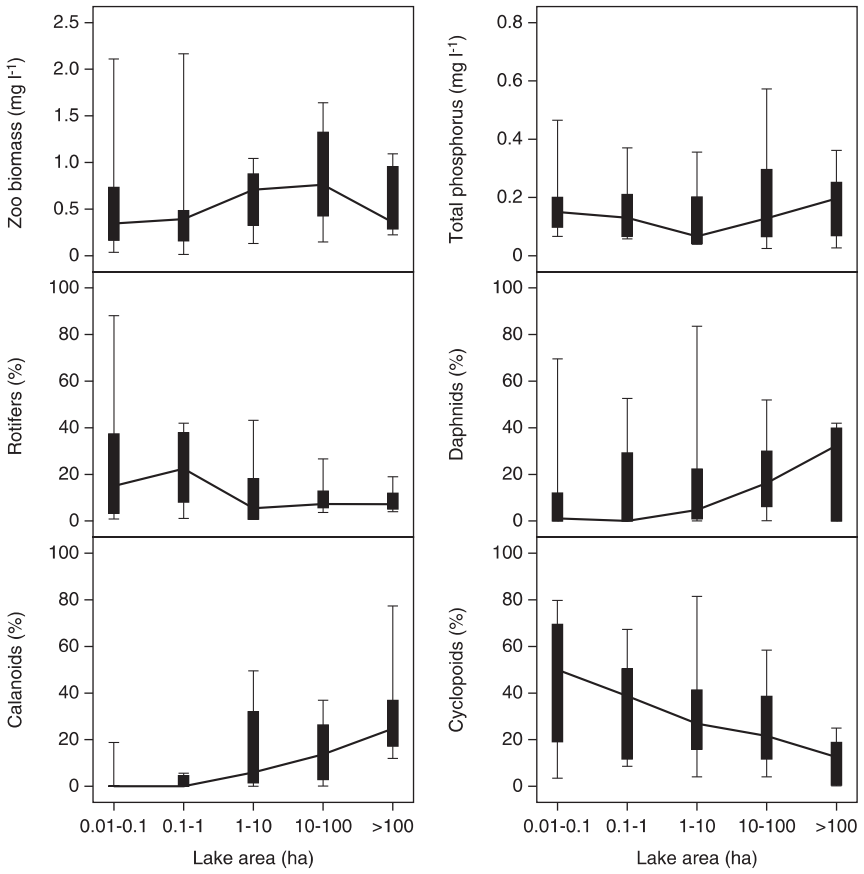



**Fig. 6.** Zooplankton biomass, relative biomass proportion of cyclopoid copepods, rotifers, *Daphnia* spp. and calanoid copepods relative to lake area shown as box-plots. See also legend to Fig. 1.

### Zooplankton

Zooplankton biomass was relatively unaffected by lake size, but tended to be lowest in the smallest lakes (Fig. 6). In a multiple regression (n = 116), biomass was significantly ($p < 0.0001$, $R^2 = 0.26$) positively related to TP ($p < 0.0001$) and depth ($p < 0.002$). Taxon richness of zooplankton in the lakes ranged typically between 15 and 25, with a tendency to a higher richness in lakes smaller than 1 ha (Fig. 5). Thus, in a multiple regression taxon richness was weakly negatively related to lake area ($p < 0.0001$, $R^2 = 0.15$, n = 121), while TP and mean depth were not. The share of cyclopoid and calanoid copepods of the total biomass decreased and increased, respectively, with increasing lake size, but independently of TP and depth. In a multiple regression the share of cyclopoids was significantly ($p < 0.0001$, $R^2 = 0.22$, n = 116) negatively related to area ($p < 0.04$) and depth ($p < 0.01$) and positively so to TP ($p < 0.03$). By contrast, the shares of *Daphnia*, rotifers and calanoids were related to area only. The share of *Daphnia* ($p < 0.003$, $R^2 = 0.22$, n = 116) and cala-

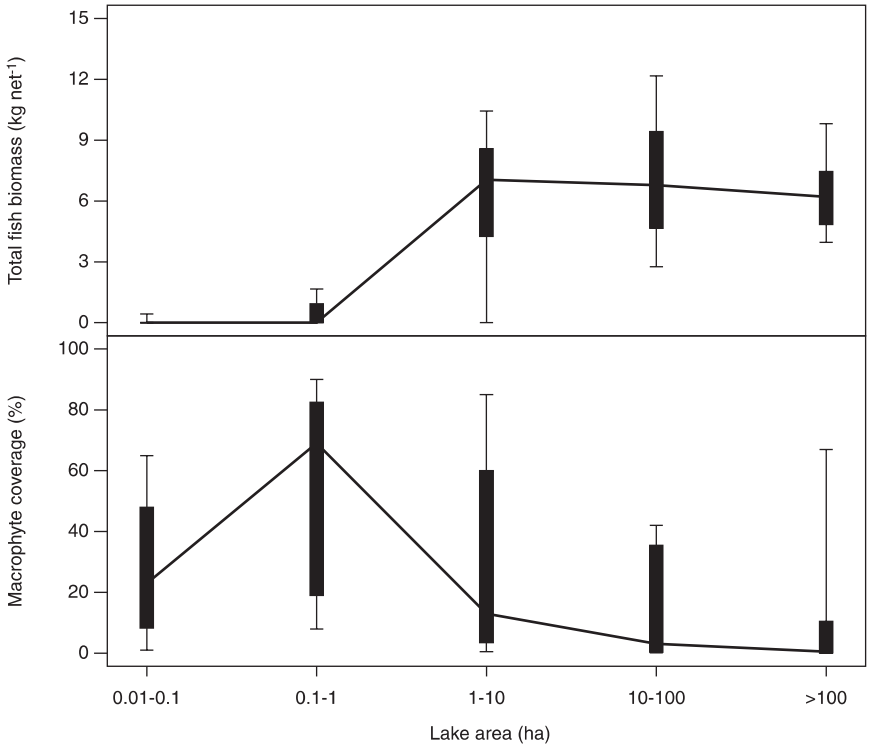



**Fig. 7.** Total biomass of fish (CPUE, n = 113) and coverage of submerged macrophytes (% of total lake area, n = 132) in relation to lake area shown as box-plots. See also legend to Fig. 1.

noids ($p < 0.0001$, $R^2 = 0.16$) increased slightly with area and rotifers ($p < 0.0001$, $R^2 = 0.16$) decreased.

### Fish

Most of the lakes < 0.1 ha were fishless and the median number of fish species recorded in lakes between 0.1 and 1 ha was only about one as many lakes in this category were also fishless (Fig. 5). Fish richness increased steadily up to a median number of 12 species in lakes > 100 ha. A multiple regression revealed that both area ($p < 0.0001$), mean depth ($p < 0.04$) and TP ($p < 0.03$) contributed significantly and positively to species richness of fish ($R^2 = 0.79$, n = 86). Correspondingly, total catch by standard fishing with multiple mesh-sized gill nets [catch per unit effort (CPUE), weight-based] was generally low in lakes < 1 ha, but increased steeply in the category 1–10 ha, the values recorded for many of these lakes still being low, however (Fig. 7). For lakes between 10 and 100 ha, CPUE values were similar to those of large lakes. Using all data in a multiple regression, CPUE was strongly linked to area ($p < 0.0001$) and to TP ($p < 0.002$, $R^2 = 0.70$, n = 109), but if only lakes larger than 10 ha are considered, only TP remains significant ($p < 0.0004$, $R^2 = 0.15$, n = 83). The most common fish species recorded were roach (*Rutilus rutilus*), perch (*Perca fluviatilis*), pike (*Esox lucius*), rudd (*Scardinius erythrophalmus*), bream (*Abramis brama*) and eel (*Anguilla anguilla*).

### Submerged macrophytes

The number of macrophyte species ranged from 0 to 23 (Fig. 5). Species number was highest in the largest lakes, increasing from a mean number of 1.1 in lakes between 0.01 and 0.1 ha to 10.6 in lakes larger than 100 ha. (Table 4). Apart from area, the distribution of macrophytes was particularly ordered along an alkalinity and a depth gradient (Table 5). Species number was lower in lakes with low alkalinity than in lakes with high alkalinity: number of macrophyte species = $2.6\ x\ \text{area}^{0.16}$ in lakes with alkalinity below 0.2 meq $l^{-1}$ and $2.6\ x\ \text{area}^{0.24}$ in lakes with alkalinity above 0.2 meq $l^{-1}$ (proc NLIN, SAS).

**Table 4.** Estimated number of submerged macrophyte species in lakes at two levels of alkalinity (TA) and at three levels of total phosphorus (TP). Number of species = a $x$ area$^b$, where area = lake area in ha. Proc NLIN, SAS was used.

| TP | TA ≤ 0.2 meq $l^{-1}$ (n = 59) | | TA > 0.2 meq $l^{-1}$ (n = 87) | |
|---|---|---|---|---|
| | a = | b = | a = | b = |
| 0–25 µg P $l^{-1}$ | 4.26 | 0.23 | 3.60 | 0.33 |
| 25–50 µg P $l^{-1}$ | 3.01 | 0.16 | 3.13 | 0.30 |
| 50–100 µg P $l^{-1}$ | 1.97 | 0.19 | 2.55 | 0.11 |

**Table 5.** CCA-analyses of submerged macrophytes using the forward selection procedure. F ratio is the sum of all canonical eingenvalue and λ the eigenvalue.

| Parameter | λ | F-ratio | P value |
|---|---|---|---|
| Area | 0.47 | 1.91 | 0.049 |
| Alkalinity | 0.42 | 1.76 | 0.006 |
| Chlorophyll-a | 0.28 | 1.2 | 0.229 |
| Mean depth | 0.33 | 1.43 | 0.047 |
| Total nitrogen | 0.23 | 0.98 | 0.495 |
| Total phosphorus | 0.1 | 0.4 | 0.986 |

The relative coverage of submerged macrophytes was highest in lakes < 10 ha, with a mean coverage of 31 %, 52 % and 35 % in lakes < 0.1, 0.1–1 and 1 to 10 ha, respectively (Fig. 7). In lakes between 10 and 100 ha coverage decreased to 7 % and to 3 % in lakes > 100 ha. There was a weak, but significant relationship between TP and coverage ($p = 0.04$, $R^2 = 0.06$).

## Discussion

In the vast majority of the study lakes, the nutrient concentrations were high and well above the levels expected to occur in lakes situated in natural areas without anthropogenic influence. They were, however, comparable with levels found in other studies of lakes affected by agricultural run-off (Bennion & Smith 2000, Nairn & Mitch 2000, Schell et al. 2001). The increasing agricultural dominance of lake surroundings found with decreasing lake size emphasises that small lakes in the agricultural landscape have a high risk of impact from nearby farming activities. This is also indicated by the positive relationship found between land use and nutrient concentrations in the lakes. Therefore, nutrient concentrations in small lakes and ponds in an agricultural landscape can be strongly impacted by catchment activities, even though they are often devoid of surface inflows.

Many of the small lakes and ponds had high nutrient concentrations, particularly of total phosphorus and ammonia (Fig. 2). Similarly, Bennion & Smith (2000) in a study of shallow ponds in south-east England found high interannual variability in phosphorus, which tended to be highest in the most enriched water. Possibly, this reflects the high impact by the sediment on seasonal nutrient concentrations, which tend to be most important in eutrophic and shallow waters with a large sediment to water interface (Waiser 2001, Søndergaard et al. 2003). The importance of internal processes is high in small lakes as these usually have no surface outflows and all phosphorus entering the lakes will be retained and potentially recycled within the lake. For nitrogen, the higher nitrate concentrations recorded in lakes > 10 ha likely re-

flect higher hydraulic loading, including surface inflows rich in nitrate, and the fact that nitrogen retention is strongly affected by the hydraulic retention time (OECD 1982). Thus, nitrogen in small lakes with low hydraulic flushing will eventually be removed from the systems through denitrification. This might also explain why small lakes generally have higher submerged macrophyte coverage than larger lakes, as also found by VAN GEEST et al. (2003). Low nitrogen concentrations have a positive impact on the potential presence of submerged macrophytes and their biodiversity (MOSS 2001, SAGRARIO et al. 2005). Thus, macrophytes in small lakes may benefit from a small catchment, because of low nitrogen concentrations.

The generally high phosphorus concentrations of both TP and SRP in lakes < 1 ha was not reflected in the chlorophyll concentrations, and the chlorophyll-a yield per unit of phosphorus was particularly low in lakes < 1 ha. Correspondingly, pH decreased markedly with lake size, which indicates reduced primary productivity as alkalinity was almost unaffected by lake size. In some small lake types, low CHLA may relate to reduced light availability because of shading from trees (PALIK et al. 2001) or the higher humic content generally seen in the smaller lakes (KLUG 2002); however, CHLA did not correlate significantly with water colour in the multiple regression including also TP, TN lake area and depth. Strong control of phytoplankton by zooplankton in the absence of fish, as observed in large lakes at low zooplanktivorous fish abundance (MAZUMDER 1994, MAZUMDER & HAVENS 1998, QUIROS 1998) might also be an important factor. This effect could be even further enhanced by the high coverage of macrophytes in the smallest lakes. Thus, LEIBOLD (1999) found a positive relationship between the densities of zooplankton and plants in 31 fishless ponds, and PATERSON (1993) recorded large numbers of microcrustaceans in association with macrophytes. Our zooplankton data do not, however, support the view that zooplankton has any strong control on phytoplankton, as indicated by low biomass in the small lakes, although we cannot exclude that zooplankton biomass was underestimated as sampling was undertaken during the day (BURKS et al. 2002). A high grazing impact and filtration rate may alternatively arise from sessile filtrators or macrophyte associated filter-feeding microcrustaceans, such as *Sida* (STANSFIELD et al. 1997), or macroinvertebrates when macrophytes are present and fish absent (BRÖNMARK & VERMAAT 1998). We have no data to support this, but the generally higher coverage of submerged macrophytes in lakes < 10 ha implies a higher potential influence. Although the mechanisms are not fully elucidated, our findings overall suggest a lesser nutrient control on phytoplankton abundance in small lakes compared with larger lakes, and support the view that major shifts in the functional coupling of grazers and phytoplankton occur across a size gradient (TESSIER & WOODRUFF 2002).

In the absence of fish, invertebrate predators may become an important factor structuring the food web, which may explain why we found a relatively low zooplankton biomass also in the small lakes normally without fish. For example, Hobæk et al. (2002) in a study of 36 mainly small Norwegian lakes found that lakes without pelagic fish predators had a distinct zooplankton assemblage normally confined to ponds and that these lakes appeared to be dominated by the predatory phantom midge *Chaoborus*. Similarly, Yan et al. (1991) found that *Chaoborus* regulated the zooplankton communities in acidified fish-free lakes. Other invertebrates, such as notonectids, may also contribute to reduce zooplankton abundance (Shurin 2001, Steiner & Roy 2003). Cascade-like effects caused by day-night migration by zooplankton, as usually recorded in the presence of fish, have been observed also in fishless, small lakes in the presence of a species of predatory backswimmers (*Buenoa* sp.) (Gilbert & Hampton 2001).

The occurrence of alternative predators in the small fishless Danish lakes is supported by the zooplankton composition. Although poor *Daphnia* performance in fishless ponds may also owe to abiotic conditions and resource effects (Steiner & Roy 2003), the proportion of cyclopoid copepods and rotifers was high and low for *Daphnia*, which normally indicates high predation pressure. This suggests that invertebrate predators play a significant role in the fishless small lakes, which compensates for the lack of fish predation. Furthermore, the function of submerged macrophytes as a refuge for large-bodied zooplankton shown from larger lakes (Burks et al. 2002) may be less important in small lakes, as indicated by Burks et al. (2001) who showed that in the presence of dragonfly nymphs (*Epitheca cynosura*), *Daphnia* were effectively eliminated within 24 hours regardless of macrophyte presence. An alternative explanation of the low proportion of *Daphnia* may be diel vertical migration, where *Daphnia* hide near the bottom or in the littoral zone during the day to avoid predators, as has been observed in other fishless ponds (Gilbert & Hampton 2001), or that low water depth in small lakes leads to higher predation, as the predation risk tends to increase with declining depth (Jeppesen et al. 1997).

The absence of fish or low biomass in the smallest lakes and their presence in larger lakes is probably the single-most important structuring factor for changes observed in the biological communities along the size gradient. Wellborn et al. (1996) termed this transition between permanent fishless habitats and habitats with fish "predator transition", because very distinct community types are produced. In our study, most lakes with an area < 0.1 ha were without fish. Similarly, an investigation of 20 ponds < 0.1 ha located in western Denmark showed that fish were present in only 17 % of the ponds (E. Kanstrup, County of Ringkjøbing, unpubl. results), and in another Danish investigation including 83 ponds (mean size 0.06 ha, range: 0.0025 – 0.34 ha) Henriksen (2000) found fish in only 8 % of the ponds.

Water depth is probably the most important factor regulating fish survival during cold winters or during droughts (Tonn & Magnusson 1982), and a cold winter or a dry summer may have a long-lasting effect in small lakes and ponds. As the lakes in our study also include a depth gradient parallel to the size gradient, it is difficult to disentangle the role of these two variables. However, except for Secchi depth and the SRP : TP ratio, the chemical variables were related better to area than to depth, suggesting area to be a primary factor. Data on biological variables are much more limited, but for submerged macrophytes area also seems more important than lake depth. The presence/absence of fish and the rapidity of colonisation of fish or other organisms following a fish kill also depend highly on the extent of the lake's contact with other wetlands and the frequency of dispersal events (Pont et al. 1991, Shurin 2001, Hobæk et al. 2002, Cohen & Shurin 2003). Even relatively small ponds may hold a fish stock if the spreading potential from adjacent wetlands and streams is favourable. Especially fast colonizers such as three-spined stickleback (*Gasterosteus aculeatus*) are known to rapidly invade new wetlands where they quickly reach a significant population size (Berg & Mæhl 1998). Sticklebacks often have a highly negative impact on large-sized zooplankton in previously fish-free ponds (Pont et al. 1991). In lakes with frequent occurrence of winter kill, connectedness is also important for the fish structure (Tonn & Magnusson 1982). The small lakes included in our study exhibiting the greatest species number (7–8 species) were lakes connected with other lakes via streams (Søndergaard et al. 2002).

Species richness relative to lake size has been a frequent subject of debate, and the general finding is that species richness increases along a size gradient according to island biogeographic predictions (Tonn & Magnusson 1982, Dodson 1992, Allen et al. 1999, Oertli et al. 2002). Other factors like lake depth (Keller & Conlon 1994), pelagic primary productivity (Dodson et al. 2000) or phosphorus concentrations (Jeppesen et al. 2000) may also influence species richness and often exhibit unimodal relationships (Dodson et al. 2000). Overall, our study showed that taxon richness of zoo- and phytoplankton varied only slightly along a size gradient, whereas species richness of fish and submerged macrophytes increased markedly with lake size, as shown in other studies (Amarasinghe & Welcomme 2002, Bazzanti et al. 2003).

The weak effect of lake size on zooplankton taxon richness is in accordance with Schell et al. (2001), whereas Dodson et al. (2000) found a significant positive relationship between lake area and species richness of rotifers and cladocerans. However, the latter study only included five lakes < 10 ha and might not be comparable with the lakes in our study. Cottenie & De Meester (2003) have suggested that local environmental variables related to the clear-water/turbid state alternative equilibria are more important for cladoceran species richness than connectivity of ponds. The weak area dependency for

phytoplankton diversity is in accordance with the finding of Dodson et al. (2000) and to Rojo et al. (2000), who suggested that phytoplankton dynamics are more complex in ponds than in lakes owing to the large number of interacting factors.

For submerged macrophytes, area was the most important factor explaining species richness, but alkalinity was also important as in other mainly larger lakes (Friday 1987, Rørslett 1991), explained by increased occurrence of elodeids with increasing alkalinity (Vestergaard & Sand-Jensen 2000). Like fish, macrophyte species richness may also be influenced by the distance between ponds, as shown for vascular plants in a study by Møller & Rørdam (1985) and a study of neighbouring waterbodies by Linton & Goulder (2003).

The on-off appearance of fish in small lakes may for specific taxonomic groups challenge the expected increased richness with increasing size. Thus, the absence of fish in small lakes would enable a more diverse community of macroinvertebrates to occur, and this may explain the weak or missing effect of lake size in our study. Also, biodiversity relative to lake size can be expected to be higher in small lakes and ponds where the littoral habitat heterogeneity interfaces with pelagic regions (Wetzel 2001). Shurin (2001) concluded that fish facilitated invasion of more zooplankton species than they excluded, but species richness along a lake size gradient might differ among different taxonomic groups. In a study of 80 Swiss ponds sized between 6 and 94,000 $m^2$ (median area = 1800 $m^2$) Oertli et al. (2002) found that pond size was only important for species richness of odonates and concluded that a set of small-sized ponds may host more species than a single large pond of the same total area. Moreover, from a comparison of river, stream, ditch and pond biodiversity of macrophytes and macroinvertebrates, Williams et al. (2003) concluded that individual ponds varied considerably in biodiversity, but that ponds at the regional level contributed most to biodiversity by supporting more unique species.

In conclusion, lake size makes a difference. Taxon richness clearly changes for some taxonomic groups such as macrophytes and fish, whereas other groups remain unimpacted. In many aspects, however, ponds and lakes are relatively similar. The small Danish lakes and ponds situated in lowland and highly agri-cultivated areas usually exhibit high nutrient concentrations and frequently turbid water. However, in small lakes high nutrient concentrations do not necessarily lead to high phytoplankton biomass as in larger lakes owing to the effects of other controlling factors.

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# Role of sediment and internal loading of phosphorus in shallow lakes

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*Key words:* biomanipulation, iron, recovery, redox, release mechanisms, retention

**Abstract**

The sediment plays an important role in the overall nutrient dynamics of shallow lakes. In lakes where the external loading has been reduced, internal phosphorus loading may prevent improvements in lake water quality. At high internal loading, particularly summer concentrations rise, and phosphorus retention can be negative during most of the summer. Internal P loading originates from a pool accumulated in the sediment at high external loading, and significant amounts of phosphorus in lake sediments may be bound to redox-sensitive iron compounds or fixed in more or less labile organic forms. These forms are potentially mobile and may eventually be released to the lake water. Many factors are involved in the release of phosphorus. Particularly the redox sensitive mobilization from the anoxic zone a few millimetres or centimetres below the sediment surface and microbial processes are considered important, but the phosphorus release mechanisms are to a certain extent lake specific. The importance of internal phosphorus loading is highly influenced by the biological structure in the pelagic, and lakes shifting from a turbid to a clearwater state as a result of, for example, biomanipulation may have improved retention considerably. However, internal loading may increase again if the turbid state returns. The recovery period following a phosphorus loading reduction depends on the loading history and the accumulation of phosphorus in the sediment, but in some lakes a negative phosphorus retention continues for decades. Phosphorus can be released from sediment depths as low as 20 cm. The internal loading can be reduced significantly by various restoration methods, such as removal of phosphorus-rich surface layers or by the addition of iron or alum to increase the sediment's sorption capacity.

## Introduction

Phosphorus availability is regarded as the most important factor for determining the water quality of lakes. Numerous studies have shown that high loading of phosphorus leads to high phytoplankton biomass, turbid water and often undesired biological changes. The latter includes loss of biodiversity, disappearance of submerged macrophytes, fish stock changes, and decreasing top-down control by zooplankton on phytoplankton.

In order to reverse the eutrophication of lakes, much effort has been made to reduce the external loading of phosphorus. Some lakes respond rapidly to such reductions (Sas, 1989), but a delay in lake recovery is often seen (Marsden, 1989; Jeppesen et al., 1991; van der Molen & Boers, 1994), one of the reasons being that phosphorus accumulated in the sediment during the period of high loading needs time to equilibrate with the new loading level. Phosphorus release from the sediment into the lake water may be so intense and persistent that it prevents any improvement of water quality for a considerable period after the loading reduction (Granéli, 1999; Scharf, 1999).

Compared to deep lakes where a redox dependent accumulation of phosphorus occurs in the anoxic hypolimnion during stratification, shallow lakes are usually well mixed and oxidized throughout the water column. Nonetheless, the sediment of shallow lakes has often been demonstrated to release phosphorus to oxic lake water (Lee et al., 1977; Boström et al., 1982; Jensen & Andersen, 1992), suggesting that other factors than redox conditions at the sediment–water interface are involved. The importance of sediment–water interactions in shallow lakes is furthermore enhanced by the high sediment sur-

face:water column ratio, which means that the potential influence on lake water concentrations is stronger than in deeper lakes. The direct contact with the photic zone throughout the year and the regular mixing regime guarantee stable and near optimum conditions for primary production (Nixdorf & Deneke, 1995). Often the phosphorus pool in the sediment is more than 100 times higher than the pool present in the lake water, and lake water concentrations therefore depend highly on the sediment–water interactions.

In this paper we give a short review of the phosphorus retention in lake sediments and the mechanisms and factors suggested to be vital to phosphorus release from lake sediments. Our aim is to show that the sediment of shallow lakes is the domicile of numerous highly dynamic processes that may have very substantial effects on the total phosphorus budget and lake water quality.

## Retention and phosphorus in the sediment

### *Retention of phosphorus*

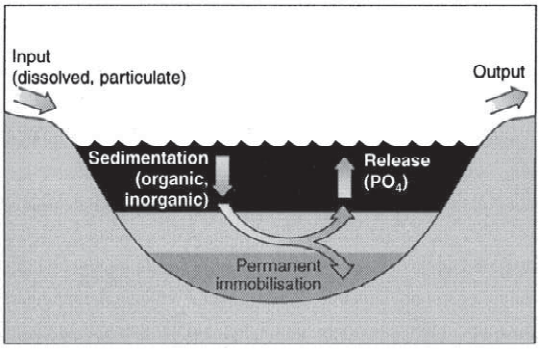
During steady state conditions a certain amount of the phosphorus entering a lake is retained in the sediment (Fig. 1). The retention percentage depends on the hydraulic retention time, as demonstrated through different simple, empirically established models of the Vollenweider type: $P_{lake}=P_{in}/(1+tw^{0.5})$, relating in-lake phosphorus ($P_{lake}$) to inlet concentrations ($P_{in}$) and hydraulic residence time (tw) (Vollenweider, 1976; OECD, 1982). These models cannot, however, adequately describe the transient phase after reduced loading when the system is not in equilibrium and strongly influenced by the internal loading of phosphorus. The net retention of phosphorus is the difference between two processes with large opposite directed flux rates: (i) the downward flux caused mainly by sedimentation of particles continuously entering the lake or produced in the water column (algae, detritus etc.) and (ii) the upwards flux or gross release of phosphorus driven by the decomposition of organic matter and the phosphorus gradients and transport mechanisms established in the sediment. Phosphorus sedimentation from the lake water can be enhanced in productive lakes via co-precipitation with calcium carbonate (House et al., 1986; Driscoll et al., 1993; Golterman, 1995; Hartley et al., 1997).

The importance of internal loading of phosphorus during lake recovery is demonstrated by the finding that phosphorus concentrations often increase during summer in shallow eutrophic lakes (Sas, 1989; Phillips et al., 1994; Welch & Cooke, 1995; Ekholm et al., 1997). In most cases this increase can only be the result of increased sediment loading, implying that summer phosphorus concentrations are largely controlled by internal processes (Jeppesen et al., 1997; Ramm & Scheps, 1997; Kozerski & Kleeberg, 1998). The most pronounced impact is often found in the most eutrophic lakes in which summer concentrations typically exceed winter concentrations by 200–300% from June until October (Søndergaard et al., 1999).

Mass balance calculations have shown that phosphorus retention exhibits a seasonal pattern mimicking the seasonal variation in lake water phosphorus. In a study of 16 Danish lakes, Søndergaard et al. (1999) showed that the retention was positive during winter irrespective of the eutrophication level, while it was negative during part of the summer. Even lakes with a phosphorus concentration below 0.1 mg P $l^{-1}$ had a 2-month negative retention in mid-summer, but the duration of negative retention increased to 5 months in lakes with a mean summer phosphorus concentration above 0.2 mg P $l^{-1}$. In the most eutrophic lakes, a strongly negative retention occurred in May, suggesting that the onset of the increasing biological activity in spring triggered the release of some of the phosphorus retained during winter. In early summer retention was found to be less negative owing to the occurrence of a clearwater phase following late-spring development of a high zooplankton biomass and its grazing on phytoplankton (Sommer et al., 1986; Luecke et al., 1990; Jeppesen et al., 1997; Blindow et al., 2000).

### *Forms of phosphorus in the sediment*

When entering the sediment, phosphorus becomes a part of the numerous chemically and biologically mediated processes and is ultimately either permanently deposited in the sediment or released by various mechanisms and returned in dissolved form to the water column via the interstitial water. It should be emphazised, however, that lake sediments can be very different and highly variable regarding chemical composition. Parameters such as dry weight, organic content, and content of iron, aluminum, manganese, calcium, clay and other elements with the capacity to bind and release phosphorus may all influence sediment–water interactions (Søndergaard et al., 1996).




P-forms in the sediment:

- Dissolved ($PO_4$, organic P)
- Particulate

| | |
|---|---|
| Iron: | Fe (III) hydroxides, Fe (OOH), (ads.) |
| | Strengite, Fe $PO_4$ |
| | Vivianite, $Fe_3$ $(PO_4)_2$ 8 $H_2O$ |
| Alum: | Al $(OH)_3$ (ads.) |
| | Variscite, Al $PO_4$ |
| Calcium: | Hydroxyapatite, $Ca_{10}$ $(PO_4)_6$ $OH_2$ |
| | Monetite, Ca H $PO_4$ |
| | Calcite (ads.) |
| Clay: | (ads.) |
| Organic: | "Labile" |
| | "Refractory" |

*Figure 1.* Schematic presentation of phosphorus pathways when entering a lake and some of the most important phosphorus compounds found in the sediment (from Søndergaard et al., 2001). Adsorbed phosphorus is indicated by ads.

Chemical sequential extractions have been widely used in order to describe the many different forms in which phosphorus occurs in the sediment (Williams et al., 1971; Hieltjes & Lijklema, 1980; Psenner et al., 1988; Golterman & Booman, 1988). The aim is often to give a more precise description of the potentials for phosphorus release from the sediment and to predict its future influence on lake water concentrations (Lijklema, 1993; Seo, 1999). By fractionation phosphorus is characterized as being bound to the variety of inorganic sediment components as described above (Stumm & Leckie, 1971; Boström et al., 1982) or in organic phosphorus compounds. Organically bound phosphorus occurs in more or less labile forms or in a refractory form that is not released during mineralization and which constitutes a fraction permanently buried in the sediment. Fractionation schemes usually yield operationally defined fractions, but it may be debated which type of sediment phosphorus the different fractionations actually measure (Pettersson et al., 1988; Jáugeui & Sánches, 1993). Often, loosely sorbed organic and inorganic fractions as well as the iron-bound and redox-sensitive sorption of phosphorus are considered potentially mobile (Boström et al., 1982; Søndergaard, 1989; Søndergaard et al., 1993; Rydin, 2000). Petticrew et al. (2001) found a close connection between total phosphorus release rates and the iron-bound phosphorus components in the sediment, and during lake recovery, this fraction can constitute a majority of the phosphorus released (Fig. 2). While fractionation schemes may provide relevant information on the overall and long-term conditions for phosphorus sorption expected to prevail in the sediment, it has been difficult so far to establish general relationships between phosphorus forms and the intensity and duration of internal loading. Knowledge coupling the mechanisms behind internal loading with sediment characteristics seems inadequate (Phillips et al., 1994; Welch & Cooke, 1995).

*Importance of biological structure*

The biological structure of a lake can significantly influence its phosphorus concentrations and retention (Beklioglu et al., 1999). For example, clearwater conditions resulting from increased top-down control on phytoplankton often ensure considerably lower in-lake nutrient concentrations (Søndergaard et al., 1990; Benndorf & Miersch, 1991; Nicholls et al., 1996). A positive relationship between clearwater conditions and increased phosphorus retention was documented by the changes recorded in Danish Lake Engelsholm after biomanipulation involving a 66% removal of the fish stock (Søndergaard et al., 2002a). Here, decreased turbidity led to significantly lower phosphorus concentrations and higher phosphorus retention. Furthermore, the period with negative retention during summer was reduced from 6 to 4 months. The annual net retention changed from −2.5 to +3.3 g phosphorus $m^{-2}$ $y^{-1}$. These observations indicate that if a lake returns to a turbid state after being clear for some years, it has the risk of suffering from a high internal loading again. It also suggests that when biomanipulation is considered to improve lake water quality after a loading reduction it might be advantageous to wait for some years until the influence of internal phosphorus loading is reduced.

Several mechanisms are probably involved in the increased phosphorus retention when the clearwater state is achieved. These include the reduced sedimentation of organic matter, which again reduces oxygen

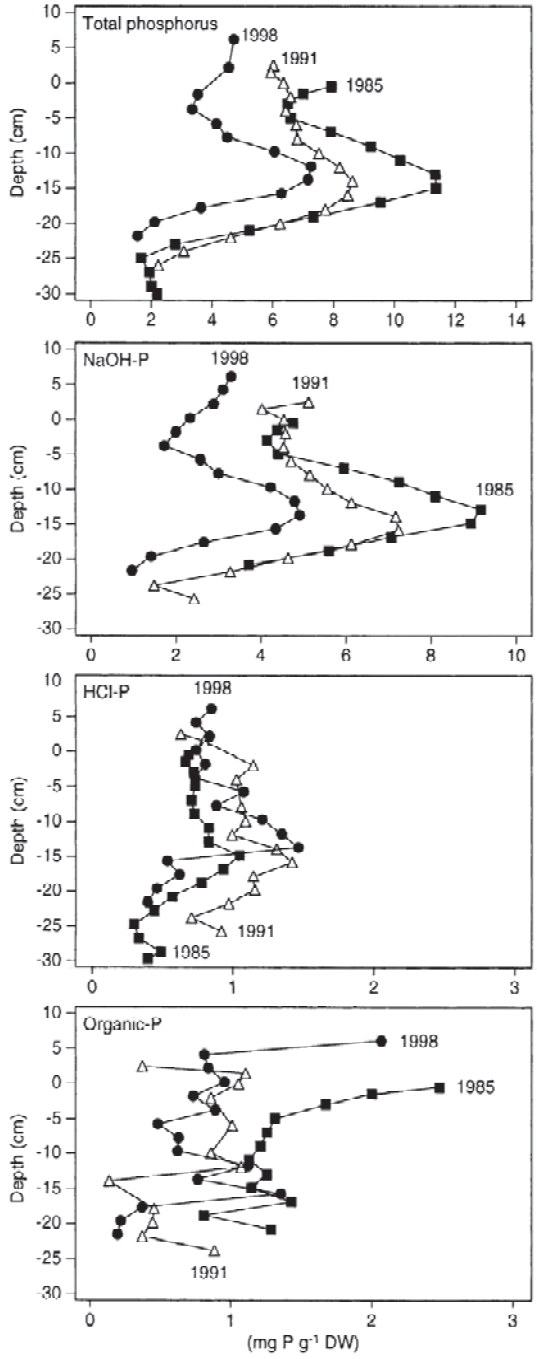



*Figure 2.* Changes in sediment phosphorus profiles of hypertrophic Lake Søbygaard, Denmark, during recovery (measured in 1985, 1991 and 1998). The external phosphorus loading to the lake was reduced by 80–90% in 1982. Fractionation was conducted according to Hieltjes & Lijklema (1980). Organic-P was calculated as Tot-P – ($NH_4Cl$-P+NaOH-P+HCl-P). After 20 years the annual retention of phosphorus is still negative, and the total recovery period is estimated to last more than 30 years after the loading reduction. The upper panel is from Søndergaard et al. (1999) and is reprinted with kind permission from Kluwer Academic Publishers.

consumption and prevents low redox conditions. In addition, the improved light conditions enhance benthic primary production and with it the phosphorus uptake and oxidation of the sediment surface (van Luijn et al., 1995; Woodruff et al., 1999). If submerged macrophytes become abundant they assimilate phosphorus, but may also effect the retention in other ways (see below). Benthivorous fish as for example bream (*Abramis brama*) have a significant impact on the resuspension of sediment (Breukelaar et al., 1994) as well as on the concentration of phosphorus and its release from the sediment (Havens, 1991; Søndergaard et al., 1992). A reduction in their abundance may be a factor of particular importance in biomanipulated lakes formerly dominated by bream or other benthivorous species (Persson et al., 1993; Hansson et al., 1998). Fish-mediated phosphorus release from the sediment is sometimes believed to be stronger and more important for lake water quality than that achieved through reduced planktivory and top-down control on phytoplankton (Havens, 1993; Horppila et al., 1998).

*Duration of internal loading*

The duration and importance of internal loading relate mainly to the flushing rate, loading history and chemical characteristics of the sediment (Marsden, 1989). Some lakes respond rapidly to an external loading reduction by an immediate or only shortly delayed decline in lake concentrations following the changes in loading (Edmonson & Lehman, 1981; Sas, 1989; Welch & Cooke, 1995; Beklioglu et al., 1999). A fast response may be ensured by a high flushing rate provided that the period with high phosphorus loading was relatively short. On the other hand, a long loading history with a high loading rate is reflected in the size of the phosphorus pool accumulated in the sediment, and, if large, a rapid flushing rate may not suffice to ensure a fast return to low concentrations (Jeppesen et al., 1991).

The sediment depth interacting with the lake water is probably lake specific and highly dependent on lake morphology, sediment characteristics and wind exposure. Most often, phosphorus in the upper approximately 10 cm is considered to take part in the whole lake metabolism (Boström et al., 1982), but mobility of phosphorus from depths down to 20–25 cm has been seen (Fig. 2, Søndergaard et al., 1999). The internal phosphorus loading may be very persistent and endure at least 10 years after an external loading re-

duction has been effected (Welch & Cooke, 1999). In some lakes, phosphorus retention can remain negative even for 20 years or more after the nutrient loading reduction (Søndergaard et al., 1999).

**Release mechanisms**

Numerous mechanisms have been proposed to be responsible for the release of phosphorus from lake sediments. In the following we will give a short review of some of the most important bearing in mind, however, that one should be careful when generalising and that the phosphorus release can be governed by very different mechanisms in different lakes.

*Resuspension*

In shallow lakes, wind-induced resuspension is a mechanism that frequently causes increased concentrations of suspended solids in the lake water. Particulate bound forms of phosphorus settling to the bottom may be resuspended several times before permanent sedimentation (Ekholm et al., 1997). In very shallow lakes, resuspension events increase, more or less continuously, the contact between sediment and water (Kristensen et al., 1992; Hamilton & Mitchell, 1997). An example is shown in Figure 3 from a shallow Danish lake in which suspended solids and total phosphorus increased by a factor 5–10 within a few days during two events of increasing wind. In some shallow lakes, year-to-year variation in internal phosphorus loading has been shown to be largely controlled by wind mixing (Jones & Welch, 1990).

Resuspension increases turbidity, but does not necessarily lead to increased release of phosphorus. This is because the overall process depends on the actual equilibrium conditions between sediment and water and on the capability of phytoplankton to take up phosphorus (Søndergaard et al., 1992; Ekholm et al., 1997; Hansen et al., 1997). In the example from Lake Vest Stadil Fjord (Fig. 3), there was no or only a very slight persistent increase in total phosphorus concentrations after the wind events. In other lakes it has been shown that resuspension increases release rates (Fan et al., 2001), or at least during some parts of the season may cause a release, while there may be no effect later in the season due to changed concentrations in the lake water (Søndergaard et al., 1992). From measurements in a shallow Finnish lake, Horppila & Nurminen (2001) concluded that in early summer, the concentration of suspended solids had a highly significant positive effect on soluble reactive phosphorus concentrations in the water, whereas in late summer no effect was found.

*Temperature*

Temperature reflects many of the biologically mediated processes in the lake. The pronounced seasonality in internal loading and retention capacity strongly indicates that the release mechanisms are linked to temperature and biological activity (Jensen & Andersen, 1992; Boers et al., 1998; Søndergaard et al., 1999). These include stimulation of the mineralization of organic matter, the release of inorganic phosphate with increasing temperatures (Boström et al., 1982; Jeppesen et al., 1997; Gomez et al., 1998), and increased sedimentation of organic material related to the seasonal variation in phytoplankton productivity (Ryding, 1981; Istvánovics & Pettersson, 1998).

As organic loading increases during spring and mineralization processes are strengthened, the penetration depth of oxygen and nitrate into the sediment declines (Tessenow, 1972; Jensen & Andersen, 1992). Jensen and Andersen (1992) observed that the temperature effect on phosphorus release was strongest in lakes with a large proportion of iron-bound phosphorus. They also noticed a decrease in the thickness of the oxidised surface layer with increasing temperatures, suggesting a redox-sensitive release. The thickness of the top oxic sediment can thereby influence the concentration of phosphorus in the whole water body (Gonsiorczyk et al., 2001).

*Redox*

Redox conditions in the surface sediment are the classical explanation of sediment water interactions. Einsele (1936) and Mortimer (1941) very early described how the phosphorus release was determined by redox-sensitive iron dynamics. In oxidised conditions, phosphorus is sorbed to iron (III) compounds, while in anoxia iron (III) is reduced to iron (II) and subsequently both iron and sorbed phosphate returned into solution. In shallow lakes the whole water column is usually oxic, which also establishes an oxic surface layer of the sediment with a high capacity to bind phosphorus. In agreement herewith, Penn et al. (2000) suggested that an oxidized microlayer at the sediment–water interface partially inhibits sediment phosphorus release under well-mixed conditions in spring and au-

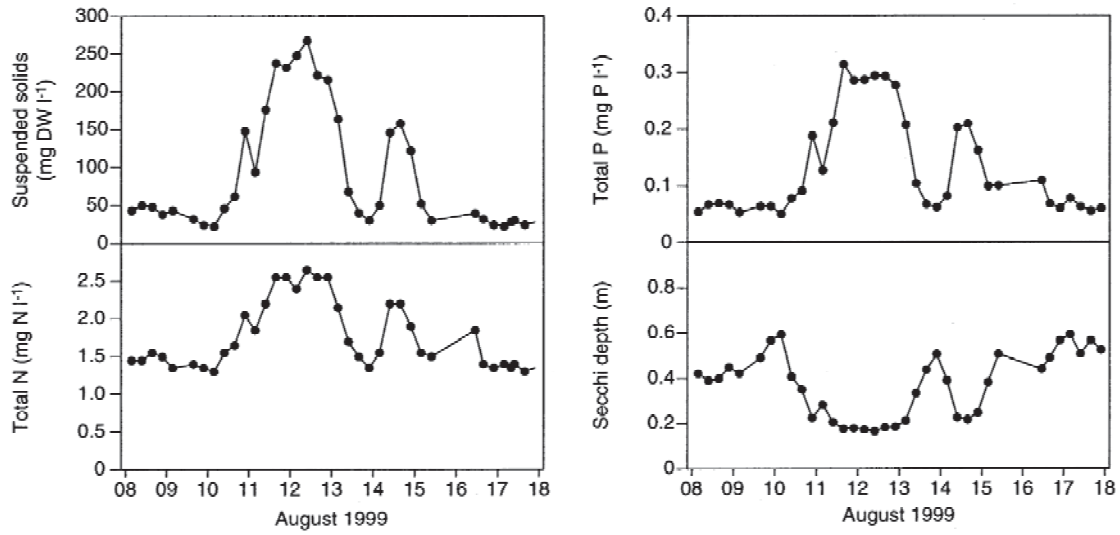



*Figure 3.* Lake water changes in Lake Vest Stadil Fjord, Denmark, during 10 days of varying wind speed (from 0–2 to 5–7 to 2–3 m $s^{-1}$). Lake area is 450 ha and mean depth 0.8 m.

tumn. On the other hand, phosphorus trapped in the oxic microlayer can be freed when the microlayer is chemically reduced at the onset of anoxia. Then, high phosphorus release rates are observed. In this way, the oxidized microlayer may serve to regulate seasonality in rates of sediment phosphorus release, but does not influence long-term sediment–water exchange. If the oxic surface layer becomes saturated with phosphorus, phosphorus transported upwards from deeper sediment layers may simply pass through the oxic layers into the water column.

The presence of nitrate, which normally penetrates deeper into the sediment than oxygen and, like oxygen, has the capability to keep iron in its oxidised form, can also be important for the redox sensitive sorption of phosphorus (McAuliffe et al., 1998; Duras & Hejzlar, 2001). For example Kozerski et al. (1999) found that high summer phosphorus release rates were related to low nitrate input to Lake Müggelsee. In contrast, Jensen et al. (1992) showed that the presence of nitrate during winter and early summer diminished the release rates, whereas nitrate addition in late summer enhanced the phosphorus release in the same lakes, probably by stimulating the mineralization process.

*pH*

pH is particularly important in lake sediments where the capacity to retain phosphorus depends on iron, because the phosphorus binding capacity of the oxygenated sediment layer decreases with increasing pH as hydroxyl ions compete with phosphorus ions (Lijklema, 1976). The impact of pH on release has been illustrated by Koski-Vahala and Hartikainen (2001), who demonstrated that high pH, which is common in eutrophic lakes during summer, may markedly increase the internal phosphorus loading risk when linked with intensive resuspension. In the sediment of eutrophic lakes, photosynthetically elevated pH can establish more phosphorus, which is loosely sorbed to iron, and thus increase release rates (Lijklema, 1976; Søndergaard, 1988; Welch & Cooke, 1995; Istvánovics & Pettersson, 1998).

*Iron:phosphorus ratio*

The combined ferric oxides and hydroxides available in the sediment may bind phosphate very effectively. The involvement of iron in the dynamic equilibrium between the sediment and water has led to the suggestion that an iron dependent threshold exists for the sediment's ability to bind P. Jensen et al. (1992) showed that the retention capacity was high as long as the Fe:P ratio exceeds 15 (by weight), and when above this ratio internal phosphorus loading may be prevented by keeping the surface sediment oxidised. Caraco et al. (1993) suggested that the Fe:P ratio should exceed 10 if it was to regulate phosphorus release. The presence of a threshold is supported by the strong positive relationship between the concentra-

tions of phosphorus and iron in the surface sediment of shallow lakes (Søndergaard et al., 2001). In hardwater lakes, iron may be less important for the phosphorus release compared to the solubilisation of apatite due to decreased pH during mineralization (Golterman, 2001). Yet, it has been suggested that even in calcareous systems, iron and aluminium, when present in high concentrations, are involved in regulating the phosphorus cycling (Olila & Reddy, 1997).

*Chemical diffusion and bioturbation*

The interstitial water of the sediment, which normally contains less than 1% of the sediment's total phosphorus pool, is important for the phosphorus transport between sediment and water as interstitial phosphate constitutes the direct link to the water phase above and the solid–liquid phase boundary between water and sediment (Boström et al., 1982; Löfgren & Ryding, 1985). An upward transport of phosphorus is created via a diffusion-mediated concentration gradient, normally appearing just below the sediment surface.

Bioturbation from benthic invertebrates or through gas bubbles produced in deeper sediment layers during the microbial decomposition of organic matter may significantly enhance the process (Ohle, 1958, 1978; Fukuhara & Sakamoto, 1987). There is some evidence that bioturbation from benthic chironomids can enhance phosphorus release rates, particularly in sediments low in total iron (Phillips et al., 1994). Benthic invertebrates can also inhibit phosphorus release by supplying oxic water into the sediment and increasing the oxidised surface layer of the sediment (Boström et al., 1982). Similarly, low phosphorus flux rates can be recorded despite a steep interstitial water gradient, provided that the top centimetres of the sediment either have a high phosphorus sorption capacity (Moore et al., 1998), or its chemical processes are controlled by a photosynthetically active biofilm (Woodruff et al., 1999).

*Mineralization and microbial processes*

In shallow and eutrophic lakes, the sediment continuously receives high amounts of freshly produced organic material that is not decomposed before reaching the sediment. Thus, sediment bacteria may have a significant role in the uptake, storage and release of phosphorus (Pettersson, 1998). High organic input creates the potential for a high mineralization rate, provided that the supply of oxiders such as oxygen or nitrate is sufficient. Subsequently, the typical sediment profile will have oxygen penetrating a few millimetres into the sediment, followed by nitrate which can be found several centimetres into the sediment depending on the decomposition rate and the nitrate input.

If nitrate concentrations are low, but sulphate levels and the supply of biodegradable organic matter high, desulphurication and sulphur cycling may become important parts of the sediment processes (Holmer & Storkholm, 2001). Hydrogen sulphide formed from sulphate reduction induces the formation of iron sulphide and decreases the potential of phosphorus sorption and thereby the potential phosphorus release from the sediment (Ripl, 1986; Phillips et al., 1994; Kleeberg & Schubert, 2000; Perkins & Underwood, 2001). The internal, dynamic P-release from lake sediments may thereby be determined by the ligand exchange of phosphate against sulphide with iron. During winter, low sedimentation rates and sufficient supply of oxygen or nitrate to the sediments establish a high redox potential, maintaining the sedimentary iron in its oxidized form.

*Submerged macrophytes*

In shallow lakes, submerged macrophytes have the potential of being very abundant with a high plant-filled volume. Macrophytes may, however, influence the phosphorus cycle both negatively and positively. Decreased release is seen when oxygen released from the roots increases the redox-sensitive phosphorus sorption to iron-compounds (Andersen & Olsen, 1994; Christensen et al., 1997), and when high abundance of macrophytes diminishes the resuspension rate and reduces the phosphorus release from the sediment (Granéli & Solander, 1988; Van den Berg et al., 1997). Increased phosphorus release may be recorded in dense macrophyte beds and beneath macrophyte canopies due to low oxygen concentrations (Frodge et al., 1991; Stephen et al., 1997), or due to increased pH (James et al., 1996). From experiments and measurements in the Broads in U.K., Stephen et al. (1997) concluded that if rooted macrophytes have a significant effect on phosphorus release they increase it. Barko & James (1997) have given a comprehensive review of the effects of submerged aquatic macrophytes on nutrient dynamics, sedimentation and resuspension.

## Concluding remarks

Because of their strong impact on lake water concentrations, it is clear that knowledge of sediment–water interactions and the processes behind retention and release of phosphorus is fundamental for understanding the function of shallow lakes. Many different mechanisms may be involved in the sediment release, but two types are often of particular importance: (i) redox-dependent release of phosphorus bound to iron and (ii) microbial processes. The redox-dependent release mechanism is also relevant in well-mixed eutrophic lakes with an organic-rich sediment. Here, the oxic surface layer is often too thin to prevent a release from deeper parts of the sediment. Microbial processes being fuelled by degradable matter accumulated in the sediment or sediment settling from the lake water, together with the supply of oxidizers (oxygen, nitrate and sulphate), are important for the cycling of phosphorus.

Presently, we do not have sufficient knowledge to develop general models for the release mechanisms of shallow lakes, descriptions that could be used as a predictive tool in lake management following a nutrient loading reduction. More clear relationships between easily measurable sediment characteristics and net release rates of phosphorus have to be established. It should also be noted that phosphorus release mechanisms to some extent are lake specific: resuspension being important particularly in very shallow and wind-exposed lakes, redox-sensitive release in iron-rich systems, etc.

In order to combat internal phosphorus loading and accelerate lake recovery after decreased external loading, numerous lake restoration techniques have been developed and tested (Dunst et al., 1974; Born, 1979; Cooke et al., 1993; Phillips et al., 1999; Welch & Cooke, 1999; Søndergaard et al., 2000; Perkins & Underwood, 2001). They comprise both physical measures, such as sediment dredging by which nutrient rich sediment is removed, as well as chemical methods. The chemical methods aim to influence the redox-dependent phosphorus fixation by either improving the sorption capacity of the elements already present in the lake/sediment or by adding new sorption capacity, as for example iron, alum or calcium (Søndergaard et al., 2002b). For all types of restoration measures, an important prerequisite for obtaining success and long-term effects is elimination of the underlying reasons for the impoverished water quality, i.e. a sufficient reduction of the external phosphorus loading (Benndorf, 1990; Jeppesen et al., 1990; Hansson et al., 1998; Søndergaard et al., 2000).

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# Seasonal dynamics in the concentrations and retention of phosphorus in shallow Danish lakes after reduced loading

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*To investigate how phosphorus concentrations and retention were affected during recovery after reduced external phosphorus loading, phosphorus concentrations and loading were seasonally monitored for 11 years in 15 shallow eutrophic Danish lakes. Hydraulic loading, inlets, and lake water were sampled monthly during the winter (October–April) and bimonthly during the summer (May–September) to estimate monthly mass balances. Total phosphorus concentrations in the lakes were two to three times higher during summer than during winter; internal loading seems to be of great significance. The retention of phosphorus was negative from April until October, the most negative values generally occurring in May and July (–30% of external loading). In June, retention was less negative (–12%), probably influenced by clearwater phases which often appear in early summer. During the 11 years, the period with negative phosphorus retention decreased from five to four months and, as a median, the maximum phosphorus concentration declined from ca. 0.4 to 0.2 mg phosphorus $l^{-1}$. In Lake Engelsholm, which was biomanipulated by a 66% reduction of the zooplanktivorous fish stock (mainly bream, Abramis brama), marked changes were observed in biological structure. Also affected were phosphorus concentrations and retention. Simultaneously, with an increase in Secchi depth from 0.7 to 2.0 m, summer concentrations of total phosphorus decreased to less than 50% of the pre-biomanipulation level. The period exhibiting negative phosphorus retention declined from six months before the intervention to four months afterwards. It is concluded that the recovery period after reduced phosphorus loading may last long in shallow eutrophic lakes, but that the recovery is significantly influenced by biological structure. During recovery, both the duration and net phosphorus release rates from the sediment declined progressively.*

*Keywords:* recovery, mass balance, internal loading, retention capacity, biomanipulation

## Introduction

Chemical resistance in lakes being in the recovery phase after a reduction in the external loading of nutrients is a well known phenomenon in both shallow and deep lakes, particularly in the case of reduced phosphorus inputs (Marsden 1989; Jeppesen et al., 1991; Güde et al., 1998). Initially, in-lake P concentrations do not decline as markedly as does the external loading, and improvements in lake water quality may thus be delayed or insufficient to meet established water quality criteria (Graneli, 1999; Scharf, 1999). Long-term monitoring after nutrient loading reductions has revealed that the transient phase leading up to equilibrium with the lower external loading may last for a long interval; that is, for decades rather



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than years (Welch and Cooke, 1995; Seo, 1999; Søndergaard et al., 1999). The cause of the observed resistance is the release of P from a pool accumulated in the upper sediment during the period with high external loading (Ryding, 1981). Therefore, less P input → increased release from sediment to the water → reduced P retention → continued high in-lake P and poor lake water quality. Quantification of the processes behind the equilibration and recovery period is generally lacking, which makes lake management more complicated.

Basically, the recovery period and the importance of internal loading for overall lake concentrations are considered to depend on the flushing rate, loading history, and chemical characteristics of the sediment (Marsden, 1989). If the period with high P loading was relatively short, a high flushing rate may ensure a rapid return to low in-lake concentrations after the external loading reduction. Conversely, a long loading history is reflected in the amount of P accumulated in the sediment and a rapid flushing rate may not be sufficient to ensure a return to low concentrations (Jeppesen et al., 1991). Generally, lakes with a long history of eutrophication are considered to respond slowly to a loading reduction (Marsden, 1989). Characteristics of the sediment and its sorption capacity include a strong coupling of P to oxidized iron, implying that the highest P concentrations in lake sediment occur when both the loadings of P and Fe were high (Søndergaard et al., 1996). Furthermore, numerous potential release mechanisms, both chemical and biological, may strongly influence the pattern and the duration of the release (Boström et al., 1982; Welch and Cooke, 1995; Boers et al., 1998). Apart from direct sediment-water interactions, changes in lake biological structure also influence both nitrogen and P concentrations. Clearwater conditions resulting from either increased top-down control on phytoplankton or increased abundance of submerged macrophytes are good examples of how lower in-lake nutrient concentrations may be established (Søndergaard et al., 1990; Nicholls et al., 1996; Jeppesen et al., 1998a).

Most attempts to describe the concentrations and retention of nutrients in lakes in the recovery phase after a loading reduction have been based on yearly calculations (Vollenweider, 1976; Sas, 1989; Seip et al., 1990; Rossi and Premazzi, 1991, Nürnberg, 1998), whereas seasonal aspects are not well described. Elevated summer concentrations of P in shallow lakes, which often reach a level more than twice as high as during winter (Welch and Cooke, 1995; Petterson, 1998; Søndergaard et al., 1999), show, however, that the internal cycling of P significantly affects seasonal concentrations (Ramm and Scheps, 1997; Graneli, 1999; Bennion and Smith, 2000). In addition, the growing season appears to be the most sensitive part of the year to elucidate the impact on biological structure and overall water quality of a lake, such as the quantitative effect of nutrients on phytoplankton (Anneville and Pelletier, 2000).

This article describes the seasonal changes recorded in the retention and concentrations of P in 15 Danish lakes in which detailed mass balance studies have been conducted for 11 years, and in which total phosphorus (TP) concentrations have been reduced significantly. Changes in retention and concentrations following a shift between the clearwater and the turbid state are described for Lake Engelsholm, which was biomanipulated by removal of zooplanktivorous fish. The aims of the study were to elucidate how the recovery phase is expressed seasonally in the concentration and retention of P in shallow lakes following an external loading reduction, and to describe how P retention may be affected by a major change in the biological structure established through biomanipulation measures.

## Methods and study area

The 15 lakes presented in this study are mainly eutrophic with high P concentrations and low Secchi depth (Table 1). All the lakes are shallow and the water columns are mixed and oxic throughout the summer. Monthly mean temperature ranges from 1 to 3° C during winter and from 16 to 18° C during summer. Most of the lakes are small; only two are larger than 100 ha. They are all included in a nation-wide monitoring programme including 37 lakes which are believed to represent all Danish lake types (Kronvang et al., 1993). In one lake (Lake Engelsholm), marked changes occurred in biological structure. Lake Engelsholm was biomanipulated from April 1992 to September 1994 by removing cyprinid fish, mainly bream (*Abramis brama*), which constituted 85% of the total catch and roach (*Rutilus rutilus*). In total, 19.2 t, or 438 kg wet weight $ha^{-1}$, were removed, corresponding to 66% of the total fish stock estimated for 1990 (County of Vejle, 1996; Jeppesen et al., 1998a). In this study, pre-conditions refer to the years 1989–1993 and post-conditions to 1994–1999. External TP loading remained unchanged during the investigation period (Table 1).

In all lakes, sampling of the lake water was conducted monthly during the winter (October–April)

**Table 1.** Characteristic (mean values and range in parentheses) of the 14 lakes and of Lake Engelsholm before and after biomanipulation.

| | | Lake Engelsholm | |
|---|---|---|---|
| | 14 lakes | Before | After |
| Area, ha | 388 (15–4200) | 44 | 44 |
| Mean depth, m | 3.1 (1.0–9.9) | 2.6 | 2.6 |
| Hydraulic loading, $10^6$ m$^3$ y$^{-1}$ | 27 (0.4–200) | 1.7 | 2.1 |
| Hydraulic retention time, years | 0.6 (0.03–1.7) | 0.23 (0.22–0.24) | 0.22 (0.18–0.30) |
| P-loading, g P m$^{-2}$ y$^{-1}$ | 4 (0.2–40) | 0.70 (0.62–0.84) | 0.81 (0.47–1.05) |
| Mean summer TP, mg P l$^{-1}$ | 0.21 (0.03–0.70) | 0.16 (0.12–0.20) | 0.07 (0.05–0.08) |
| Mean summer Secchi depth, m | 1.1 (0.4–2.5) | 0.70 (0.53–0.86) | 1.99 (1.57–2.49) |
| Mean summer chlorophyll, mg l$^{-1}$ | 110 (10–300) | 97 (69–123) | 38 (19–50) |

and bimonthly during the summer (May–September), yielding a total of 19 samplings per year from 1989 to 1999, except for one of the lakes which was not sampled in 1998 and 1999. The main inlet of each lake was sampled 18–26 times annually, depending on seasonal variations in discharge, while minor inlets were sampled less frequently, depending on their relative contribution to total loading. Outlets were sampled with the same frequency as that of lake water. Total P was analyzed as molybdate reactive phosphorus following persulphate digestion according to Koroleff (1970).

Total discharge of tributaries and outflow ($Q_m$) was measured monthly with an OTT-propeller (Windolf et al., 1996). Water level (H) in the inlet and outlet streams was automatically and continuously recorded and daily discharge was calculated by use of the relationship obtained for H and $Q_m$. Monthly water balances were calculated for each lake using the following equation:

$$Q_{inm} + Q_{inu} + Prec = Vol_{dif} + Q_{outm} + Q_{outu} + Evap,$$

where $Q_{inm}$ and $Q_{outm}$ are the total discharge measured in inlets and outlets, respectively. Evap and Prec are total monthly evaporation and precipitation obtained from meteorological stations in the vicinity of the lakes. $Vol_{dif}$ is the monthly change in lake volume and $Q_{inu}$ and $Q_{outu}$ are the unmeasured input from the catchment without stream inlets to the lake and the output (seepage) from the lake, respectively. The net value of $Q_{inu}$ and $Q_{outu}$ was determined monthly by adjusting the mass balance, if $Vol_{dif} + Q_{outm} + Evap > Q_{inm} + Prec$, then $Q_{inu}$ equals the difference; otherwise $Q_{outu}$ is equal to the difference. Total P transport was estimated for each inlet and outlet as the product of the daily water discharge and phosphorus concentration obtained by linear interpolation. The TP value of the unmeasured discharges to and from the lake ($Q_{inu}$, $Q_{outu}$) were assumed to equal the Q-weighted concentrations in the measured inlets and outlets. Atmospheric deposition on the lake surface was estimated using an average rate of 0.2 kg P ha$^{-1}$ year$^{-1}$.

## Results

Mean summer concentrations of TP have declined significantly ($p < 0.1$) during the 11-year study period from 1989 to 1999. Seasonal TP varied considerably (Figure 1), increasing in spring and early summer and peaking in August when mean TP for the 14 lakes reached 260% of mean winter concentrations. The seasonal P retention varied correspondingly (Figure 2), being positive during the winter (October to March) and negative from April to September. The mean retention in June was relatively high (–12%) compared to the other summer months.

During 1989 to 1999, the mean loading of P to the 14 lakes declined considerably (Figure 3). The highest loading reduction was recorded from 1989 to 1991, although it varied considerably from year to year depending on hydraulic loading. Thus, the lowest loading was recorded in years with low hydraulic loading, as for example the two dry years 1996 and 1997. In the same period, annual P retention changed from positive values in 1989–1991 to nearly zero or slightly negative levels in 1992 to 1996, and slightly positive values from 1997 to 1999.

The seasonal retention pattern differed correspondingly (Figure 4). In the first period with relatively high but decreasing external loading (1989–1991), mean P retention was positive throughout the year except for July, August, and September. In the period with relatively low loading and low retention (1992–1996), retention was negative for five months (May to

September), with maximum negative retention occurring in July and August. In the last period (1997–1999), when annual retention was more effective, it was still negative from May to September, but the values were lower than previously.

During the study period, maximum TP declined (Figure 5). The most dramatic changes in median values were observed from 1989 to 1994. A gradual decline in 90%- and 75%-fractiles was recorded for the whole period, except for the two dry years 1994 and 1995. From the early to the late 1990s, the median number of months with negative P retention was reduced from five to four, and the number of months with a P retention below –1 mg P $m^{-2}$ $d^{-1}$ decreased from 4.5 to 3 months. In 1989, when external TP loading was highest, the number of months with negative P retention or < –1 mg P $m^{-2}$ $d^{-1}$ was about three.

In Lake Engelsholm, marked changes in biological structure ,were recorded after reductions in the stock of cyprinid fish and a significant increase in number and size of *Daphnia* and a marked reduction in chlorophyll *a* were recorded (Jeppesen et al., 1998a). Secchi depth increased from a summer level of 0.5–0.6 m to 1.2–2.0 m, while chlorophyll *a* decreased from 90 to120 to 35 to 70 µg $l^{-1}$ (Figure 6). The changes were recorded throughout the season, including winter when Secchi depth increased from less than 1.5 m to more than 3 m and chlorophyll *a* decreased from about 50 µg $l^{-1}$ to about 10 µg $l^{-1}$. Mean summer TP decreased from 0.15 to 0.07 mg P $l^{-1}$ (Table 1, Figure 6), whereas winter TP was unaffected. Both the duration and levels of seasonal TP retention changed (Figure 7). Prior to biomanipulation, P retention was negative for six months (April to September), but after biomanipulation the period with negative retention declined to four months (May to August). Minimum mean retention, –6 mg P $m^{-2}$ $d^{-1}$ (July), was observed before biomanipulation; it was only –2 mg P $m^{-2}$ $d^{-1}$ (June and August) after the intervention. Mean TP retention in the winter was unchanged and varied, except for a high retention in October before the manipulation, between 1 and 3 mg P $m^{-2}$ $d^{-1}$ both before and after the manipulation.

## Discussion

Chemical resistance following nutrient loading reduction has been observed in many lakes (Marsden,

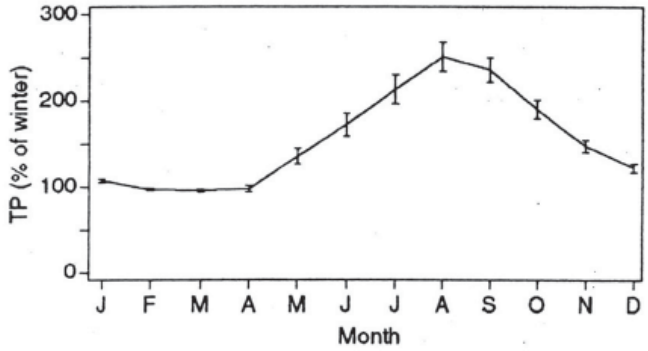



**Figure 1.** Mean (±SE) monthly lake water concentrations of TP relative to winter concentrations (1 January–31 March) in the 14 lakes during 1989–1999.

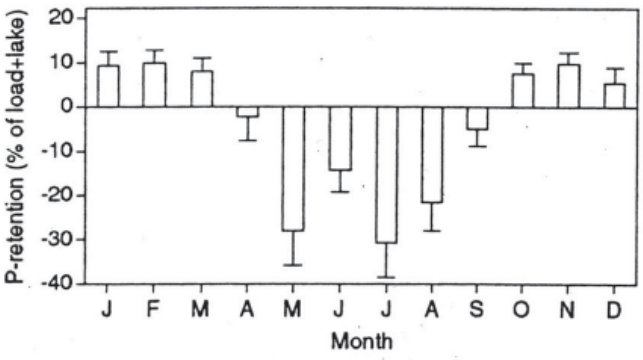



**Figure 2.** Seasonal retention (in % of inlet + pool present in the lake water) of TP in the 14 lakes. Mean values (± SE) are shown.

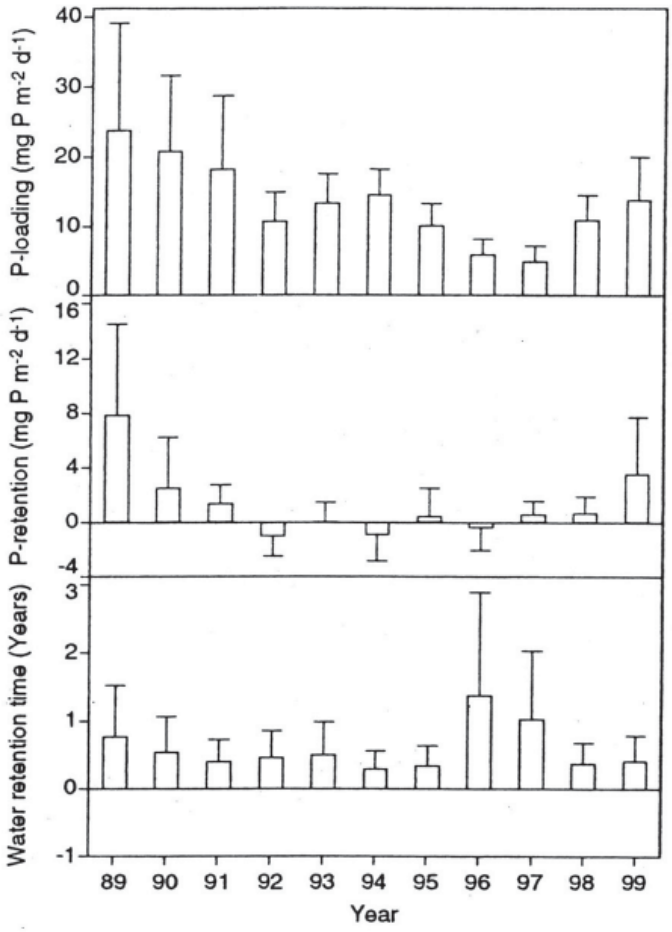



**Figure 3.** Yearly (± SE) loading and retention of P and mean hydraulic retention (± SE) from 1989 to 1999 in the 14 lakes.

1989; Jeppesen et al., 1991; Søndergaard et al., 1999), but a rapid decline in in-lake P concentrations following reduced external loading has also been recorded (Edmondson and Lehman, 1981; Sas, 1989; Beklioglu et al., 1999). Carpenter et al. (1999) divided eutrophic lakes into three groups according to their response to reduced P loading: (1) reversible lakes responding immediately and proportionally to the loading reduction, (2) hysteretic lakes responding only after a period with comprehensive loading reduction, or (3) irreversible lakes where reduction of loading does not suffice to ensure recovery. Seen in this perspective, a general feature of these lake studies is a prolonged recovery period; however, the tendency to diminished TP concentrations and increased retention suggests that most lakes belong to the hysteretic type. Nevertheless, most of the lakes are still far from being in equilibrium sensu Vollenweider (Søndergaard et al., 1999), and when the lakes will reach a new equilibrium remains an open question.

External loading to the investigated lakes generally decreased from 1989 to 1991, but no exact data prior to this period are available. Several case studies suggest that a gradual decrease in the loading of lakes by means of sewage treatment and by improving the retention capacity of catchment areas has occurred. Chemical measurements show that mean TP in Danish streams from 1972 to 1992 declined from 0.4 to 0.2 mg P $l^{-1}$ (Jeppesen et al., 1999). Thus, a gradual reduction in the external loading of the investigated lakes has most likely occurred 10 to 20 years before the initiation of the present investigation (Søndergaard et al., 1999), and the loading decreases observed in the early 1990s may thus be regarded as belonging to the latter part of a period when major reductions in external loading took place. This implies that the total duration of the recovery period from the time when the external loading reduction was initiated until equilibrium was reached in some instances exceeds 20 years.

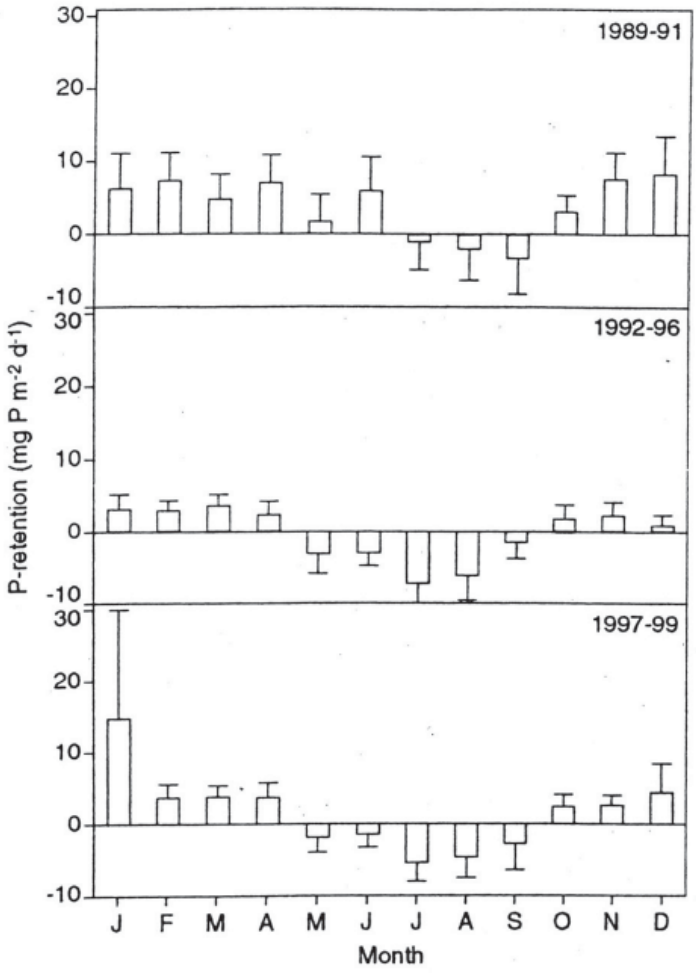



**Figure 4.** Seasonal changes in the mean retention of TP in the 14 lakes. Upper: 1989–1991, middle: 1992–1996; lower: 1997–1999.

The considerable seasonal variations in phosphorus concentrations reflect a significant influence by internal lake processes and cannot be attributed only to lower inlet concentrations (Søndergaard et al., 1999). A similar strong seasonality and variable retention capacity have been observed in many other lakes (Ryding, 1981; Van der Molen, 1991) and appear to be a general phenomenon in shallow eutrophic lakes. Changes in the seasonal capacity of lakes to retain P depend on several factors, but the simultaneity with temperature suggests that the main factors involved are significantly coupled with biologic processes, as also proposed earlier (Jensen and Andersen, 1992; Boers et al., 1998; Søndergaard et al., 1999). These may include changes in pH and sedimentation rates related to the seasonal variation in phytoplankton productivity (Ryding, 1981; Søndergaard, 1989; Istvanovics and Petterson, 1998) and to reduced capacity of the uppermost sediment to retain P as organic loading and mineralization increase, and the penetration depth into the sediment of oxygen and nitrate declines.

The seasonal duration of internal loading declined during recovery. Although not directly comparable, this is in accordance with the recovery of deep Lake Constance following a reduction in nutrient loading in the 1970s, where Güde et al. (1998) reported that the period of P deficiency in the upper euphotic zone was extended from two to three months to six to seven months in the late re-oligotrophication period. In the first phase of re-oligotrophication, summer concentrations of particulate P decreased only in late summer (August–September), possibly indicating reduced input from hypolimnetic waters during destratifica-

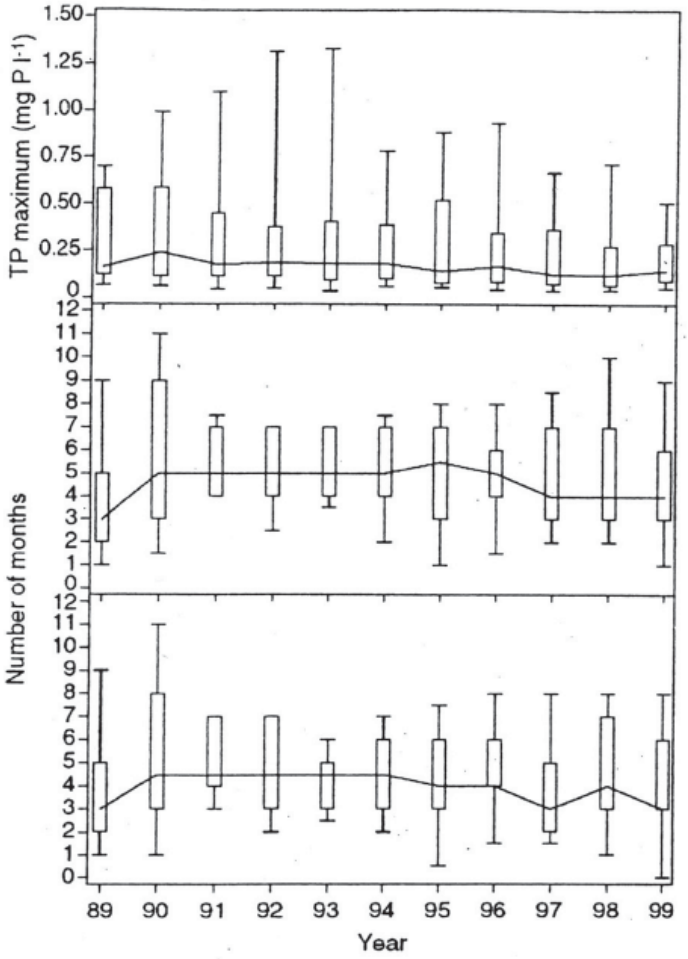



**Figure 5.** Trends in annual median of maximum TP concentrations in the 14 lakes during 1989–1999 (upper), number of months with negative TP retention (middle) and numbers of months with TP retention <–1 mg P $m^{-2}$ $d^{-1}$ (lower). Mean, quartiles, and 10 and 90 percentiles are also shown.

tion. In shallow Lake Müggelsee, Köhler et al. (2000) recorded a decline in TP only in the spring and not in the summer following reduced external loading. The latter recovery pattern may be parallel to the difference in seasonal retention appearing in lakes with different TP levels. Søndergaard et al. (1999) showed that in lakes with TP <0.1 mg P $l^{-1}$ retention was negative for only two months (July and August), whereas in lakes with TP >0.2 mg P $l^{-1}$ retention was negative for five months (April to August), the retention capacity of less eutrophic lakes thus being reduced, especially in spring and early summer. If this can be regarded as a general pattern of recovery, it appears that the first signs of improved water quality in shallow lakes should be expected during the early phase of the growing season.

Summer net internal loading declined during the recovery process and maximum TP decreased. Similarly, Ryding (1981) recorded reduced TP concentrations in Swedish lakes after an external loading reduction. The gradual decline primarily in the median, 75%- and 90%-fractiles suggests that the most significant response occurs in the most eutrophic lakes. The tendency to a slight decrease in negative retention in June has previously been interpreted as a consequence of the clearwater phase appearing in early summer in many lakes (Søndergaard et al., 1999). At this time, a late spring peak in zooplankton biomass often develops before the young-of-year zooplanktivorous fish start foraging in the pelagic (Luecke et al., 1990; Jeppesen et al., 1997; Blindow et al., 2000).

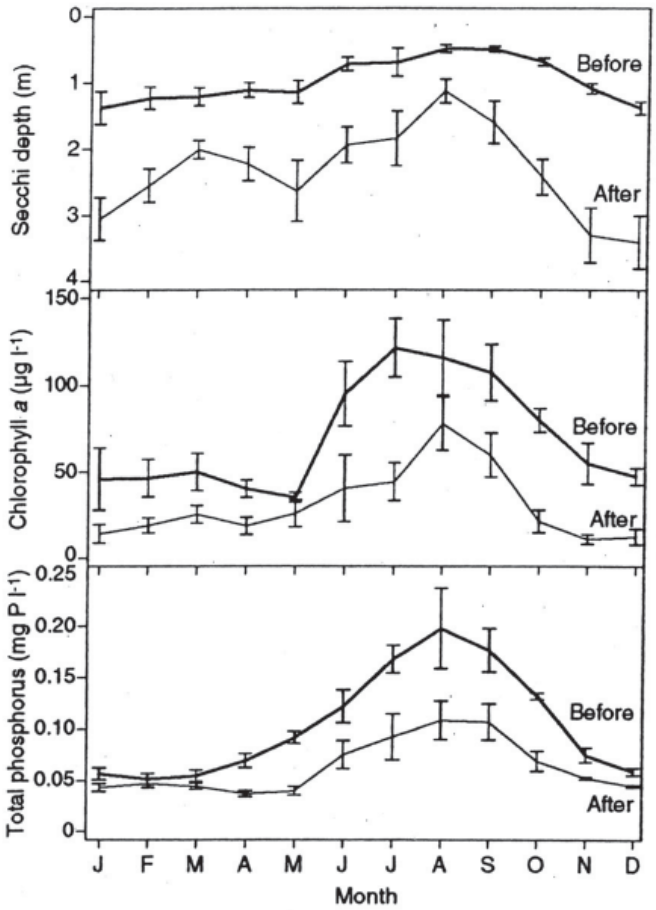



**Figure 6.** Mean Secchi depth, chlorophyll *a* and TP (± SE) in Lake Engelsholm before (1989–1993) and after (1994–1999) biomanipulation by cyprinid reduction (from Søndergaard et al., 2001).

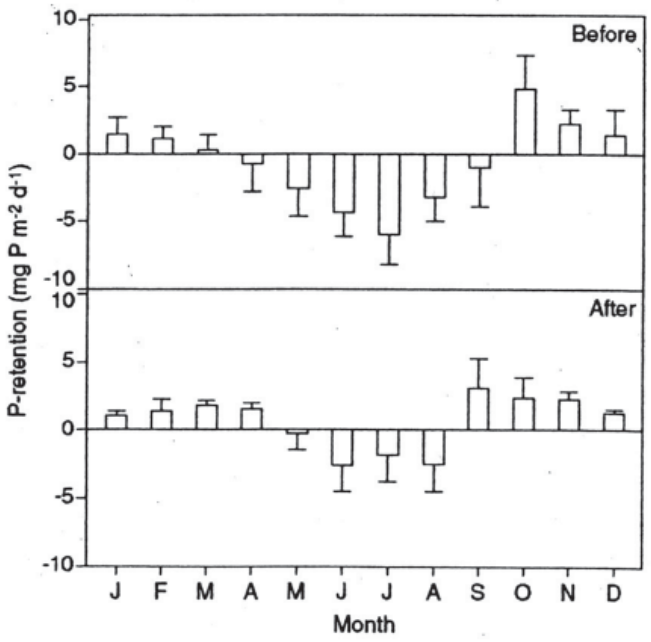



**Figure 7.** Mean (± SE) monthly retention of TP in Lake Engelsholm before (1989–1993) and after (1994–1999) biomanipulation by cyprinid reduction (from Søndergaard et al., 2001).

The presence of a positive relationship between clearwater conditions and increased P retention is supported by the observations from Lake Engelsholm. Here, dramatic changes were recorded in zooplankton and fish following the reduced abundance of zooplanktivorous fish (Jeppesen et al., 1998a), as well as on P and phytoplankton (Figures 6 and 7). The considerable changes in nutrient concentrations and retention recorded after the biomanipulation coincided with a shift to clear water and demonstrate the strong coupling between biological structure and nutrient cycling. Submerged macrophytes, which should be able to increase in abundance as transparency improves, did not appear in significant quantities after the biomanipulation perhaps due to the impact from herbivorous waterfowl (T. Lauridsen et al., Department of Lake and Estuarine Ecology, NERI, pers. communication). This indicates that as long as the water remains clear the presence of macrophytes is not a prerequisite for obtaining increased P retention. The mechanisms behind increased P retention when shallow lakes shift from the turbid to the clearwater state remain to be elucidated as we have no information on how sediment structure was changed. However, several mechanisms which increase the sediment sorption capacity may be involved, including reduced sedimentation of organic matter, enhanced benthic primary production taking up P and oxidizing the sediment surface (Van Luijn et al., 1995), and increased sediment oxidation due to increased abundance of benthic invertebrates. If submerged macrophytes, which have a positive influence on lake water clarity when present in adequate abundance (Moss, 1990; Jeppesen et al., 1998b), appear this can also effect the retention of P (Søndergaard and Moss, 1998). Although the most pronounced changes in Secchi depth and chlorophyll *a* occurred in winter, neither TP concentrations nor retention were affected then, indicating that winter TP concentrations are less influenced by internal processes. The reduction in the abundance of especially bream, which through its feeding behaviour may have a significant impact on the resuspension of sediment (Breukelaar et al., 1994), as well as on TP concentrations and the release of P from the sediment (Søndergaard et al., 1992), may be a factor of particular importance in biomanipulated lakes formerly dominated by bream (Hansson et al., 1998).

The example from Lake Engelsholm demonstrates that increased retention capacity of P both prolongs the period of positive P retention in the sediment and also diminishes the summer net release rates. This observation is in agreement with earlier findings recording both considerably lower TP and TN concentrations after a change in biological structure (Meijer et al., 1994; Hansson et al., 1998; Jeppesen et al., 1998a). The importance of the biological structure for determining the extent of chemical resistance has also been shown by Beklioglu et al. (1999), who recorded a rapid decline in P concentrations and high retention in the clearwater Little Mere, U.K. after an effluent diversion. Even though some general trends and patterns emerge from our study of the response of Danish lakes to reduced P concentrations, it is obvious that considerable variability between lakes may occur and remains to be explained. Also, lakes should be regarded individually as they may exhibit highly different response patterns and mechanisms (Ryding, 1981).

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# 10 • Chemical treatment of water and sediments with special reference to lakes

MARTIN SØNDERGAARD, KLAUS-DIETER WOLTER AND WILHELM RIPL

## INTRODUCTION

The impact of human activities on the aquatic environment has increased during the past century. Chemical pollutants have increased in rivers, lakes and coastal areas due to rising population densities, farming and industrialisation. The effects have included acid rain and acidification of surface water over large areas where catchment soils as well as bedrock are poor in limestone, and increased deposition of heavy metals and other chemicals causing contamination and bioaccumulation of toxic products. A marked increase in the use of pesticides and the enhanced production of organic substances used in various industries have led to increased pollution by a wide variety of organic micropollutants (Kristensen & Hansen, 1994).

Measures to combat industrial sources of pollutants have been implemented at least in some parts of the world, although improvements in many areas are still needed, just as the environmental impact of many organic micropollutants remains to be elucidated (Kristensen & Hansen, 1994). However, the influence from nutrient-rich wastewater from cities or aquaculture and the use and leaching of fertilisers in agriculture still constitute significant problems that often overshadow other environmental problems. Apart from more local industrial influences, increased nutrient loading, resulting in eutrophication and a loss of the natural functionality of many ecosystems, is considered to be the most important and widespread environmental problem of lentic and coastal waters. In lakes, one of the most important factors is the increased availability of nutrients, especially phosphorus which – via its limiting effect on the growth of phytoplankton and thus indirectly on the community of higher organisms – has had a very important influence on lake metabolism and lake water quality (Ohle, 1953; Thomas, 1969; Jeppesen *et al.*, 1999). For decades, a reduction of external nutrient loading, especially phosphorus, has therefore been of paramount importance in lake management for counteracting undesired eutrophication effects and for improving water quality.

Multiple measures have been employed in the catchments to diminish nutrient loading. The first step has often been to establish sewage works in communities above a certain size to remove organic pollution and to avoid low oxygen concentrations in rivers. Biological sewage treatment has, however, only minor effects on nutrient loading and sewage works have often been expanded to incorporate varying degrees of phosphorus stripping and nitrogen removal. The effort to reduce phosphorus loading has often been supplemented with increased use of phosphate-free detergents. More recently, measures have also been taken to reduce nutrient loading from arable soils such as increased storage capacity of animal manure on farms, establishment of uncultivated buffer strips along streams and rivers, maintenance of green cover of fields in winter and retirement of agricultural land (Jeppesen *et al.*, 1999). Finally, improved nutrient removal and retention may also be achieved through new constructed wetlands, remeandering of channelised streams and biomanipulation of lakes.

Besides reducing the external nutrient and organic loading, restoration of inland freshwaters has mainly focused on lakes where chemical restoration techniques have been widely used. Rivers and streams are open systems and have a greater potential for natural recovery as they are flushed and noxious substances more easily diluted. In rivers, the general trend is thus simply to stop the input of

materials and not to establish internal control by chemical means.

The reason why lakes have long been subjects of in-lake restoration measures is that they often respond slowly to a reduction in external phosphorus loading, and lake water phosphorus concentrations are not reduced to the same extent as external loading (Marsden, 1989; Sas 1989; Søndergaard *et al.*, 1999). Correspondingly, the desired improvement in water quality fails to occur even if phosphorus loading has been reduced to a level where improvements were expected. The reason for this resilience is internal loading of phosphorus from the sediment where phosphorus was accumulated when external loading was high. For a period following an external load reduction, part of this phosphorus pool is released concurrently with the establishment of a new dynamic equilibrium between the sediment and water phase. This internal phosphorus loading may be of great significance in both shallow, unstratified lakes where nutrients are added to the photic zone all summer (Jeppesen *et al.*, 1991; Phillips *et al.*, 1994; Søndergaard *et al.*, 1999), and in deep lakes where phosphorus is accumulated in the bottom layer of water during summer stratification. The duration and intensity of the internal phosphorus release after an external loading reduction depend on the degradability of sedimentary organic matter as an energetic basis for micro-organisms and on the size of the releasable sediment phosphorus pool. The release rates depend on the intensity of the microbial metabolism and transport mechanisms within the sediment. Internal phosphorus release is recorded especially in eutrophic lakes, but in summer even lakes with relatively low nutrient concentrations may experience a short-term net internal loading (Fig. 10.1). In shallow eutrophic lakes, summer phosphorus concentrations may rise to values more than twice as high as the concentrations derived from external loading (Jeppesen *et al.*, 1997; Søndergaard *et al.*, 1999). The release may be very persistent and endure for many years, with a conservative estimate of at least ten years after an external load reduction (Welch & Cooke, 1999). In some lakes, phosphorus retention has, in fact, remained negative for more than 15 years after the nutrient loading reduction (Søndergaard *et al.*, 1999).

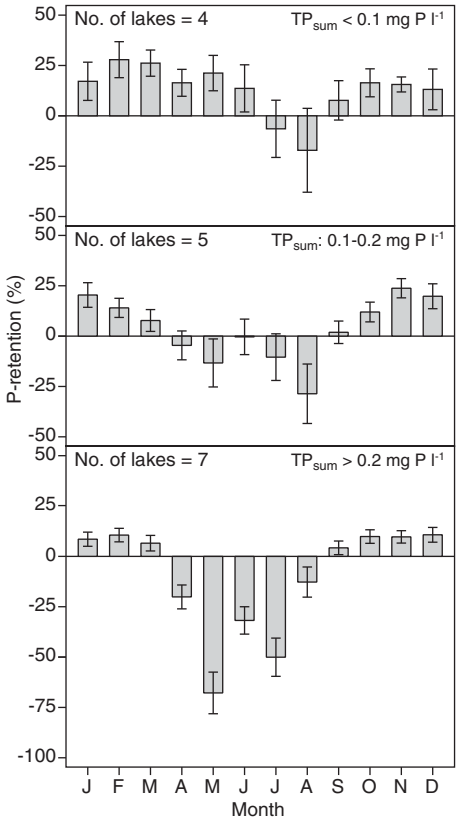



Fig. 10.1. The seasonal retention of phosphorus at different nutrient concentrations (mean summer concentration of total phosphorus, $TP_{sum}$) based on mass balance calculations for eight years (20 annual inlet–outlet samplings) in 16 Danish lakes. Phosphorus retention is given as percentage of external loading. From Søndergaard *et al.* (1999).

In an attempt to reduce internal phosphorus loading and accelerate lake recovery after a decrease in external loading, numerous experiments and lake restoration projects have been undertaken using various methods aimed at decreasing the sediment phosphorus release. Many methods have been chemical and have focused on reducing the effects of eutrophication by influencing phosphorus availability. The methods for chemical restoration of lakes have been applied to both stratified and shallow lakes with the objectives of influencing bioactivity and redox-dependent phosphorus fixation.

This chapter describes the background and the chemical and biological relations in lakes affecting the design and evaluation of the different chemical restoration tools. We focus mainly on the many chemical processes in which phosphorus may be part, and on the mechanisms controlling the exchange between the sediment and water phase. The last part of the chapter briefly describes the application of different types of chemical restoration tools and the underlying hydrological, chemical and biological principles and techniques, and possible problems that may arise.

## INTERACTIONS BETWEEN SEDIMENT AND WATER IN LAKES AND THEIR IMPLICATIONS FOR LAKE RESTORATION

### Stratification and oxygen supply

Due to the temperature-dependent water density (maximum at 4 °C), most lakes deeper than 5–10 metres exhibit thermal stratification during part of the season. However, the depth required for stratification to occur varies considerably, depending on the surface area and surrounding topography. In temperate regions where most restoration projects have been implemented, dimictic lakes are the most common lake type. These lakes circulate freely twice a year: in spring when surface layer temperatures increase above 4 °C, and in autumn when surface stratum temperatures decrease and approach 4 °C. In summer, stratification divides dimictic lakes into three zones: a warm and less dense upper stratum (epilimnion), usually being more or less completely mixed, a cold and dense bottom stratum with more quiescent water (hypolimnion), and a transient zone with a steep temperature gradient (metalimnion), separating and minimising the exchange of nutrients and other substances between the epilimnion and hypolimnion.

Because of the summer temperature stratification in deep lakes, the chemical environment, including redox conditions, undergoes a significantly different development from that of shallow lakes. In shallow and completely mixed lakes, water tends to be saturated with oxygen except immediately above the sediment surface all summer. In calm periods, interim stratification may occur, this being, however, quickly eliminated as soon as the wind rises or thermal homogeneity is achieved at night.

In the deeper and more permanently summer-stratified lakes, oxygen concentrations in the hypolimnion decrease following the onset of thermal stratification in early summer, with stratification minimising the input of oxygenated water from the epilimnion. The rate of hypolimnetic oxygen depletion depends on the volume of hypolimnion, the water movement across the metalimnion, and on sediment oxygen consumption. In eutrophic lakes, where high primary production and sedimentation normally create an organically rich sediment with a proportionately high oxygen consumption, hypolimnion oxygen concentrations will be depleted sooner than in more nutrient-poor lakes with less production as well as sedimentation of organic matter. The thickness of the oxygen-depleted bottom layer increases concurrently with the consumption of oxygen in the hypolimnion in nutrient-rich lakes, which, as a consequence, become more and more undersaturated. When stratification sets in, oxygen is first depleted in the layers nearer the bottom sediments. Over the course of the summer the oxygen-depleted bottom layer includes more of the hypolimnion.

Following oxygen depletion, the hypolimnion concentrations of nitrate and sulphate typically decrease as they are alternative electron acceptors to oxygen. In contrast, the concentrations of phosphate, ammonium, iron, alkalinity and pH increase due to the metabolic processes in the sediment. In nutrient-poor lakes, sediment oxygen consumption may be so insignificant that hypolimnion oxygen concentrations may be more or less saturated all summer. In extremely nutrient-poor lakes, summer oxygen concentrations may even increase towards the bottom (orthograde oxygen profile) since oxygen is dissolved in higher concentrations at low temperatures during the spring overturn. Minimum oxygen concentrations are expected in the metalimnion in such lakes.

The different availability and consumption of oxygen implies that the concentration of various substances at the sediment/water interface differs widely between both shallow and deep as well as nutrient-rich and nutrient-poor lakes. In shallow, well-oxidized lakes, the gradients are established close to the

sediment/water interface where a diffusive boundary layer of variable thickness, depending on mixing conditions, is established. In shallow nutrient-poor lakes, oxygen may penetrate the sediment by as much as a few centimetres, while penetration is limited to a few millimetres in nutrient-rich lakes, where the abundance of biodegradable organic matter is higher. Correspondingly, the penetration depth of other terminal electron acceptors into the sediment (nitrate and sulphate), depleted during the course of decomposition, is considerably less in nutrient-rich lakes. Different penetration depths, expressed by the classical redox stratification of various electron acceptors (Boström *et al.*, 1982), divide the sediment into an upper oxidised layer of variable depth (depending on eutrophication level) and a lower, reduced sediment layer. In eutrophic stratified lakes, steep concentration gradients of the numerous organic and inorganic compounds, involved in oxidation–reduction reactions, are found in the water phase established close to, or below, the thermocline where marked decreases in oxygen concentrations occur.

Many of the chemical and biochemical processes in water and sediment are redox processes, i.e. involve electron transfer. The extent of oxidative or reduced conditions in a solution can be described by the redox potential (Eh), the capability to oxidise or reduce the surrounding environment. The redox potential decreases 58 mV for each pH unit increase and strongly depends on oxygen concentrations. The theoretical redox potential in water saturated with oxygen (pH = 7 and T = 25 °C) is 800 mV, and well-oxidised water will normally have a potential ranging between 400 and 600 mV. Not until oxygen concentrations become very low ($<0.1$ mg $l^{-1}$) will the redox potential decrease to *c.* 200 mV. This is the level (200–300 mV) where iron is reduced from $Fe^{3+}$ to $Fe^{2+}$, which is of crucial importance for the sorption of phosphorus. Oxidised iron oxides and hydroxides such as $Fe(OH)_3$ have a high affinity to adsorb phosphorus. In contrast, reduced iron is mostly incapable of adsorbing phosphorus. In the complete absence of oxygen, even lower redox potentials are established, and may reach −100 to −200 mV in the sediment. The presence of nitrate is, however, able to maintain the redox potential at a relatively high level, thus preventing the reduction of $Fe^{3+}$.

## The sediment and phosphorus fixation

The sediment is to a large extent the terminal site for the particulate substances added to or produced in the water column and which are not mineralised during sedimentation before reaching the sediment surface. Therefore, sediments usually act as nutrient sinks for autochthonous and allochthonous particulate material. In nutrient-rich and productive lakes this net accumulation adds several millimetres or more to the sediment annually. Some lakes and reservoirs also receive significant input from river inlets rich in suspended solids. The phosphorus reaching the sediment is mostly in the particulate form inorganically bound to active surfaces of iron, aluminium or calcium, or as organic debris.

The fixation of phosphorus in the sediment varies depending on four processes: (1) transport of soluble phosphate between solid components; (2) adsorption–desorption mechanisms; (3) chemosorption; and (4) biological assimilation (Jacobsen, 1978). Chemosorption normally signifies chemical fixation of soluble compounds subsequently unaffected by changes in solute concentrations. Adsorption, on the other hand, is a physical fixation of soluble compounds on surfaces in constant equilibrium with solute concentrations. Both adsorption and chemosorption of phosphate by sediments involve numerous compounds, the most important being iron, calcium, aluminium, manganese, clay and organic matter. Apart from concentrations, adsorption and chemosorption processes are often dependent on both pH and redox potentials, which themselves are consequences of bacterial metabolism.

After a long period of high nutrient loading, a lake's processes of production and respiration will be out of equilibrium and the P/R quotient will stay well below 1, causing accumulation of nutrients and biodegradable organic matter in the sediment. Because of the high affinity of iron to bind phosphorus, total phosphorus concentrations in the surface sediment not only depend on the external phosphorus loading, but also on the concentrations of iron (Søndergaard *et al.*, 1996). With increased loading, the sediment undergoes significant concurrent changes in structure and function. With the onset of anoxic conditions and increased activity of anaerobic bacteria at the sediment surface, the sediments become anaerobic and

sapropelic. Poisonous hydrogen sulphide may also be liberated with the consequent destruction of higher fauna.

To define sediment characteristics, sequential extractions with various chemical compounds have been developed for fractionation and description of the sediment phosphorus pool (Williams *et al.*, 1971; Hieltjes & Lijklema, 1980; Psenner *et al.*, 1988). The sediment phosphorus pool is often divided into inorganic and organic fractions, with the latter comprising a loosely sorbed and easily releasable form and a more tightly fixed, refractory form. The inorganic fraction is often subdivided into a loosely sorbed fraction and fractions bound to iron, aluminium and calcium.

From a management perspective, the sediment's pool of releasable phosphorus determines the magnitude and duration of internal phosphorus loading that continues following a reduction in nutrient loading. Often, both the loosely sorbed organic and inorganic fractions, as well as the iron-bound and redox sensitive sorption of phosphorus, are considered potentially mobile and releasable. However, as yet, it has not been possible to establish any simple and reliable relationships between the different sediment phosphorus fractions and the ultimate pool of releasable phosphorus. Although such knowledge may provide information on the overall and long-term conditions expected to prevail concerning the sorption of phosphorus in the sediments, such information on static phosphorus binding gives only limited insight into actual changes in phosphorus forms released under dynamic conditions. Another problem associated with the use of static parameters is determining the sediment depths from which phosphorus release may be expected. Traditionally, phosphorus in the upper 10 cm is considered potentially mobile. However, some studies indicate that phosphorus may be transported upwards from depths up to 20–25 cm (Søndergaard *et al.*, 1993, 1999).

## Phosphorus release from sediments

Although the interstitial water normally contains $<1\%$ of the sediment's total phosphorus pool (Boström *et al.*, 1982), this pool, nevertheless, has a significant bearing on the phosphorus transport between sediment and water. This is because the interstitial water's phosphate content constitutes the direct link between the particulate phosphorus pool and the water phase above. The transport of phosphorus between the sediment and water phase results from a diffusion-mediated concentration gradient, normally appearing just below the sediment surface. Bioturbation from benthic invertebrates or through gas bubbles produced in deeper sediment layers during the microbial decomposition of organic matter may, however, significantly enhance the upward transport of phosphorus. Ohle (1958, 1978) reported that released methane gas is a significant transport process for further mixing of excessive phosphate concentrations from the interstitial into the overlying water.

Biodegradability of an organic substrate is necessary for bacterial degradation and for the release of phosphorus. At low sedimentation rates in oligotrophic lakes, organic matter is so resistant that further decomposition of the settled material does not occur. In contrast, organic matter settling at high rates in the upper sediment layers in eutrophic lakes is usually abundant and easily degraded. Bacterial metabolism in these lakes is therefore only limited by the delivery of the electron acceptors, oxygen, nitrate and sulphate, in order to oxidise organic matter. High rates of phosphorus redissolution are dominated by the process-conditioned modifications of the redox potential and pH, caused by this increased metabolism of the micro-organisms.

A number of factors influence the exchange of phosphorus between water and sediments, including redox conditions, pH, iron:phosphorus ratio and resuspension (Boström *et al.*, 1982; Søndergaard, 1988; Jensen *et al.*, 1992; Søndergaard *et al.*, 1992). The solid/liquid phase boundary between water and sediment, or between sediment particles and interstitial water, as well as the different possibilities of transport of matter across these boundary layers, are of crucial importance for the understanding of the dynamic release of phosphorus from sediments. On one hand, the actively metabolising bacteria in the interstitium cannot be supplied with the necessary electron acceptors (oxygen, nitrate and sulphate) by molecular diffusion from the supernatant water alone (Duursma, 1967). On the other hand, the inhibitory metabolic final products (e.g. hydrogen sulphide) cannot be removed by diffusion alone. Thus, individual velocity gradients between water and the solid phase are potential controls for

processes and biota. Beyond the scale of diffusion at the boundary water/sediment layer, water flow, microturbulences, eddies, waves, a slowly circulating hypolimnion and seiches influence the processes.

Phosphorus retention and release depend on temperature. During the cold season, with low sedimentation rates and sufficient supply of oxygen or nitrate to the sediments, high redox potentials develop, maintaining the sedimentary iron in its oxidised form. The combined ferric oxides and hydroxides available in the sediment may effectively bind phosphate. Oxidised iron and/or molecular sulphur from $H_2S$ oxidation usually colour the surface sediments (approx. 1–10 cm) light brown to orange-yellow (Gorham, 1958). Even the reduced iron sulphide can be oxidised again to Fe(III) with its high phosphate-binding capacity, via nitrate. Thus, even eutrophic lakes, suffering from a significant net annual internal phosphorus loading, are capable of retaining phosphorus during the winter season (Fig. 10.1).

At higher temperatures during spring, and high supply of biodegradable organic matter, oxygen and nitrate, which are transported into the sediment from the water above, are consumed in the highest millimetres to centimetres of the sediments as well as immediately above the sediment surface. Thus, at slow water flow, oxygen consumption and mineralisation of organic matter are the result of coupling between nitrification and denitrification (Ripl & Lindmark, 1978). If nitrate is consumed at sufficiently high sulphate concentrations and with a sufficient supply of biodegradable organic matter, sulphate reduction becomes the dominant sediment process. This phase can be determined from a definite decrease in sulphate concentrations in interstitial water and from steep sulphate gradients into the sediment. Hydrogen sulphide formed from sulphate reduction causes the reduction of Fe(III) and the formation of iron sulphide (FeS) by the following formula:

$$2\ FeO(OH) + 3\ H_2S \rightarrow 2\ FeS + S + 4\ H_2O$$

In iron-rich sediments, the colour changes from brown (oxidised status) to black (reduced status). In contrast to Fe(III), the reduced Fe(II) cannot efficiently fix phosphate. Therefore, the process leads to redissolution of phosphate into the interstitial water and, finally, into the water.

From the stoichiometry of the reactions it follows that sulphate reduction usually takes place in the sediments only up to a molar sulphur: iron ratio of about 1.5. Hydrogen sulphide, formed after reaching this ratio, can no longer be detoxified, implying that even reduced sulphur products are restricted by negative feedback due to $H_2S$ as the final product. Thus, in the sediments of the Schlei estuary (Northern Germany), sulphate concentrations in the interstitial water increased after exhaustion of the dissolved Fe(II). This was a visible indication of inhibited sulfate reduction activity. Not until the exhaustion of the sediment iron buffer was almost complete, did a release of iron-bound phosphorus occur. This, in turn, led to an increase in the phosphorus concentration in the interstitial water (Ripl, 1986*a, b*).

In some shallow highly eutrophic lakes, the phosphorus release may be so high that the release over a few weeks amounts to the entire phosphorus content of the upper 1–2 mm of sediment. In hypertrophic Lake Søbygaard, several periods of low phytoplankton biomass and low net sedimentation rates resulted in a net sediment release of phosphorus as high as 100–200 mg P $m^{-2}$ $day^{-1}$ (Søndergaard *et al.*, 1990). In agreement with these processes, a negative correlation was found between sediment phosphorus (as % of the acid-soluble fraction) and the molar sulphur: iron ratio in the sediments of Lake Tegel, Berlin (W. Ripl *et al.*, unpubl. data) (Fig. 10.2). If the organic matter is still further degradable, methane fermentation may occur below the sulphate reduction zone.

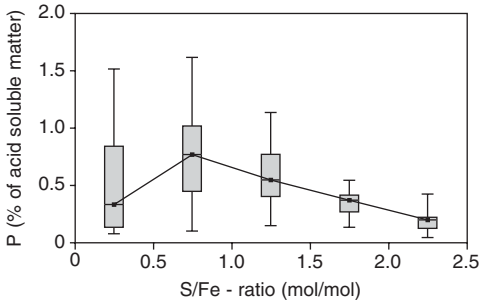



Fig. 10.2. Distribution of phosphorus as percentage of acid-soluble matter depending on the molar sulphur: iron ratio in sediments (0–25 cm) of Lake Tegel, Berlin, 1985–9. 5%, 25%, 50% (line with squares), 75% and 95% percentiles for each sulphur: iron ratio classes. Total number $n = 508$.

Consequently, the depletion of oxygen and the subsequent decrease in the redox potential may not be the principal reason (*sensu* Mortimer 1941, 1942) behind the phosphorus release from highly enriched sediments. Instead, the formation of hydrogen sulphide and its reaction with iron (Hasler & Einsele, 1948) after complete consumption of the nitrate appears to be a significant factor. The internal dynamic phosphorus release from lake sediments can, in most cases, be understood as a ligand exchange of phosphate versus sulphide with iron. Additionally, these processes are accelerated mainly by sedimentation of fresh organic matter after, or during, a planktonic algal bloom. In a $H_2S$-free anoxic environment, phosphate would be dissolved simultaneously with iron, but immediately reprecipitate as iron phosphate in an oxic environment.

## CHEMICAL METHODS FOR LAKE RESTORATION

As pointed out earlier, a sufficient reduction of the external phosphorus loading is of paramount importance for achieving a high water quality in lakes, and every lake restoration project should start by examining and, if necessary, controlling the external loading of nutrients. In some instances, reduction of external loading may suffice to achieve a satisfactory water quality, while in other cases the internal loading is so high that in-lake measures are required. The possibilities of establishing permanent effect via in-lake restoration techniques are, however, poor if the external nutrient has not been brought to a level so low that equilibrium phosphorus concentrations can ensure the desired lake quality.

Either diversion, eliminating totally the anthropogenic source, or advanced wastewater treatment normally removing about 90% or more of the phosphorus content, have been most frequently employed to limit external nutrient loading. In principle, sewage plant removal of phosphorus is based on the same chemical principles as those used in in-lake restoration techniques (see below), i.e. phosphorus is precipitated from the wastewater solution by addition of metal salts, where the metal phosphate is insoluble and subsequently removable. Most often used are salts of alum, iron or calcium (Table 10.1). In some instances also treatment of river inflows with iron salts, e.g. $FeCl_3$, is used to precipitate phosphorus.

As for in-lake measures, there are two kinds of fundamental chemical restoration techniques to counteract eutrophication and these will be described below: (1) improvement of the phosphorus sorption of the substances already present in the lake, or (2) supply of new chemical sorption capacity to the lake (Table 10.1). Phosphorus control by improving existing sorption potentials is usually obtained using oxygen or, less often, nitrate to improve the redox-dependent phosphorus sorption. Phosphorus control by increasing the sorption capacity is usually obtained using alum or iron, or, more rarely, calcium. Although the different techniques are described in isolation below, the maximum effect may be achieved by a combination of techniques.

Besides counteracting eutrophication, chemical techniques may be used to restore soft water lakes from the effects of acid rain. In some countries acidification is the most serious environmental problem (Sandøy & Romundstad, 1995). In such cases, calcium is used to increase pH and restore a suitable environment for several organisms.

### Hypolimnetic oxygenation

#### Aim and chemical background

Hypolimnetic oxygenation or aeration normally aims to increase the oxygen concentration and input of oxygen to the hypolimnion, in order to increase the sorption capacity of phosphorus through increased sorption to oxidised iron components in iron-rich lakes (McQueen *et al.*, 1986; Cooke *et al.*, 1993). Increased availability of oxygen may also decrease the gas formation in the sediment and dim- inish the resuspension of sediment particles and phosphorus release resulting from gas ebullition (Matinvesi, 1996). Long-term oxygenation may decrease the organic content, total nitrogen and the biological oxygen demand in the uppermost sediment (Matinvesi, 1996), which, provided that the external phosphorus loading has been sufficiently reduced, may produce a more permanent improvement of lake water quality.

Table 10.1. *Chemical measures used to restore lakes*

| Chemical compound used | Aim | Techniques |
|---|---|---|
| **Catchment measures** | | |
| Iron, alum, calcium | To precipitate and remove phosphorus with iron or alum hydroxides and calcium | Addition of iron ($FeCl_3$ and $FeSO_4$), alum ($Al_2(SO_4)_3$) or calcium ($Ca(OH)_2$) in wastewater treatment |
| Iron | To reduce loading by precipitation of phosphorus with iron hydroxides | Addition of iron ($FeCl_3$) in river inflows |
| **In-lake measures** | | |
| Oxygen | To improve redox potential and sorption of phosphorus on iron; to enhance the distribution of fish and invertebrates | Injection of pure oxygen or atmospheric air into the hypolimnion or use of full-lift aerators |
| Nitrate | To oxidise organic matter and to improve redox potential and sorption of phosphorus on iron; to decrease hypolimnetic oxygen deficit | Injection of nitrate into the sediment or the hypolimnion |
| Iron | To increase phosphorus sorption capacity | Dosing of iron to water or sediment |
| Alum | To increase phosphorus sorption capacity in aluminium compounds | Dosing of alum to water |
| Calcium | Increased sorption of phosphorus in calcium–phosphorus compounds | Dosing of calcium to water |

Oxygenation may also be used to improve and expand living conditions for cold-water fish (Prepas *et al.*, 1997) and other fauna in the hypolimnion (Dinsmore & Prepas, 1997*a, b;* Field & Prepas, 1997). For example, in the oxygen-treated basins of Amisk Lake, Alberta, cisco (*Coregonus artedii*) were able to feed throughout the water column, whereas in untreated basins, hypoxia restricted cisco to epilimnetic and metalimnetic waters (Aku & Tonn, 1997). Hypolimnetic aerators have also been installed in water supply reservoirs to improve raw water quality by lowering the concentrations of iron, manganese and sulphide (Cooke *et al.*, 1993; Burris & Little, 1998). Hypolimnetic aeration may also be combined with iron addition to facilitate phosphate precipitation and retention in the sediment (McQueen *et al.*, 1986; Jaeger, 1994).

Hypolimnetic oxygenation should be distinguished from complete mixing techniques that do not retain thermal stratification. Among the advantages of avoiding destratification are prevention of the transport of nutrient-rich bottom water to the epilimnion and maintenance of suitable habitats for cold-water fish and other species adapted to cold water. Artificial circulation or destratification (Cooke *et al.*, 1993; Simmons, 1998), hypolimnetic withdrawal (Nurnberg, 1987; Livingstone & Schanz, 1994) and other techniques manipulating the physical environment are not addressed in this chapter.

### Techniques

Over the years, numerous different restoration methods and aerator designs have been implemented to increase hypolimnetic oxygen concentrations (Cooke *et al.*, 1993). Often either pure oxygen or atmospheric air is pumped into deep parts of the lake where it is dissolved into the hypolimnion via diffusers (Fig. 10.3). Oxygen can be supplied from a storage tank at the shore containing liquid oxygen (Prepas *et al.*, 1997; Gächter & Wehrli, 1998). Other

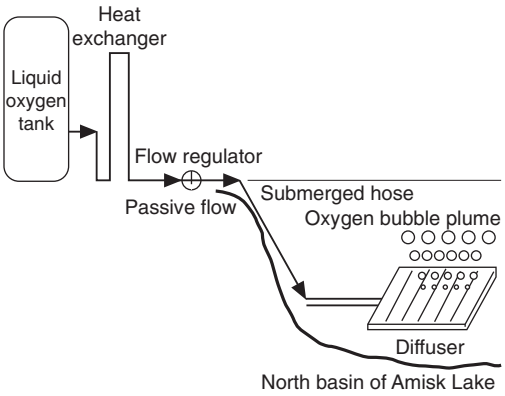



Fig. 10.3. Schematic illustration of the oxygenation system used in Amisk Lake, Alberta. From Prepas *et al.* (1997).

techniques include deep-water aeration where oxygen-depleted bottom water is brought to the surface via a full-lift aerator where it is aerated and returned to the bottom (Ashley *et al.*, 1987; Jaeger, 1994; Burris & Little, 1998).

Hypolimnetic oxygenation during summer stratification can be supplemented with winter aeration, thus ensuring complete mixing of the water column. Pressurised air is released in deep parts of the lake to enhance vertical mixing and dissolved oxygen concentrations and to prolong the oxic period following stratification (Gächter & Wehrli, 1998). However, disturbance of winter stratification during very cold periods may decrease deep-water temperatures, which may cause more extensive freezing of the lake.

### Possible problems in connection with treatment and their effects

If the installation configuration is not designed properly, hypolimnectic aeration may lead to destriction of the thermocline and destratification of the lake, particularly in weakly stratified lakes (Lindenschmidt, 1999). Aeration also often leads to increased hypolimnetic temperatures even when destratification does not occur, but this is usually restricted to a few degrees (Prepas *et al.*, 1997).

Compared to untreated lakes, the hypolimnetic oxygen demand often increases during oxygenation. Several factors may be involved, including enhanced oxygen consumption due to induced circulation currents above the sediment, diminished thickness of the diffusive sublayer adjacent to the sediment and fewer transport-limited processes (Sweerts *et al.*, 1989; Moore *et al.*, 1996; Nakamura & Inoue, 1996). This implies that more oxygen may be needed than that previously calculated using the oxygen demand at stagnant conditions as a basis.

An anoxic hypolimnion and high phosphorus release rates may not be 'cause–effect' related, but two parallel symptoms of common cause are: (1) excessive organic matter sedimentation exhausting dissolved oxygen and (2) high sedimentation rates of phosphorus exceeding the phosphorus retention capacity of the anoxic sediment (Gächter & Wehrli, 1998). Even if oxygenation affects the transitory binding of phosphorus, it is questionable whether hypolimnetic oxygenation causes the permanent burial of phosphorus and, in turn, produces permanent effects on the trophic state of a lake. This may be important if external loading is not reduced before, or simultaneously with, the oxygenation (Gächter, 1987; Gächter & Wehrli, 1998).

## Nitrate treatment

### Aim and chemical background

Nitrate treatment of anaerobic sapropelic sediments aims to reduce the high potential reactivity of sediments by the oxidation of biodegradable organic matter. *In situ* oxidation of sedimentary biodegradable organic matter has its highest potential to control phosphorus in iron-rich systems, enabling a fixation of phosphorus with iron, and an increase of the iron buffer in the sediment (Ripl, 1976, 1978).

Following oxygen consumption, denitrification denotes the first step in the bacterially mediated oxidation of organic matter. Denitrification proceeds during consumption of nitrate, organic matter being oxidised with nitrate to carbon dioxide and water. Although dissimilatory reduction of nitrate to ammonia in a reducing environment has been suggested (Priscu & Downes, 1987; Søndergaard *et al.*, 2000), nitrogen is usually thought to be released as molecular, gaseous nitrogen:

$$5\ CH_2O + 4\ NO_3^- + 4\ H^+ \rightarrow 2\ N_2 + 5\ CO_2 + 7\ H_2O$$

For a non-equilibrium system, a definite redox potential cannot be specified, but the redox potential

of denitrification is always in the positive range where iron is oxidised to the $Fe^{3+}$, which binds phosphate very effectively. Trivalent iron binds phosphorus, either directly as iron phosphate or via adsorption to ferric oxide hydroxides (Stumm & Morgan, 1981; Wetzel, 1983).

Nitrate treatment increases the activity of ubiquitous bacteria and their natural processes at the sediment surface. Through nitrate treatment, oxygen uptake of sediments is reduced, the anaerobic sapropelic sediments are stabilised and putrefaction, sulphate reduction and methane fermentation is diminished. Hydrogen sulphide formation is lowered and sulphides already present can be oxidised bacterially by nitrate to sulphate:

$$5\ S^{2-} + 8\ NO_3^- + 3\ H^+ + H_2O \rightarrow 5\ SO_4^{2-} + 4N_2 + 5\ OH^-$$

The detoxification of anaerobic sapropelic sediments, accompanying this process, allows renewed colonisation of the sediments with benthic fauna. Iron present as sulphide is also transformed into oxidised, trivalent iron by denitrification. As described earlier, the increased phosphorus binding capacity of sediments causes lower internal phosphorus release and lower nutrient supply to the pelagic zone in eutrophic lakes (Ripl, 1976, 1978).

In a carefully planned nitrate treatment, a single or double treatment is sufficient to achieve permanent improvement. However, the quantity of nitrate (and iron) should be adjusted to ensure that the phosphorus concentration in the water declines to such a low level that renewed algal blooms and sedimentation of biodegradable organic matter do not result in renewed feedback establishing phosphorus release and planktonic production.

**Techniques**

Since nitrate treatment significantly affects the metabolism of a lake, pre-treatment and accompanying investigations are necessary to plan and control for the extent of the intervention required. Sediment experiments are needed to calculate the preparatory quantities of nitrate to be used for oxidation of biodegradable organic matter and of iron available for phosphorus binding. If nitrate treatment is to be effective, the iron content of the sediment must be high enough, i.e. well above the lithospheric average of approximately 50 mg $g^{-1}$ of mineral substance (Mackereth, 1966).

Nitrate treatment can be used in both deep and shallow lakes and is normally performed during the last phase of spring circulation and should be finished within a maximum of two months. If combined with iron, iron addition should precede that of nitrate. A solution of calcium nitrate is used, since calcium has a stabilising effect on sediments. This can be dosed into deep water under slow water circulation (Fig. 10.4).

Commercially produced nitrate as well as the outflow from sewage treatment plants may also be used. With the latter, nitrified and clarified water with low phosphorus should be added above anaerobic sapropelic sediments over a longer period (Ripl, 1985). Solid calcium nitrate has also been used (Søndergaard *et al.*, 2000), but this can be difficult to distribute evenly and may lead to negative effects. Since nitrate is generally used up within a few weeks, the water exchange should preferably last from weeks to months. If water residence time is short, the exchange time may be prolonged (e.g. by physical inclusion within a sheet pile wall or rubber apron) or nitrate may be injected directly into the sediment (Fig. 10.5).

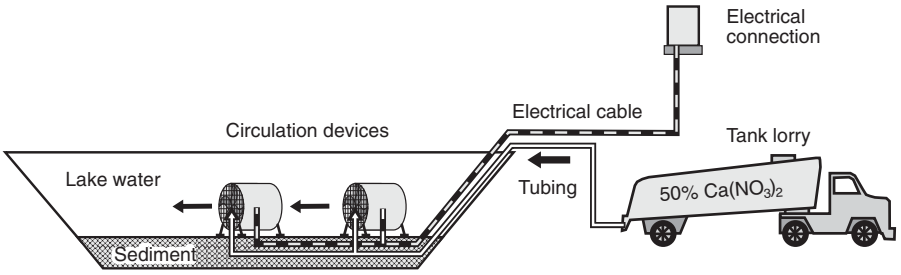



Fig. 10.4. An Example of treatment with calcium nitrate: distribution of nitrate by circulation devices.

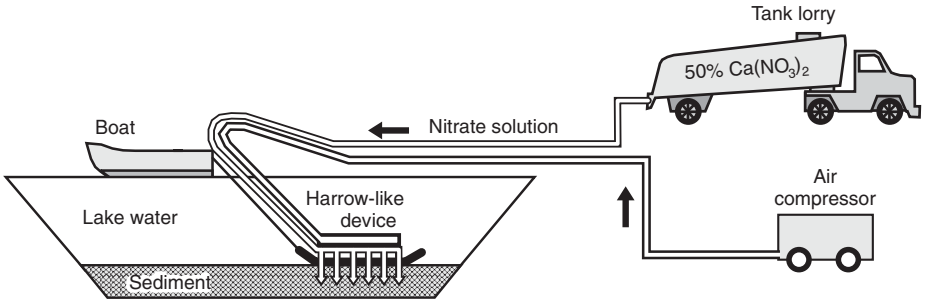



Fig. 10.5. An example of treatment with calcium nitrate: injection into the sediment.

**Possible problems arising from nitrate treatment**

The nitrate used might also act as a nutrient for phytoplankton and lead to increased growth if nitrogen were limiting. However, when phosphate is the target limiting factor for algal growth, nitrate should not increase phytoplankton biomass. A sufficient reduction of external phosphorus loading is crucial for successful nitrate treatment to avoid planktonic algal blooms. If sedimentation of fresh organic matter is not reduced, phosphorus release from the sediments via bacterial metabolism may recur causing feedback reinforcement of planktonic production. Furthermore, release of ammonium from nitrate ammonification should not occur, because ammonium usually originates from reduction of organic matter (Gottfreund & Schweisfurth, 1982). In fact, it has been reported that high ammonium concentrations in interstitial water decrease during and after nitrate treatment (Ripl, 1978, 1986*a, b*).

Where lake water is used for a potable supply, water-quality standards for nitrate may be exceeded by nitrate treatment. However, there is virtually no danger of nitrate export into groundwater, since hydraulic gradients usually enable only infiltration of groundwater into the lake and not vice versa, and efficient denitrification occurs in the lake sediment.

In some cases, there may be a short-term increase in nitrite and gaseous nitrogen oxide concentrations as intermediate products of the nitrification–denitrification process. However, gaseous nitrogen oxides are metabolised in the aqueous environment so quickly that major release is avoided. Consequently, damage to sensitive fish fauna has never been observed. Rapid metabolism of any trace metals released also occurs. For example, through the oxidation of sulphides, even sulphur-bound trace metals can be dissolved. Moreover, pH rises during the process of denitrification and since nitrate treatment is undertaken in iron-rich systems, binding of trace metals occurs in hydroxides or iron-trace-metal hydroxides.

## Iron addition

### Aim and chemical background

Iron addition is used to increase the iron buffer within the sediment and is often used in combination with nitrate (see above) (Ripl, 1976; Donabaum *et al.*, 1999; Dokulil *et al.*, 2000). In lakes with low iron content and low biodegradable organic matter in the upper sediments, iron addition may suffice without simultaneous nitrate treatment (see above). Such conditions are often found in very shallow lakes in which water movement and oxygen supply of the sediment surface are sufficient all year.

The objectives of iron treatment are: (1) precipitation of phosphorus from the water body; (2) increase of the sediment's phosphorus-binding capacity; and (3) decontamination or precipitation of surplus hydrogen sulphide. In many eutrophic lakes, sulphate reduction in the sediment plays a substantial role in the oxidation of organic substances. The $H_2S$ generated is on the one hand toxic for benthic fauna but, on the other hand, performs a ligand exchange with phosphate when binding to iron (Hasler & Einsele, 1948; Brümmer, 1974). At increasing sediment iron concentrations, the buffer capacity for decontamination and $H_2S$ binding increases, with negative consequences only becoming apparent.

In contrast to other adsorbents (e.g. aluminium), iron forms a dynamic phosphate- and sulphide-binding redox buffer at the sediment/water boundary layer. Dissolution of iron in deeper sediment layers occurs at small redox potentials and, in the absence of $H_2S$, increases its concentration in interstitial water. When iron is transported upwards along the concentration gradient, it accumulates in the boundary layer between the reductive and oxidative zone. This zone is also particularly active in the processes of phosphorus release and binding in which iron may constitute a dynamic sink for phosphorus.

When iron(III) chloride is introduced, hydrolysis occurs with the formation of gelatinous flocs of ferric hydroxide, which may go on to form a mixture of iron oxide and hydroxide with dewatering. Phosphorus binds to iron either by adsorption to these iron flocs or as iron phosphate (Stumm & Morgan, 1981):

$$\text{hydrolysis: } FeCl_3 + 3H_2O \rightarrow Fe(OH)_3 + 3H^+ + 3\,Cl^-$$

$$\text{dewatering: } Fe(OH)_3 \rightarrow FeO(OH) + H_2O$$

formation of iron phosphate:

$$FeO(OH) + H_3PO_4 \rightarrow FePO_4 + 2\,H_2O$$

adsorption to iron oxide–hydroxide:

$$FeO(OH) + PO_4^{3-} \rightarrow FeO(OH) \sim PO_4^{3-}(aq)$$

Orthophosphate is best precipitated by this reaction. In contrast to treatment by concentrated aluminium (Cooke *et al.*, 1986), co-precipitation of organic particles does not usually occur during iron treatment. Therefore, iron treatment during an expanded planktonic algal bloom may not successfully precipitate phosphorus.

Trivalent ferric oxide–hydroxide reacts with 1.5 mole hydrogen sulphide per mole iron. First, trivalent iron is reduced by $H_2S$ to Fe(II) and 0.5 $S^{2-}$, which then precipitates with sulphide ions:

binding of hydrogen sulphide:

$$2\,FeO(OH) + 3\,H_2S \rightarrow 2\,FeS + S^0 + 4\,H_2O$$

A combined technique of phosphorus precipitation by hypolimnetic injection of a $FeCl_2$ solution in combination with transport of hypolimnetic water rich in free carbon dioxide into the upper layers to reduce *Microcystis* blooms has also been used (Deppe *et al.*, 1999).

**Techniques**

To understand iron treatment attempts, pre-treatment, accompanying and post-treatment monitoring has to be undertaken. This must include spatial and temporal distribution of phosphorus fractions (phosphate, particulate and organically bound phosphorus). The most suitable time for iron treatment can be determined from the annual circulation pattern and it should be effected when the phosphate fraction reaches its relative maximum (Cooke *et al.*, 1986), usually in late autumn to early spring. The dose should contain sufficient iron for phosphorus binding even if sulphate reduction should occur. In relation to phosphorus in water, iron should always be utilised over-stoichiometrically.

Most frequently, iron(III) chloride ($FeCl_3$) has been used as the iron salt, although iron sulphate has also been used (Daldorph, 1999). The latter may be less suitable because of its negative effect on sediment sulphate reduction. As a consequence of hydrolysis of acid iron bonds and the associated formation of protons, the buffer capacity (alkalinity) of the water is particularly important. With hydrolysis of iron(III) chloride, about 3 moles of protons per mole iron chloride are formed. At low alkalinity, this acid must be buffered or neutralised by the addition of fine particles of lime (calcium carbonate) with high reactivity:

$$\text{hydrolysis: } FeCl_3 + 2\,H_2O \rightarrow FeO(OH) + 3\,H^+ + 3\,Cl^-$$

$$\text{neutralisation: } 6\,H^+ + 3\,CaCO_3 \rightarrow 3\,CO_2 + 3\,H_2O + 3\,Ca^{2+}$$

Acid-forming iron bonds should be used with caution if dosing is to take place from a helicopter or aeroplane, since they can be transported as fine dust which may damage the surroundings.

Apart from adding iron as soluble iron salts, ferric oxide–hydroxides in a solid paste can be used. The latter may originate from the processing of iron-rich ground and mine waters for drinking supply, and may be dispersed by machine on winter ice cover, if this is thick enough. However, since the structure of the ice changes after the treatment, dispersal should take place rapidly and no further visits on to the ice should be undertaken. Ferric

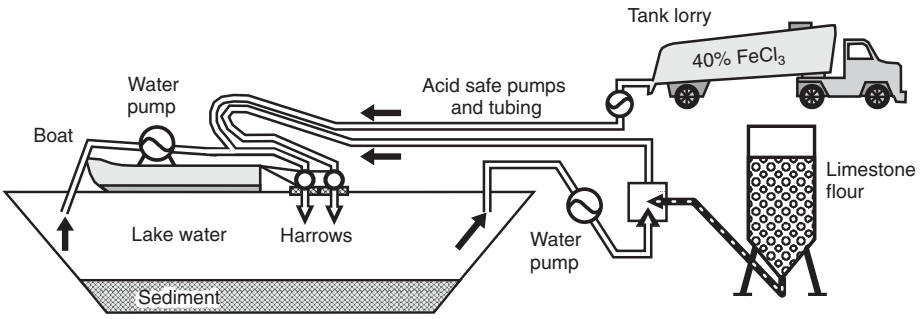



Fig. 10.6. Exemplary scheme of iron treatment with $FeCl_3$ and lime.

oxide–hydroxides should not contain strongly aged, drained material which transforms to oxides that cannot ensure phosphate and sulphide binding. In addition, undesirably high concentrations of phosphorus can be found in some preparations. Solid ferric oxide–hydroxide should therefore always be tested as to its suitability.

Usually iron is added in dissolved form. It can be brought to the lake as a concentrated solution, be mixed ashore and subsequently mixed with lake water on barges or in the boat, immediately prior to dosing. If added from land, the preparation can be pumped to a boat with, for example, acid-resistant equipment (pumps and flexible high-pressure polyethylene tubings). From the boat it is injected over several harrow-like arranged injection nozzles below the water surface (Fig. 10.6). The laborious direct injection of iron into the sediment, as was used in Lake Lillesjön, Sweden and Lake Schlei, Germany (Ripl, 1976, 1978, 1985) has since been proved unnecessary. The distribution of iron over the sediment should be as even as possible. If uneven, the method is less efficient. During precipitation of phosphorus, a clear reduction of total phosphorus to a level below 30-40 $\mu g\ l^{-1}$ should be the target. At higher levels the subsequent algal bloom could jeopardise the restoration attempt.

#### Possible problems in connection with iron treatment

As with the other restoration techniques, iron treatment should be preceded by a significant reduction of catchment nutrient loading to achieve effective and long-lasting results. Otherwise, the positive effects of the treatment may disappear in a few months as a result of continued external loading (Boers *et al.*, 1994).

The lake under treatment should be closely monitored to respond to any adverse effects of treatment. The most important of these is lowering of pH, which could change biological structure. However, toxic effects on fish and the benthic community are rarely observed if the pH does not drop below 6. Reduction of pH may also be prevented by the addition of lime (Ripl, 1976; Dokulil *et al.*, 2000) (see below). Other minor effects include the potential for brown staining of the water column during and, for a few days, after treatment, and a short-term increase in the occurrence of iron floc. Bathing should be directed to other areas for the sake of operational safety.

When adding iron chloride, the chloride concentration in the water column rises to 50 to several hundred milligrams per litre depending on hydraulic retention time and dose(s) used. This sort of increase is generally thought to be unimportant. For example, drinking-water limit values for chloride are *c.* 250 $mg\ l^{-1}$, and even an increase to 500 $mg\ l^{-1}$ or more will probably not result in any biological damage (Schönborn, 1992).

### Alum treatment

#### Aim and chemical background

Aluminium sulphate ($Al_2(SO_4)_3$) or alum has been used for decades to precipitate and increase the sorption capacity of phosphorus and to remove it from internal cycling (Dunst *et al.*, 1974; Cooke & Kennedy, 1978). Alum treatment may be used in

both stratified and unstratified lakes. Also, but less frequently, combined iron–aluminium additions in the form of ferric aluminium sulphate have been used (Foy, 1986; Foy & Fitzsimons, 1987). Aluminium complexes and polymers have the advantage over iron of requiring a low redox potential for the reduction of insoluble $Al^{3+}$ to soluble $Al^{2+}$, meaning that adsorbed phosphorus will not be released from the sediment during periods of anoxia (Foy, 1986; Welch *et al.*, 1988). Alum treatment aiming to reduce the amount of natural organic matter has also been investigated in reservoirs used for drinking supply (Chow *et al.*, 1999).

When added to water, alum forms an aluminium hydroxide complex ($Al(OH_3)$), which has a cotton-like appearance called 'floc' (Dunst *et al.*, 1974; Soltero & Nichols, 1981; Cooke *et al.*, 1993):

$$Al^{3+} + H_2O \rightarrow Al(OH)^{2+} + H^+ + 2\,H_2O$$
$$\rightarrow Al(OH)_3 + 3H^+$$

Phosphorus adsorbs to the floc and sinks to the bottom where it can be permanently removed from the phosphorus cycle and fixed and buried in the sediment. If alum treatment is capable of transforming loosely sorbed and iron-bound phosphorus to aluminium-bound phosphorus, it may reduce the internal phosphorus loading caused by anoxia in the hypolimnion (Ryding & Welch, 1998). The floc also tends to physically entrap algae and other particulate matter (Soltero & Nichols, 1981; Connor & Martin, 1989).

### Techniques

Alum is usually applied as concentrated liquid alum which is dispersed into the lake from a small boat or pontoon barges (Soltero & Nichols, 1981; Foy, 1986; Cooke *et al.*, 1993). Alum may be injected at prescribed depths and into different parts of the lake to facilitate complete coverage and obtain maximum effect. In most cases, aluminium is added in quantities ranging from 5 to 100 g Al $m^{-2}$ or 5 to 25 g Al $m^{-3}$ (Welch & Cooke, 1999; Ryding *et al.*, 2000). The amount added may be adjusted according to alkalinity in the lake and mobile sediment phosphorus concentrations.

Aluminium in the sediment of alum-treated lakes is usually indistinguishable and the alum floc is believed to settle gradually through the usually low-density sediments of most lakes and become buried by newly formed sediment (Welch & Cooke, 1999). In some cases, aluminium has been detected in the sediment at a depth corresponding to the time of treatment (Ryding *et al.*, 2000).

### Possible problems in connection with treatment and their effects

Alum is normally added as a single treatment based on the current water and sediment phosphorus content, implying that the capacity to adsorb further phosphorus will eventually cease. Thus, a single alum treatment usually does not have a long-term effectiveness. If the external phosphorus loading remains high or is not reduced sufficiently, only a short-term effect on lake trophic state can be expected. In most cases the longevity of an effective treatment has been reported to last for about ten years, fluctuating between one and 20 years (Welch & Cooke, 1999). Treatments have had greater longevity and been more successful in stratified rather than unstratified lakes (Foy, 1986; Welch *et al.*, 1988; Welch & Cooke, 1999). However, treatments in shallow lakes are more certain to affect phosphorus availability in the photic zone than in stratified lakes where sediment-released phosphorus to the hypolimnion is unavailable.

Aluminium hydroxy complexed phosphorus is sensitive to pH, and phytoplankton or macrophyte-induced photosynthetically elevated pH has been blamed for the failure of one alum treatment in a shallow lake (Welch & Cooke, 1999). Dense macrophyte beds may also diminish the effectiveness of the treatment as they may cause uneven floc distribution or sediment phosphorus recycling from below the floc layer through plant senescence and decay (Welch & Cooke, 1999). Depending on the dosage, alum treatment may elevate sulphate levels and thereby lead to increased hydrogen sulphide production eventually reversing the lake back to eutrophy (Soltero & Nichols, 1981).

Acidification of alum-treated lakes to below pH 6 may result in increased aluminium concentrations and adverse toxic effects associated with enhanced metal solubility (Soltero & Nichols, 1981; Cooke *et al.*, 1993). At pH 4 to 6, various soluble intermediate

forms occur, while at a pH below 4, soluble $Al^{3+}$ dominates. This form is particularly toxic to biota (Cooke *et al.*, 1993). Because hydrogen ions are liberated when alum is added, pH decreases in the lake water at a rate depending on alkalinity. To avoid toxic effects, the maximum dosage of alum has been defined as the maximum amount of aluminium which, when added to lake water, would ensure a dissolved aluminium concentration below 50 $\mu g\, l^{-1}$ (Cooke & Kennedy, 1981; Kennedy & Cooke, 1982). Buffering agents, such as sodium aluminate and sodium bicarbonate, have been added in treatments of soft-water lakes to maintain pH above 6. Adverse effects in terms of reduced invertebrate populations have usually not been observed and no fish kills have been reported (Cooke *et al.*, 1993). Usually several days are required to treat a 300-ha lake so there is ample time for fish to avoid areas of water disturbance.

## Lime treatment to reduce eutrophication

### Aim and chemical background

Slaked lime (calcium hydroxide ($Ca(OH)_2$)) has been added to eutrophic lakes to diminish phosphorus availability by the formation of calcite (calcium carbonate, $CaCO_3$) and the precipitation of phosphate into insoluble Ca–$PO_4$ complexes (hydroxyapatite):

$$10\ CaCO_3 + 6\ HPO_4^{2-} + 2\ H_2O \rightarrow Ca_{10}(PO_4)_6(OH)_2 + 10\ HCO_3^-$$

In the short term (<15 days), calcium hydroxide treatment may also directly decrease phytoplankton biomass and chlorophyll *a* through the precipitation of phytoplankton cells or colonies (Zhang & Prepas, 1996).

Co-precipitation of inorganic phosphorus with calcite in hard-water lakes is a natural process usually triggered by an increase in pH caused by photosynthesis (Otsuki & Wetzel, 1972; Murphy *et al.*, 1983; Hartley *et al.*, 1997). It is believed that phosphate initially adsorbs to the surface of calcite crystals and later becomes incorporated into the crystal during crystal growth (Kleiner, 1988; House, 1990). Calcite sorbs phosphate especially when pH exceeds 9 and hydroxyapatite has its lowest solubility at high pH (>9.5).

### Techniques

Fundamentally, lime application involves the same techniques as those developed for sewage treatment plants. *In situ* lake treatment, however, cannot be controlled similarly (i.e. by adjusting pH) and one of the problems is to find an application technique promoting carbonate precipitation at an acceptable pH remaining within its natural range (Murphy *et al.*, 1988).

Lime is usually added from a boat as a slurry of hydrated lime mixed with water, which is then sprayed over the lake surface or injected at a depth of a few metres. Alternatively, lime has been injected into the hypolimnion in combination with aeration, in order to shift the equilibrium of the calcite–carbonic acid systems towards calcite precipitation in the hypolimnion (Dittrich *et al.*, 2000). Repeated low-dose treatments or a single high-dose treatment have been used. Calcium hydroxide dosage normally ranges from 25 to 300 mg $l^{-1}$. When the calcium hydroxide slurry has been added, large particles will sink through the water column while small particles dissolve in the water and form calcite (Zhang & Prepas, 1996).

Less frequently, calcite or calcite-rich lake marl taken from natural deposits in the littoral zone or from the lake sediment and then spread over the lake surface has been used as an alternative to slaked lime in order to co-precipitate phosphorus (Stuben *et al.*, 1998; Hupfer *et al.*, 2000). Calcite is generally believed to have a lower capacity to sorb phosphorus than lime, where freshly nucleated calcite crystals are generated in the presence of phosphate.

### Possible problems in connection with treatment and their effects

Turbidity increases after the lime treatment, but usually only for a few days at most. Lime treatment and the following pH shock may, however, have a negative impact on the macroinvertebrate community and other animals, and may last for a year or more after the treatment (Miskimmin *et al.*, 1995; Yee *et al.*, 2000). The extent of pH elevation after the addition depends on the buffering capacity of the lake and the dosage applied. In hard-water lakes, it is usually possible to keep pH below 10, while in soft-water lakes pH may increase to above 11 (Zhang & Prepas,

1996), this having severe implications for most organisms.

The affinity of calcium to sorb phosphorus in natural systems is relatively low compared with elements like iron. Several studies have thus shown that there is no relationship between the calcium carbonate content in the sediment of lakes and the amount of calcium carbonate-bound phosphorus (Søndergaard *et al.*, 1996; Rzepecki, 1997; Gonsiorczyk *et al.*, 1998).

Phosphorus precipitated with calcium carbonate may redissolve and thus prevent permanent effects of the treatment. Redissolved calcite may reprecipitate later as conditions change, establishing more long-term mechanisms (Murphy *et al.*, 1988). The solubility of precipitated phosphate increases in the hypolimnion and close to the sediment, where bacterial respiration causes lowered pH (Driscoll *et al.*, 1993).

## Other chemical methods combating eutrophication

A number of other chemicals have been used to increase the binding capacity of phosphorus, many of them being relatively inexpensive industrial by-products.

Gypsum ($CaSO_4{}^*2\ H_2O$) has been used in a few cases in a parallel manner to the use of calcium hydroxide to establish calcium–phosphorus compounds (hydroapatite) and reduce phosphorus release from the sediment (Wu & Boyd, 1990; Salonen & Varjo, 2000). However, the addition of sulphate may, in the longer term, lead to increased internal loading via the ligand exchange of sulphide and iron-bound phosphorus.

Slag, a by-product in the refining process of iron ore with caustic lime (Yamada *et al.*, 1986), has also been used. It contains large amounts of calcium and other elements like aluminium and iron that can be used to adsorb dissolved inorganic phosphate. Clay and fly ash have also been considered as sorption agents for phosphorus (Dunst *et al.*, 1974).

## Liming to adjust pH

### Aim and chemical background

Liming or base addition of acidified lakes is used to counteract and mitigate the decrease of pH in lakes where the acid deposition exceeds buffering capacity. Liming is thus used to enhance or prevent a decrease in species richness and species diversity, and to ensure that the natural fauna and flora can survive or recolonise (Stenson & Svensson, 1995; Nyberg, 1998). Normally, the aim is to raise pH to above 6 and the alkalinity to 0.1 meq $l^{-1}$, in order to establish an acceptable buffering capacity (Svenson *et al.*, 1995).

Most often, limestone powder or gravel (calcite, $CaCO_3$) and, less often, other buffering agents such as magnesium (dolomite, $CaMg(CO_3)_2$), are added to increase the cation pool. Wood ash from forest residues has been used alternatively in catchments, adding, besides calcium, also a number of other buffering elements (Bramryd & Fransman, 1995; Fransman & Nihlgård, 1995).

Apart from restoring faunal diversity, liming can also cause a net precipitation of phosphorus equivalent to the effects seen after the addition of slaked lime (Smayda, 1990). Liming also promotes the precipitation of aluminium, iron and manganese (Andersen & Pempkowiak, 1999), which may influence phosphorus availability. The reverse process can also be seen, as increased pH in the catchment may lead to increased phosphorus availability by stopping the acidification process which tends to increase the precipitation of phosphorus with aluminium in the soil matrix (Broberg & Persson, 1984). Increased pH may also increase the mineralisation rate in the sediment and the release of nutrients (Dickson *et al.*, 1995; Roelofs *et al.*, 1995).

### Techniques

Liming using limestone powder or dolomite powder is mostly applied directly to the lake, but liming can also be conducted as a watershed treatment depending on hydrology (Svenson *et al.*, 1995). In-lake treatment is often conducted from a boat connected with a pipeline to a container on shore. Small lakes can be limed manually from a boat or on the ice during winter. Lakes situated in remote areas may be limed from a helicopter. Wetland liming can be used as a supplement to lake liming. Rivers can be limed by continuous automatic dosers (Sandøy & Romundstad, 1995).

### Possible problems in connection with treatment and their effects

Liming is usually regarded as a temporary solution where no permanent effects are established. Contrarily, if the loading of acids to the lake continues, pH will eventually decrease again unless reliming is conducted.

Increased nutrient mobilisation effected by liming can cause internal eutrophication in shallow lakes followed by changes in the macrophyte and plankton community (Brandrud & Roelofs, 1995; Dickson *et al.*, 1995). Some plants (e.g. *Juncus bulbosus*) may benefit from higher nutrient concentrations in limed lakes when the carbon dioxide concentrations in the water are relatively high, as is the case after reacidification (Lucassen *et al.*, 1999). When liming in wetlands, undesirable effects, such as changes in mosses and lichens (Svenson *et al.*, 1995), may occur. Liming may lead to decreased transparency, but usually the lake water returns to the pre-treatment situation (Pulkkinen, 1995).

## CONCLUDING REMARKS

Numerous chemical restoration measures have been developed to combat resilience in lake recovery during the past decades. In many lakes, internal loading of phosphorus from lake sediments prevents improvements in water quality despite a reduction of the external loading. In the case of acidification, liming has been used to counteract a pH decrease in lakes where the acid deposition exceeds the buffering capacity.

For both types of restoration measures, an important prerequisite for obtaining success and long-term effects is elimination of the underlying reasons for the undesirable water quality; i.e. a sufficient reduction of external phosphorus loading in the case of eutrophication, and a decline in the deposition of acids in the case of acidification.

Restoration measures to neutralise eutrophication effects focus on either increasing the phosphorus sorption capacity of compounds (especially of iron by improving redox conditions) already present in the sediment, or on increasing the sorption capacity by the addition of new sorption capacity (mainly alum, iron and calcium).

Five categories of chemical restoration measures can be summarized:

1. Hypolimnetic oxygenation, with pure oxygen or atmospheric air being injected into the hypolimnion with various types of equipment, to improve the redox sensitive sorption of phosphorus to iron and the living conditions of benthic animals.
2. Oxidation of the hypolimnion and the sediment using nitrate as an electron acceptor to oxidise organic matter in the sediment and improve the sorption of phosphorus to iron by preventing the formation of iron sulphide.
3. Addition of iron to increase the phosphorus sorption capacity of the sediment, this often being used as a supplement to oxidation with nitrate or oxygen.
4. Alum treatment to increase the phosphorus sorption capacity by increasing the non-redox sensitive binding of phosphorus to aluminium hydroxides.
5. Addition of slaked lime to increase the formation of calcite and the precipitation of phosphorus into hydroxyapatite.

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# Hypolimnetic Nitrate Treatment to Reduce Internal Phosphorus Loading in a Stratified Lake

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## ABSTRACT

Søndergaard, M., E. Jeppesen and J. P. Jensen. 2000. Hypolimnetic nitrate treatment to reduce internal phosphorus loading in a stratified lake. Lake and Reserv. Manage. 16(3):195-204.

Nitrate [$Ca(NO_3)_2$] was added to the hypolimnion of eutrophic Lake Lyng (10 ha, maximum depth 7.6 m), Denmark, over a 2-year period to study the impact on sediment phosphorus release and to evaluate the potential of hypolimnetic nitrate dosing as a lake restoration method. Dissolved (1995) or granulated (1996) nitrate was added 10 to 11 times at a depth of 5 m during stratification. The total dose was 8 to 10 g N $m^{-2}$ $yr^{-1}$. Hypolimnetic nitrate concentration was <0.01 mg N $L^{-1}$ in years without dosing (1994 and 1997), and reached 1.2 and 2.2 mg N $\cdot L^{-1}$ in treatment years, respectively. Maximum concentrations of dissolved inorganic phosphorus (DIP) in the hypolimnion were 1.6 to 2.9 mg P $\cdot L^{-1}$ in untreated years, but were reduced to 0.8 to 1.2 mg P $\cdot L^{-1}$ in treatment years. Correspondingly, total phosphorus (TP) accumulation in treatment years was 48-77% that of non-treatment years. Dissolved nitrate seemed to reduce TP concentrations more efficiently than granulated nitrate, which sank partly into the loose sediment, and led to lower nitrate as well as lower iron concentrations in the hypolimnion. Lower iron release and/or lower nitrate concentrations resulted from the use of dissolved nitrate, which seemed to reduce precipitation of phosphate with iron and left a larger proportion of particulate phosphorus in the hypolimnion. Ammonia concentrations were higher with nitrate addition, probably due to enhanced mineralization of organic sediment and nitrate ammonification. The results from Lake Lyng suggest that the potential of using nitrate as a lake restoration method in deep lakes suffering from internal loading needs further investigation.

Key Words: lake restoration, hypolimnion, internal loading, phosphorus, nitrate dosing.

During recent decades, numerous lake restoration methods have been developed to combat the effects of eutrophication and to hasten the recovery process after an external load reduction (Cooke et al. 1993). Most methods aim at diminishing the internal loading of phosphorus released from the sediment where large phosphorus pools had accumulated when external loading was high. Case studies have documented that such loading may persist and prevent lake recovery for several years after a reduction of external loading (Marsden 1989, Sas 1989, Jeppesen et al. 1991, Søndergaard et al. 1999).

In deep and stratified lakes, most restoration methods focus on improving the capacity of the sediment to retain phosphorus, either by adding phosphorus-binding substances, such as aluminum (Cooke and Kennedy 1978, Welch and Schrieve 1994, Sonnichsen et al. 1997) and iron (Winszuk and Kalbe 1978, Yamada et al. 1987, Boers et al. 1994, Wisniewski 1999), or by improving the phosphate binding capacity in the sediment. The latter method usually aims at improving the redox-sensitive binding to iron (Mortimer 1941, Jensen et al. 1992, Søndergaard et al. 1996).

Hypolimnetic oxygenation is widely used to increase the sorption capacity of phosphorus (McQueen et al. 1986, Cooke et al. 1993, Prepas and Burke 1997). In the "Riplox Method" (Ripl 1976), nitrate is added to the sediment to enhance mineralization of organic matter and to improve redox-sensitive sorption of phosphorus to iron in the surface sediment. Nitrate is injected directly into the top 15 to 20 cm of the sediment with a "harrow" which is dragged along the lake bottom. Nitrate dosing may be supplemented with iron to increase the sediment P-binding capacity (Ripl 1976), unless the natural sediment content of iron is found to be sufficiently high (Foy 1986). The use of nitrate instead of pure oxygen has the advantage that a larger quantity of electron acceptors can be added over a short period or possibly only in one injection. The method has, however, been subject to criticism as the sediment phosphorus release, particularly in shallow and non-stratified lakes, may depend on other factors than redox conditions, such as pH (Petterson and Boström 1982). Moreover, the Riplox method has quite a dramatic effect on several other variables. Nitrate may, for instance, increase to extremely high levels in the interstitial water and sedi-

ment pH may decrease to 3, which implies that lime treatment is necessary to raise pH to promote denitrification (Ripl 1976).

Even though some of the early nitrate treatment experiments in the 1970s looked promising (Ripl and Lindmark 1978), our knowledge of sediment nitrate dosing as a lake restoration method is currently limited. On the laboratory scale, Willenberg et al. (1984) found that injection of calcium nitrate could eliminate virtually all phosphorus release from a sediment previously releasing phosphorus at a high rate. In whole-lake experiments, Foy (1986) found that nitrate addition delayed the onset of sediment release of phosphorus and diminished the maximum quantities in the hypolimnion and Donabaum et al. (1999) used the Riplox method to successfully reduce both TP and chlorophyll *a* in an integrated management strategy to restore the clearwater and macrophyte dominated state.

Nitrate addition to the hypolimnion and deep-water injection of nitrate-rich effluent from advanced wastewater treatment plants as means to reduce internal phosphorus loading (Ripl 1978) are even more sparsely documented methods. An example of the former is Lake Kolbotnvannet in Norway. For most years since the mid-1980s, Lake Kolbotnvannet has received nitrate through a $Ca(NO_3)_2$-enriched inlet and the nitrate was injected into the hypolimnion (Erlandsen et al. 1988, Faafeng et al. 1997). In years with nitrate addition, hypolimnetic TP diminished and nitrate dosing thus seems to have had a positive impact on lake water quality, including hypolimnetic oxygen concentrations.

In this paper we describe the effects of multiple low-dose nitrate addition to the hypolimnion of summer-stratified Lake Lyng, Denmark. By using low doses only, we wanted to control the nitrate concentration to avoid unwanted side-effects, including the risk of enhancing the epilimnion nitrogen input during autumn destratification since phytoplankton production in this lake may be nitrogen-limited. The purpose of the experiment was not to restore the lake, but merely to evaluate whether frequent addition of relatively low doses of nitrate to the hypolimnion could be used as a restoration tool in stratified lakes.

# Lake Lyng

Lake Lyng is situated on the outskirts of the city of Silkeborg, Denmark. The lake area is 10 ha, 0.89 ha having a depth larger than 5 m (Fig. 1). Maximum depth is 7.6 m and mean depth is 2.4 m. The deep part of the lake stratifies during summer, from the beginning of June to the end of September (Fig. 2). The stratification occurs at approximately 4 to 5 m depth, but in calm and warm periods stratification may temporarily occur closer to the surface. The oxycline usually dev-elops at about 4 m, but may extend to 2 m below the lake surface (Fig. 2).

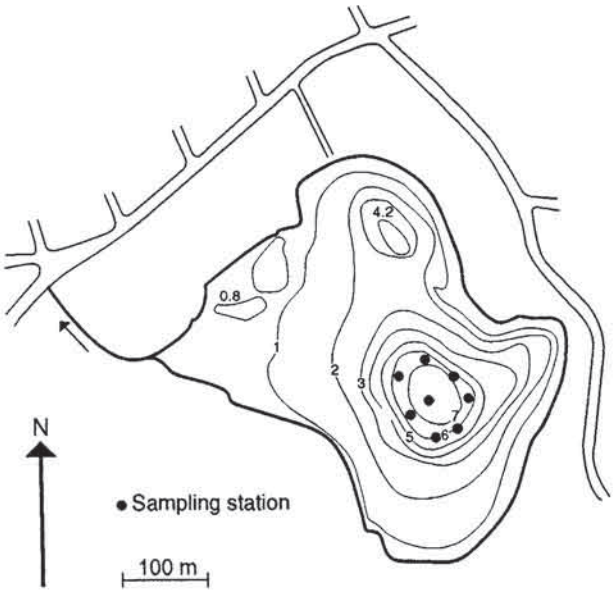



Figure 1.–Depth chart of Lake Lyng showing the nitrate-treated area (depth > 5 m) and sampling stations.

Until the mid-50s, Lake Lyng was polluted with untreated sewage from Silkeborg, but since sewage diversion the nutrient loading has been restricted to diffuse input from a forest catchment, except for periods with heavy rainfall or occasional overflow from sewers. External nutrient supply to the lake is therefore expected to be modest. Hydraulic retention time is >1 year.

Due to the former high sewage loading, the lake is highly nutrient-rich and turbid. During 1989-1994, the mean summer concentration of TP in the surface water varied between 0.42 and 0.79 mg $P \cdot L^{-1}$ and dissolved inorganic phosphorus (DIP) between 0.14 and 0.54 mg $P \cdot L^{-1}$ (Søndergaard et al. 1997). The nitrate concentration was <0.01 mg $N \cdot L^{-1}$ during summer. Average Secchi depth ranged between 0.8 m and 2.5 m.

Phytoplankton was dominated by small colony-forming bluegreens (primarily *Anabaena spiroides* and *A. flos-aque*) during summer. The fish stock was dominated by roach (*Rutilus rutilus*) and small perch (*Perca fluviatilis*), comprising 48% and 46% of the biomass, respectively (Berg et al. 1997). Hardly any submerged macrophytes were present.

The hypolimnetic sediment of Lake Lyng contained high concentrations of TP, reaching 6.2 mg $P \cdot g^{-1}$

DW in the surface sediment (Table 1). Most of the P is bound in organic compounds (Res-P), but a considerable part (38-43% in the upper 15 cm) is extractable with NaOH and most probably bound to iron. The iron concentration is relatively high, being 33 mg Fe $\cdot g^{-1}$ DW in the surface sediment.

# Methods

Nitrate was added to the hypolimnion of Lake Lyng in areas deeper than 5 m in 1995 and 1996, from the end of June to approximately September 1 (Fig. 1). Except from short periods during summer 1995 (1-8 July and 2-21 August), when nitrate treatment was stopped to avoid too high concentrations, the lake was dosed from a boat once every week at 10 to 12 positions. When dosing, which took 1 to 2 min at each position, the boat was allowed to drift freely to increase spreading. To ensure that the nitrate reached the hypolimnion, it was added at a depth of 5 m using a long PVC tube (diameter 12 cm). We used technically clean calcium saltpetre [$Ca(NO_3)_2$] with a nitrogen content of approximately 15% (14% $NO_3$-N and 1% $NH_4$-N). In 1995, dissolved calcium nitrate was used, whereas granulated calcium nitrate was added in 1996 (pellets 2 to 4 mm in diameter). Before dosing, the dissolved nitrate was stored at 4 °C to obtain maximum specific gravity. In 1995 and 1996, 500 and 600 kg $Ca(NO_3)_2$, respectively, were added, corresponding to 75 and 90 kg $NO_3$-N. Assuming that nitrate was distributed evenly at depths below 5 meter, this addition corresponds to 8.4 and 10.1 g N $m^{-2}$, respectively. The years 1994 and 1997, when no nitrate was added, served as reference years.

The effects of nitrate addition were followed by profile measurements of chemical variables at 1 m intervals down to a depth of 6.5 m in the deepest part of the lake (centre station). Furthermore, from June 26 1995 and 1996 and throughout the stratification period, another 7 (1995) or 6 (1996) samples were taken at 6.5 m or 6.0 m depths (hypolimnion stations) placed within a radius of 20 to 40 m from the centre station that covered most of the nitrate-treated area (Fig. 1). Water was sampled weekly with a Patalas

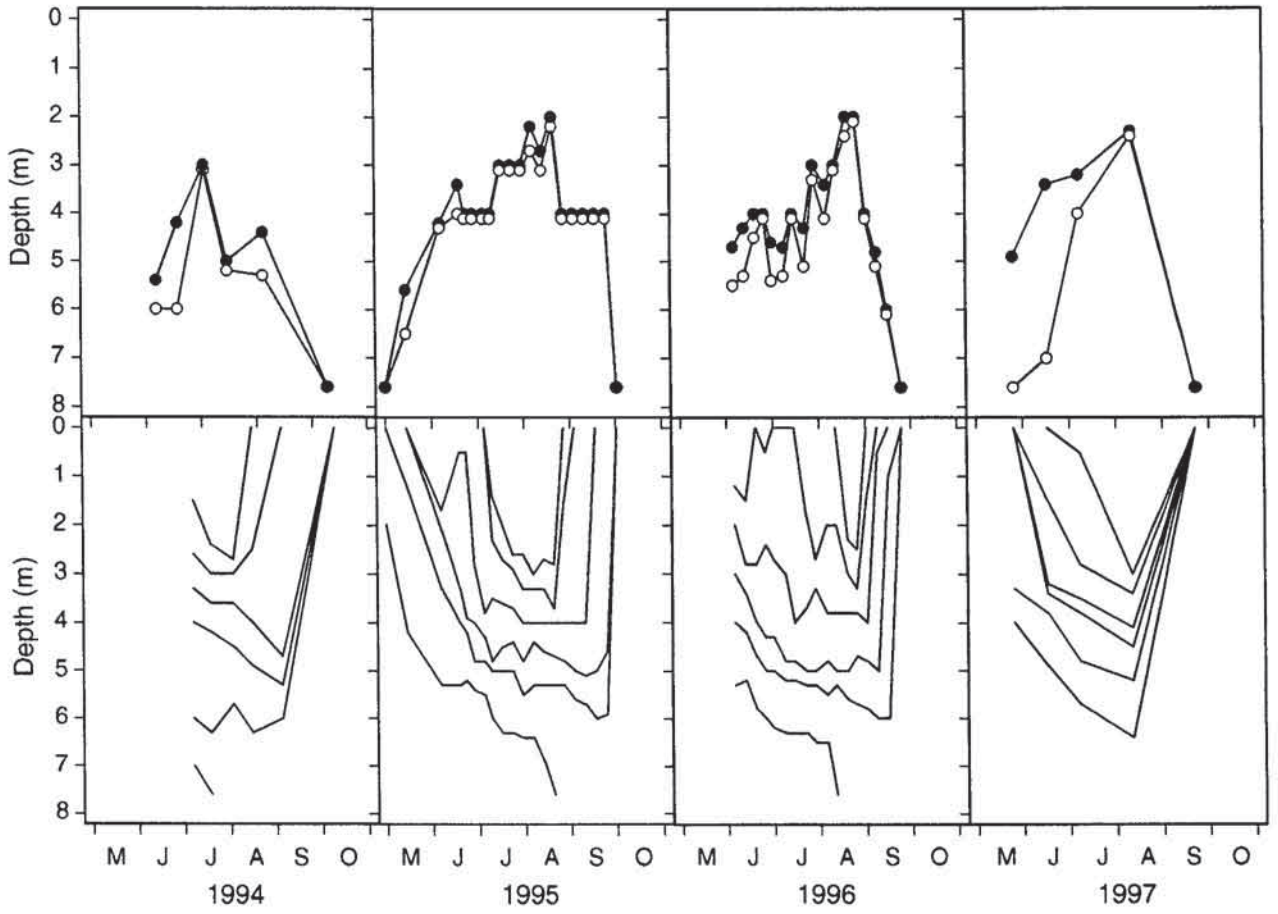



Figure 2.–Depth-time diagrams showing (upper panel): isopleths of dissolved oxygen (1 mg $O_2 \cdot L^{-1}$, o---o and 2 mg $O_2 \cdot L^{-1}$, •---•) and (lower panel): isotherms of 10, 12, 14, 16, 18 and 20 °C.

Table 1.–Sediment characteristics from the deepest part (depth = 7.0 m) of Lake Lyng. Data modified from Thorup and Skjøtt (1989). Phosphorus fractionation was conducted according to Hieltjes and Lijklema (1980) into $NH_4Cl$, NaOH and HCl-extractable phosphorus. Residual phosphorus (Res-P), believed to represent mainly organically bound phosphorus, is calculated as the difference between total phosphorus (Total-P) and $NH_4Cl$-P + NaOH-P + HCl-P. Dry weight (DW) and loss on ignition (LOI) are in % of wet weight and % of dry weight, respectively. Phosphorus concentrations are in mg P · $g^{-1}$ DW and total iron (Fe) in mg Fe · $g^{-1}$ DW. For more detailed information, see Hieltjes and Lijklema (1980) and Søndergaard et al. (1996).

| Depth (cm) | DW | LOI | $NH_4Cl$-P | NaOH-P | HCl-P | Res-P | Total-P | Fe |
|---|---|---|---|---|---|---|---|---|
| 0-2 | 2.2 | 35.7 | 0.006 | 2.48 | 0.003 | 3.70 | 6.19 | 33.4 |
| 2-5 | 3.1 | 37.0 | 0.005 | 2.27 | 0.002 | 3.03 | 5.31 | 32.4 |
| 5-10 | 3.7 | 37.6 | 0.007 | 1.84 | 0.002 | 2.98 | 4.83 | 32.8 |
| 10-15 | 4.4 | 39.3 | 0.006 | 1.93 | 0.003 | 2.68 | 4.62 | 33.1 |

sampler in 1995 and 1996 and 1 to 2 times a month in 1994 and 1997. On each sampling and treatment date, water was sampled prior to nitrate dosing.

TP was analyzed spectrophotometrically as molybdate-reactive P after digestion with 1 M HCl, and DIP on GFC filtered samples. Additionally, TP and DIP analyses were conducted both on filtered (GFC) and unfiltered samples yielding the following fractions: dissolved organic phosphorus $Diss_{or}$-P, calculated as (TP analysed on filtered samples) – (DIP); particulate inorganic phosphorus ($Part_{in}$-P, calculated as (molybdate-reactive phosphorus on unfiltered samples) – DIP); and particulate organic phosphorus ($Part_{or}$-P, calculated as TP – $Diss_{or}$-P – (DIP-reactive phosphorus on unfiltered samples). Nitrite and nitrate ($NO_3$-N) were determined as nitrite on a Tecator 5012 flow-injections analyser supplied with a copper-cadmium reductor column. Ammonia ($NH_4$-N) was determined spectrophotometrically with hypochlorite and total iron (Fe) after oxygenation with peroxidisulphate in acid solution by bipiridyl complex formation. Temperature (T) and dissolved oxygen ($O_2$) were measured with a YSI-58 oxygen meter.

# Results

In years without nitrate addition (1994 and 1997), nitrate concentrations in the hypolimnion of Lake Lyng were below the detection limit (<0.01 mg N · $L^{-1}$) shortly after the commencement of stratification, whereas in years with nitrate addition (1995 and 1996) nitrate reached concentrations up to 1.2 and 2.2 mg N · $L^{-1}$, respectively, at 6.5 m depth (Fig. 3). At the 6-7 sampling stations in the hypolimnion, nitrate concentrations during the major part of the summer of 1996 reached lower concentrations than in 1995 and also lower concentrations than at the centre station – probably because more nitrate pellets were unintentionally dosed in the deepest areas (Figs. 3 and 4). Variations in concentrations between the 6-7 hypolimnion stations, however, were generally low, indicating that the sediment was completely covered by the treatment (Fig. 4). Nitrate concentrations were also lower at 5.5 m than at 6.5 m depth, particularly in 1996 when the mean concentration at 5.5 m was only 0.007 mg N · $L^{-1}$ compared with 0.098 mg N · $L^{-1}$ in 1995 (Table 2).

In the deepest part of the lake, DIP concentrations during most of the summer were lower in treatment years than in years without dosing (Fig. 3). In 1994 and 1997, respectively, up to 2.9 and 1.6 mg DIP · $L^{-1}$ were measured at 6.5 m depth compared with 0.8 DIP · $L^{-1}$ in 1995 and 1.2 DIP · $L^{-1}$ in 1996. In 1996, DIP concentrations at 5.5 m depth were similar to those in non-treatment years and a steady increase was observed during the whole stratification period. In 1995, DIP was considerably lower than in non-treatment years also at 5.5 m depth, reaching a maximum of 0.61 mg P · $L^{-1}$ compared with 1.2 mg P · $L^{-1}$ in 1994 and 1.1 mg P · $L^{-1}$ in 1997.

TP was mostly composed of DIP; DIP comprised between 80 and 87% of hypolimnetic TP during July-August, except for 1995 when mean DIP was significantly lower and only accounted for 60% (Table 3, Fig. 6). Correspondingly, $Part_{in}$-P was markedly higher in 1995 (mean 33%) compared with other years when it ranged from 7 to 18% of TP. Dissolved and particulate organic phosphorus comprised less than 4% of TP in all years. DIP was independent of TP ($P>0.05$), except shortly before and during the 1995 autumn turnover (25 September and 2 October) when there was a strong, significant negative relation ($p<0.001$) between DIP and TP. Hypolimnetic P accumulation was lower in treatment years, especially in 1995 when

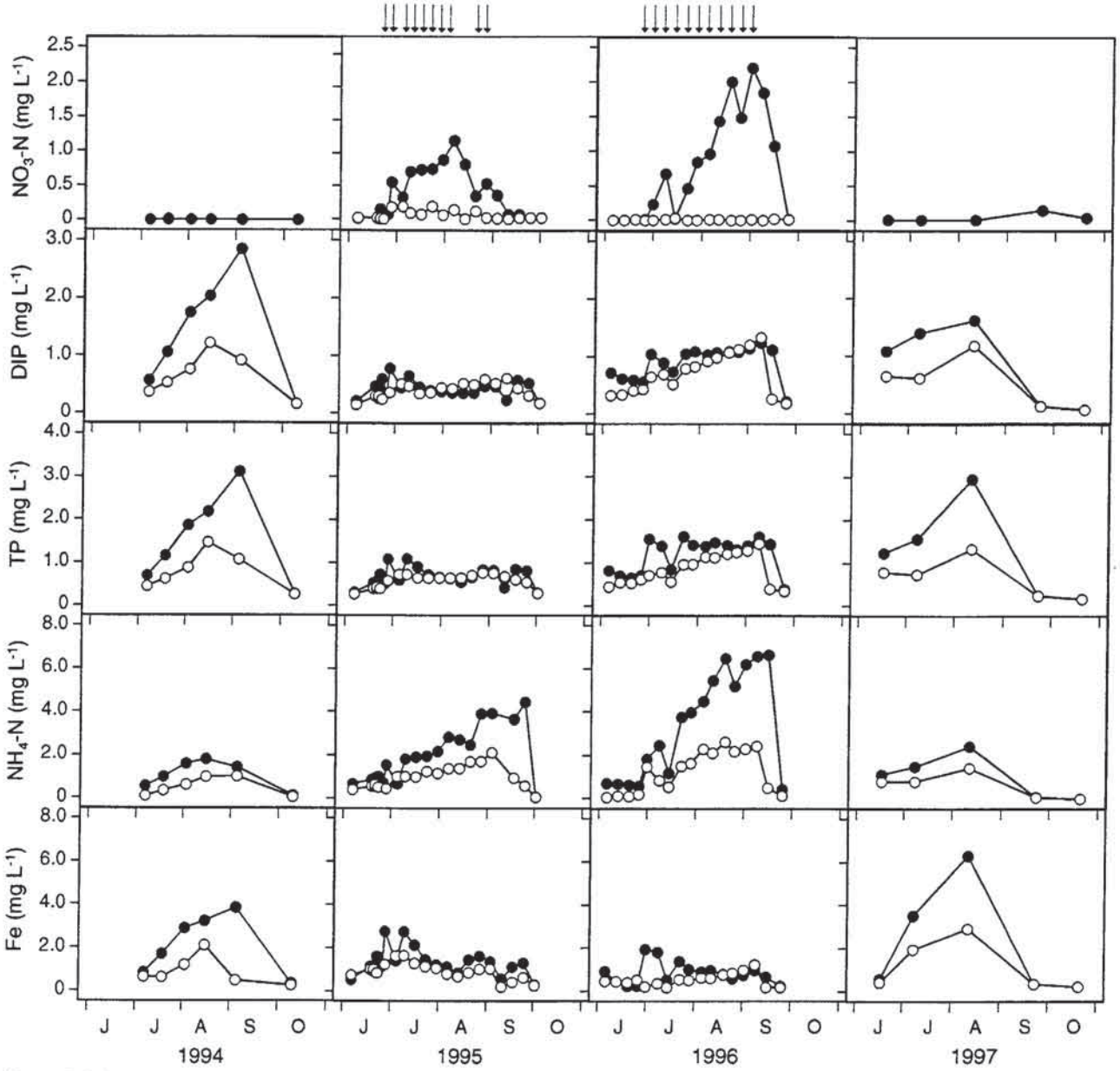



Figure 3.–The concentrations of $NO_3$-N, DIP, TP, $NH_4$-N and Fe at 5.5 (o---o) and 6.5 m (•---•) depth at the centre station in Lake Lyng, 1994 to 1997. Arrows indicate nitrate dosing.

it only reached 35% DIP and 48% of TP of the non-treatment years (Table 4). In 1996, the decrease in phosphorus content was less pronounced.

Ammonia concentrations were highest in treatment years, reaching a maximum immediately before the 1995 autumn turnover of 4.4 mg $N \cdot L^{-1}$ and 6.7 mg $N \cdot L^{-1}$ in 1996 at the centre station compared with 1.8 and 2.4 mg $N \cdot L^{-1}$ in 1994 and 1997, respectively (Fig. 3). A sharp gradient with depth developed particularly in 1995, when concentrations were 2 to 3 times higher at 6.5 than at 5.5 m depth. The hypolimnetic ammonia accumulation was higher in treatment than in non-treatment years, particularly in 1996 when it reached 265% of the non-treatment mean values (Table 4). Hypolimnetic ammonia concentrations were positively related to TP, but the $NH_4$-N/TP ratio differed markedly between the four years (Fig. 5), the mean ratio being 0.7 (SD=0.2), 3.2 (SD=1.4), 2.0 (SD=1.0) and 1.0 (SD=0.1) from 1994 to 1997, respectively.

Hypolimnetic Fe was generally highest in non-treatment years (Fig. 3, Table 2) and the lowest concentrations were recorded in 1996 using pellets. Correspondingly, the maximum accumulation of Fe was lower in treatment years, and particularly in 1996 when it only reached 31% of the non-treatment values (Table 4). The difference recorded between 5.5 and

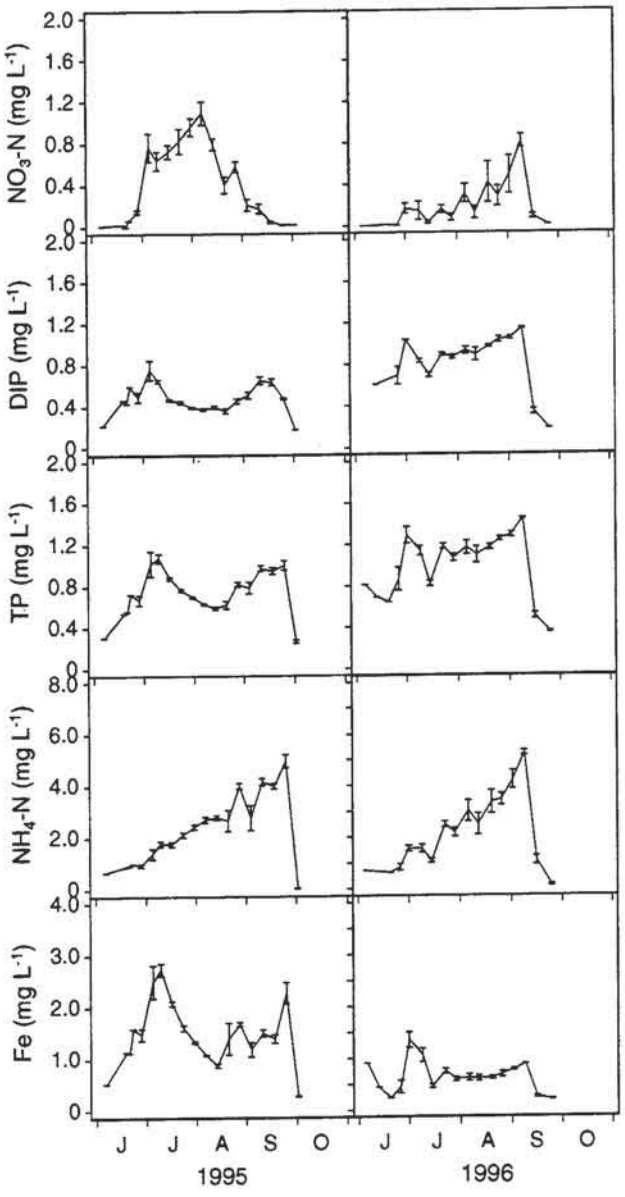



Figure 4.–Mean concentration (+/-SD) of $NO_3$-N, DIP, TP, $NH_4$-N and Fe at 6.5 (7 June 1995 to 15 July 1996 or 6.0 m (23 July - 25 Sep 1996) depth at 6-7 hypolimnion stations. The first four samplings in June 1995 (7-23 June) and the first 3 samplings in June 1996 (6-20 June) are based on centre station measurements only.

6.5 m depth was highest in non-treatment years. Fe related closely and positively to TP, but the Fe/TP ratio tended to differ among years with or without treatment and also among the treatment years (Fig. 5). In 1995, mean Fe/TP ratio in the hypolimnion was 2.0 (SD = 0.4) compared with 0.6 (SD = 0.2) in 1996.

During 1996, a slow but steady increase in hypolimnetic P occurred during the stratification period, whereas DIP and TP decreased periodically during 1995 (Fig. 4). The 1995 decrease began when nitrate concentrations reached approximately 0.5 mg $N \cdot L^{-1}$ in early July and continued around that concentration until mid-August. DIP was reduced from 0.7 to about 0.4 mg $P \cdot L^{-1}$ and TP from 1.0 to 0.6 mg $P \cdot L^{-1}$ during that period. Likewise, the increase in P in mid-July occurred when nitrate decreased to below approximately 0.5 mg $N \cdot L^{-1}$. Fe concentrations followed the pattern of TP and decreased from 2.6 mg $Fe \cdot L^{-1}$ in early July to about 0.8 mg $Fe \cdot L^{-1}$ in mid-August. The decrease in nitrate concentrations observed when nitrate dosing was stopped for three weeks in summer 1995 corresponded to a net loss rate of 0.07 mg $N \cdot m^{-2} \cdot d^{-1}$. In both years, nitrate disappeared from the water within a few weeks after dosing was stopped.

# Discussion

Treatment of the Lake Lyng hypolimnion with nitrate had significant effects on the interactions between sediment and water. Most important, with respect to lake restoration, was that the accumulation of phosphorus and iron was lower in years with, than in years without, nitrate addition. The effects were evident, even though the level of nitrate dosing was relatively low (8 to 10 g $P \cdot m^{-2} \cdot yr^{-1}$) and substantially lower than in Swedish Lake Lillesjön that received a one-time dose of 141 g $N \cdot m^{-2}$ (Ripl 1976), and also lower than the 24 g $N \cdot m^{-2}$ added to White Lough, N. Ireland (Foy 1986).

The effect of relatively low nitrate addition was illustrated by the 1995 treatment where a concentra-

Table 2.–Mean concentrations (July-August, mg $\cdot L^{-1}$) at 5.5 and 6.5 m depth at the centre station.

| | $NO_3$-N | | DIP | | TP | | $NH_4$-N | | Fe | |
|---|---|---|---|---|---|---|---|---|---|---|
| | 5.5 | 6.5 | 5.5 | 6.5 | 5.5 | 6.5 | 5.5 | 6.5 | 5.5 | 6.5 |
| 1994 | <0.01 | <0.01 | 0.714 | 1.347 | 0.847 | 1.469 | 0.507 | 1.227 | 1.138 | 2.163 |
| 1995 | 0.098 | 0.708 | 0.461 | 0.427 | 0.678 | 0.734 | 1.280 | 2.267 | 1.108 | 1.543 |
| 1996 | 0.007 | 0.899 | 0.835 | 1.001 | 0.963 | 1.374 | 1.688 | 3.867 | 0.507 | 1.091 |
| 1997 | <0.01 | <0.01 | 0.898 | 1.512 | 1.030 | 2.256 | 1.117 | 1.967 | 2.420 | 4.900 |

Table 3.–Mean DIP, $Diss_{or}$-P, $Part_{in}$-P and $Part_{or}$-P as percentages of TP in the hypolimnion during July-August. Standard deviation (SD) and number of samples (n) are given in brackets.

| Year | (n) | DIP | (SD) | $Diss_{or}$-P | (SD) | $Part_{in}$-P | (SD) | $Part_{or}$-P | (SD) |
|---|---|---|---|---|---|---|---|---|---|
| 1994 | (8) | 87 | (5) | 2 | (4) | 7 | (4) | 4 | (4) |
| 1995 | (66) | 61 | (7) | 2 | (2) | 33 | (8) | 4 | (3) |
| 1996 | (44-50) | 80 | (5) | 2 | (1) | 16 | (6) | 3 | (3) |
| 1997 | (4) | 79 | (17) | 2 | (3) | 18 | (17) | 1 | (2) |

tion above approximately 0.5 mg $NO_3$-N · $L^{-1}$ enabled the sediment to adsorb P and to diminish hypolimnetic-P concentrations from about 0.8 mg DIP · $L^{-1}$ to a new equilibrium concentration of about 0.4 mg DIP · $L^{-1}$. When nitrate addition was stopped by the end of August, and nitrate concentrations decreased below 0.5 mg N · $L^{-1}$, P concentrations increased. Redox conditions in the hypolimnion or sediment surface were not measured, but a threshold is likely to exist around a redox potential of 200-300 mV or about 0.1 mg $O_2$ · $L^{-1}$, where iron (III) is reduced (Boström et al. 1982, Lampert and Sommer 1997).

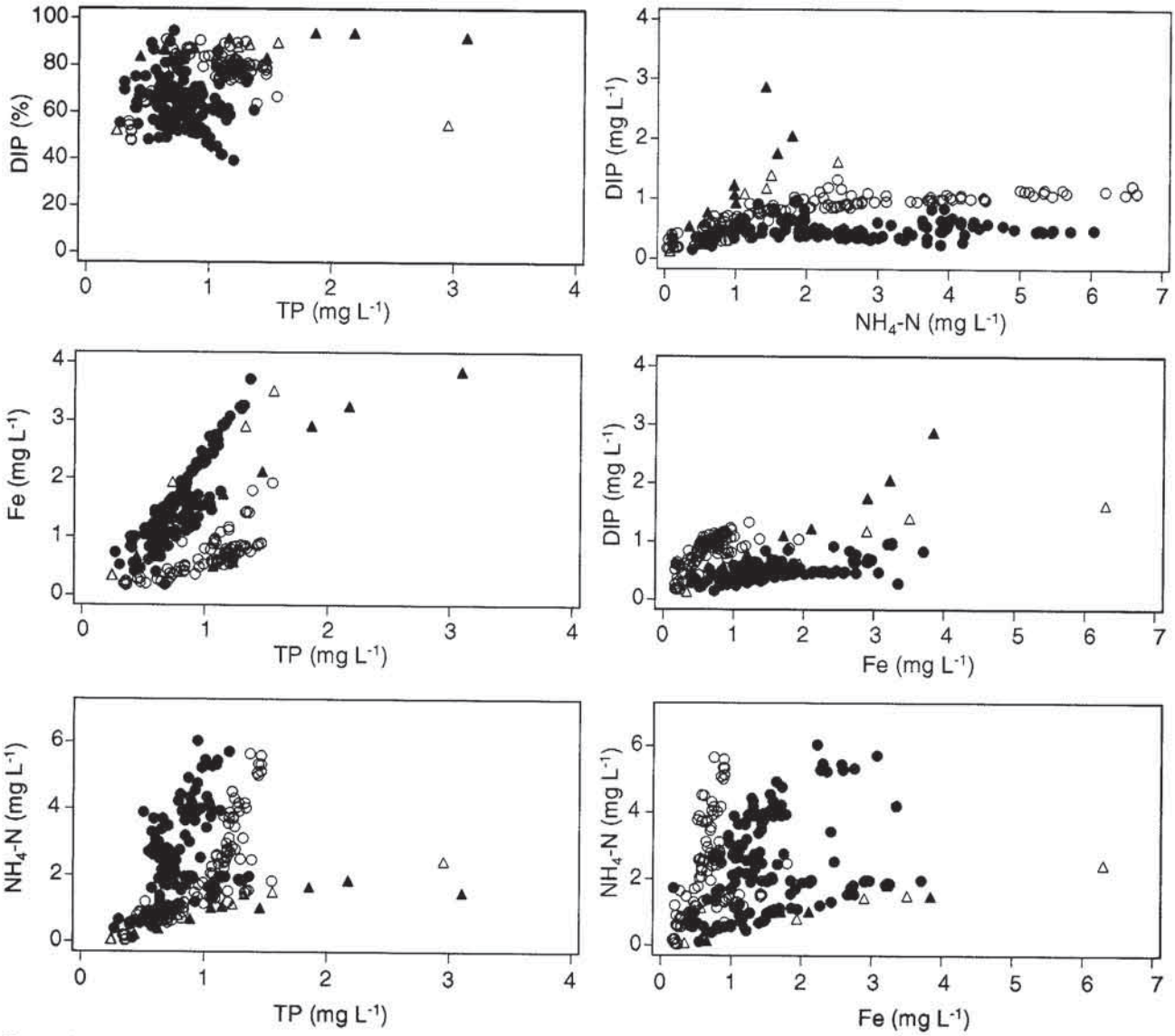



Figure 5.–Relationship among the following constituents: DIP% of TP, DIP, $NH_4$-N and Fe, in the hypolimnion (6.0 or 6.5 m depth) at the 6-7 hypolimnion stations during 1994 (▲), 1995 (•), 1996 (o) and 1997 (Δ).

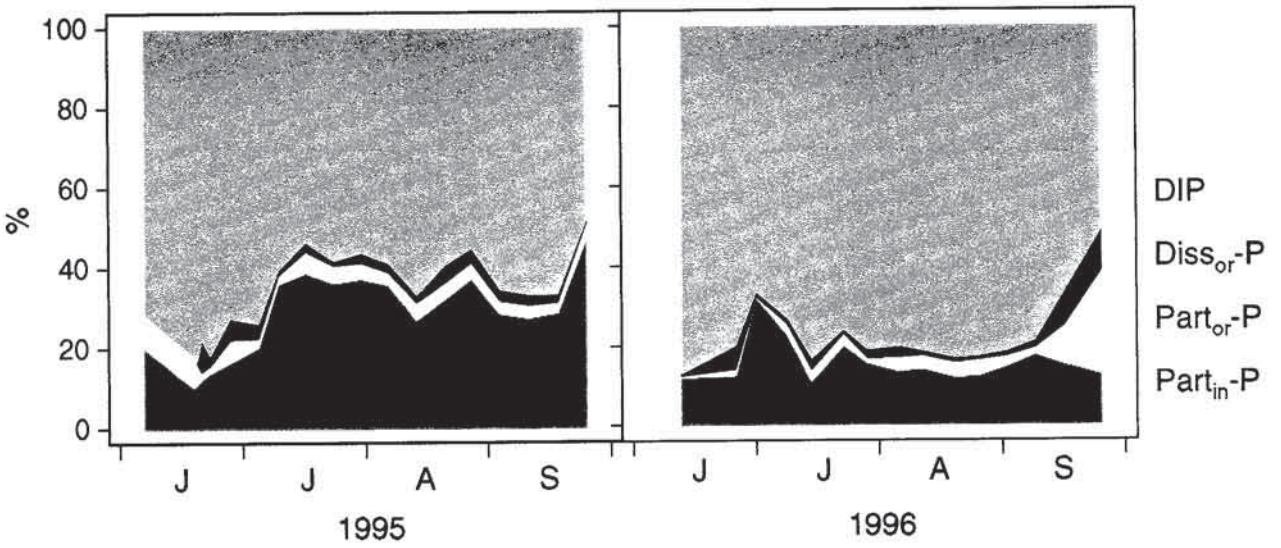



Figure 6.–Relative composition of phosphorus at the 6-7 hypolimnion stations in 1995 and 1996. From above: dissolved inorganic phosphorus (DIP), dissolved organic phosphorus (Diss$_{or}$-P), particulate organic phosphorus (Part$_{or}$-P and particulate inorganic phosphorus (Part$_{in}$-P).

The importance of even relatively low nitrate concentrations for the accumulation of phosphate in the hypolimnion has also been shown in the 29-m deep eutrophic Lake Knud, Denmark. Due to changes in primary production and nitrate loading, hypolimnetic nitrate concentrations there differed markedly between years and, simultaneously, the accumulation of phosphate changed dramatically (Andersen et al. 1980, Andersen 1994). In years with low nitrate (<0.1 mg N·L$^{-1}$) close to the sediment, DIP increased up to 0.82 and 1.49 mg P·L$^{-1}$, whereas DIP did not exceed 0.01 mg P·L$^{-1}$ in years when nitrate was present (>1.3 mg N·L$^{-1}$).

The impact of nitrate treatment in Lake Lyng differed markedly between years. The 1995 addition with dissolved nitrate seemed to counteract the accumulation of phosphate most efficiently, whereas the most significant effect of the 1996 treatment with nitrate pellets was to prevent the accumulation of hypolimnetic iron. A likely explanation of the less significant effect on phosphorus accumulation from granulated nitrate may simply be that the pellets were not completely dissolved before reaching the sediment. The sediment has a very high water content and loose structure (Table 1), enabling the pellets to sink easily into the sediment. Laboratory experiments with sediment cores taken from the hypolimnion showed that when added to the surface sediment, the pellets sank 5 to 10 cm or more into the sediment (visual observations), which may explain why hypolimnetic nitrate concentrations were lower in 1996 than in 1995. Also, P accumulation in the hypolimnion may have been higher in 1996 if P release from the sediment were assumed to depend on nitrate concentrations in the hypolimnion. The 1996 nitrate treatment was comparable to the original Riplox method where nitrate remains in the sediment and is consumed, which led to lower nitrate concentrations despite a slightly higher dose and attained high concentrations close to the sediment only. The difference in nitrate concentrations between the two treatments may have been exaggerated by the more uneven nitrate applica-

Table 4.–Maximum accumulation (kg) of DIP, TP, $NH_4$-N and Fe in the hypolimnion (5-7.6 m). % is the relative accumulation compared to the mean of values reached in the untreated years 1994 and 1997.

| | DIP | | TP | | $NH_4$-N | | Fe | |
|---|---|---|---|---|---|---|---|---|
| | kg | % | kg | % | kg | % | kg | % |
| 1994 | 23.2 | - | 25.9 | - | 17.6 | - | 34.2 | - |
| 1995 | 7.3 | 35 | 12.6 | 48 | 35.2 | 166 | 28.0 | 61 |
| 1996 | 17.0 | 82 | 20.0 | 77 | 56.2 | 265 | 14.6 | 31 |
| 1996 | 18.1 | - | 27.0 | - | 24.7 | - | 57.9 | - |

tion in 1996, as indicated by more pronounced variations observed in nitrate and phosphate concentrations than in 1995. A less efficient binding of phosphorus in the sediment can be expected with some areas when the granulated nitrate spreads over smaller areas, thus, not receiving an adequate dose to influence phosphorus release.

The considerably lower iron accumulation in the hypolimnion when nitrate was added as pellets may similarly reflect the larger proportion of nitrate introduced into the sediment with a resulting lower nitrate concentration remaining in the hypolimnion. The generally lower supply of iron to the hypolimnion and the considerably lower nitrate concentrations imply that the potential of phosphorus precipitation with iron was lower in 1996. This was supported by the observation that a decrease, which was likely a precipitation of phosphorus, was seen during part of the summer of 1995, but not in 1996. Correspondingly, the Fe:TP ratio was considerably higher in 1995 than in 1996 and the concentrations of $Part_{in}$-P were significantly higher. If iron-phosphorus complexes precipitate when nitrate concentrations are sufficiently high, this is a major mechanism controlling sediment capacity to retain phosphorus and, then, addition of dissolved rather than granulated calcium nitrate may be advantageous. Hence a higher iron release from the sediment would allow hypolimnetic phosphate precipitation and, thus, a more efficient removal of phosphate. The importance of iron in reducing phosphorus levels in the hypolimnion has also been demonstrated by hypolimnion aeration projects, which are effective only in the presence of sufficient iron (McQueen et al. 1986).

We have no data allowing an evaluation of the importance of interactions between iron sulfide precipitation and hypolimnetic phosphorus accumulation, which may be important for the iron-phosphate cycle at the oxic-anoxic boundary (Cook 1984, Ripl 1994, Murray 1995). Potentially, however, the nitrate dosing may also have influenced these processes by decreasing sulphate reduction or by forcing it deeper into the sediment and subsequently enabling a higher rate of reoxidation of ferrous iron to ferric oxides in the hypolimnion. A distinct smell of $H_2S$ from hypolimnetic water in non-treatment years, but no smell in treatment years, indicated a lower accumulation of $H_2S$ in the hypolimnion when nitrate was added.

Higher ammonia concentrations were recorded in the hypolimnion of Lake Lyng in treatment compared with non-treatment years. This is opposed to the findings from the experiments of Ripl (1976), where lower interstitial ammonia concentrations were recorded after the nitrate treatment (Ripl and Lindmark 1978), and to Foy (1986), Erlandsen et al. (1988), and Faafeng et al. (1997) who assumed that nitrate was transformed to elemental nitrogen completely through denitrification. The increase in ammonia in Lake Lyng may reflect a higher mineralization of organic matter in years with nitrate dosing, which together with very low oxygen concentrations prevented any nitrification. Dissimilatory reduction of nitrate to ammonia has also been suggested to predominate over denitrification in a reducing environment because of the extra electrons available by this process (Priscu and Downes 1987). A reduction to ammonia as high as 30 to 50% or more has been reported (Kaspar 1985, Priscu and Downes 1987, Downes 1991). The additional accumulation of $NH_4$-H of 15 to 35 kg N in treatment years compared to a total dose of 75 to 90 kg N shows that the dose level may adequately explain the differences observed between this and non-treatment years.

Nitrate addition could potentially also increase the release of phosphorus due to a higher mineralization rate and release of organic-bound phosphorus as was suggested to occur in shallow lakes at seasonally varying nitrate concentrations (Jensen and Andersen 1992). The reducing effect of nitrate addition on P concentrations in the hypolimnion, however, suggests that the effect achieved by the improved redox conditions and the increasing sediment retention capacity exceeded the effect of elevated mineralization.

# Conclusions

The differences recorded between constituents in the hypolimnion of Lake Lyng, with and without nitrate treatment, indicate that the presence of nitrate has marked effects on the accumulation of phosphate, even at low nitrate concentrations. The impact was apparently greater with dissolved rather than granulated, undissolved nitrate. Dissolved nitrate addition resulted in higher concentrations in the hypolimnion, apparently enabling phosphorus precipitation, whereas the pellet form of added nitrate tended to sink into the sediment. The advantage of low dose treatment is that no expensive equipment is required and that control and adjustment of the treatment and its impact on the ecosystem is easier. The positive results obtained from the Lake Lyng experiments suggest that nitrate addition to the hypolimnion of lakes suffering from internal phosphorus loading is a restoration method that deserves more attention and further experimental investigation.

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# Lake restoration in Denmark

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## Abstract

Lake restoration in Denmark has involved the use of several different restoration techniques, all aiming to improve lake water quality and establishing clear-water conditions. The most frequently used method, now used in more than 20 lakes, is the reduction of zooplanktivorous and benthivorous fish (especially roach (*Rutilus rutilus*) and bream (*Abramis brama*)) with the objective of improving the growth conditions for piscivores, large-sized zooplankton species, benthic algae and submerged macrophytes. Piscivore stocking (mainly *Esox lucius* (pike)), aiming especially at reducing the abundance of young-of-the-year fish, has been used in more than 10 lakes and frequently as a supplement to fish removal. Hypolimnetic oxidation, with oxygen and nitrate, has been undertaken in a few stratified lakes and sediment dredging, with the purpose of diminishing the internal phosphorus loading, has been experimented with in one large, shallow lake. Submerged macrophyte implantation has been conducted in some of the biomanipulated lakes to increase macrophyte abundance and distribution. Overall, the results from lake restoration projects, in the mainly shallow Danish lakes, show that external nutrient loading must be reduced to a level below 0.05–0.1 mg P $L^{-1}$ under equilibrium conditions to gain permanent effects on lake water quality. By using fish removal, at least 80% of the fish stock should be removed over a period of not more than 1–2 years to obtain a substantial effect on lower trophic levels and to avoid regrowth of the remaining fish stock. Stocking of piscivores requires high densities (>0.1 individuals $m^{-2}$) if an impact on the plankton level is to be obtained and stocking should be repeated yearly until a stable clear-water state is reached. The experiments with hypolimnetic oxygenation and sediment dredging confirm that internal phosphorus loading can be reduced. Experience from macrophyte implantation experiments indicates that protection against grazing by herbivorous waterfowl may be useful in the early phase of recolonization.

## Key words

**biomanipulation, hypolimnetic oxidation, lake restosation, macrophyte implantation, sediment removal.**

## INTRODUCTION

During the past 50 years, numerous lake restoration methods have been developed and tested all over the world (Born 1979; Cook *et al.* 1993; Phillips *et al.* 1999). The purpose has most frequently been to combat eutrophication, which has led to a high abundance of phytoplankton, turbid water and an overall deterioration of lake water quality and biological diversity.

Lakes that do not respond to a reduction in external nutrient loading, even when nutrient loading has been reduced to a level so low that an improvement in water quality should be observable, have become special subjects of study. The resilience to change may be chemically induced through the release of phosphorus from a pool accumulated in the sediment during the period of high loading (Marsden 1989; Phillips *et al.* 1994; Søndergaard *et al.*, in press, 1999). It may also be biologically induced and a result of a biological structure established during the period of high nutrient loading; typically, a fish community dominated by zooplanktivorous and benthivorous species or the disappearance of submerged macrophytes, which are both factors that favour the turbid state (Benndorf 1990; Jeppesen *et al.* 1990; Lauridsen *et al.* 1993). The potential of using lake restoration to establish clear-water conditions has recently been encouraged by the theory of alternative stable states in shallow lakes. The theory, which suggests that a lake may alternate between a turbid and a clear-water state within a given nutrient level (Scheffer *et al.* 1993; Scheffer & Jeppesen 1998; Bachmann *et al.* 1999), has given inspiration to and induced several management-orientated lake restoration projects with the purpose of shifting the lake from the turbid to the clear-water state.

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In Denmark, different types of lake restoration projects have been undertaken and in the present study, we give a survey of the methods applied and the results obtained so far. We also attempt to draw some general conclusions based on Danish experiments, including recommendations on the conditions that need to be fulfilled prior to and during an ongoing restoration project. As most restoration projects have been undertaken during the past 5–10 years, long-term effects and stability have not yet been fully elucidated. Furthermore, the Danish experiences are primarily based on shallow lakes.

## METHODS

### Danish lakes

The mean water depth is lower than 1.6 m in half of the Danish lakes whereas it is higher than 5 m in only 10% (Fig. 1). Most lakes are nutrient-rich because of the high sewage input from urban and cultivated areas. Even though great efforts have been made to reduce nutrient loading during the past two decades, not the least from sewage plants (Jeppesen *et al.* 1999), most lakes are still eutrophic as 50% of them have a summer average of total phosphorus above 0.15 mg P $L^{-1}$ (Fig. 1). The average summer Secchi depth is lower than 0.85 m in half of the lakes and only 13% of the lakes have a summer Secchi depth above 2 m.

Compared with unaffected lakes, the high nutrient supply has led to various changes in biological structure (Jeppesen *et al.* 1999, 2000). The fish community has changed from having a high abundance of predatory fish (generally perch (*Perca fluviatilis*) and pike (*Esox lucius*)) to almost complete dominance by zooplanktivorous species. In particular, roach (*Rutilus rutilus*) and bream (*Abramis brama*) have become dominant and generally constitute approximately 80% or more of the fish stock biomass in nutrient-rich lakes. The zooplankton is dominated by small cladocerans (*Bosmina* etc.) and cyclopoid copepods, while the number of large-sized cladocerans and more efficient phytoplankton grazers (especially *Daphnia*) has declined. As the phytoplankton biomass increases concurrently with increasing nutrient levels, the zooplankton : phytoplankton ratio biomass decreases and the zooplankton is no longer capable of controlling the abundance of phytoplankton. Finally, the enhanced turbidity leads to a significant decline in the abundance or even complete disappearance of submerged macrophytes.

### Sampling and analysis

Sampling procedures were generally conducted according to the guidelines of the Nation-wide Monitoring Programme (Kronvang *et al.* 1993); that is, lake water was usually sampled twice a month in summer (1 May to 1 October) and monthly during winter. The programme included sampling of water chemistry (nutrients, chlorophyll *a*, turbidity), phytoplankton (species, biomass) and zooplankton (species, biomass). The composition and relative abundance of fish were investigated by using standardized fishing methods and multiple mesh-sized (6.25–75 mm) gill nets. Submerged macrophyte abundance was determined in August when biomass was at a maximum. A description of the sampling programme can be found in Jeppesen *et al.* (2000).

### Restoration measures: Aims and methods

Six different types of restoration measures have been initiated in Danish lakes (Table 1). In all cases, the main objective has been to increase water transparency, either by

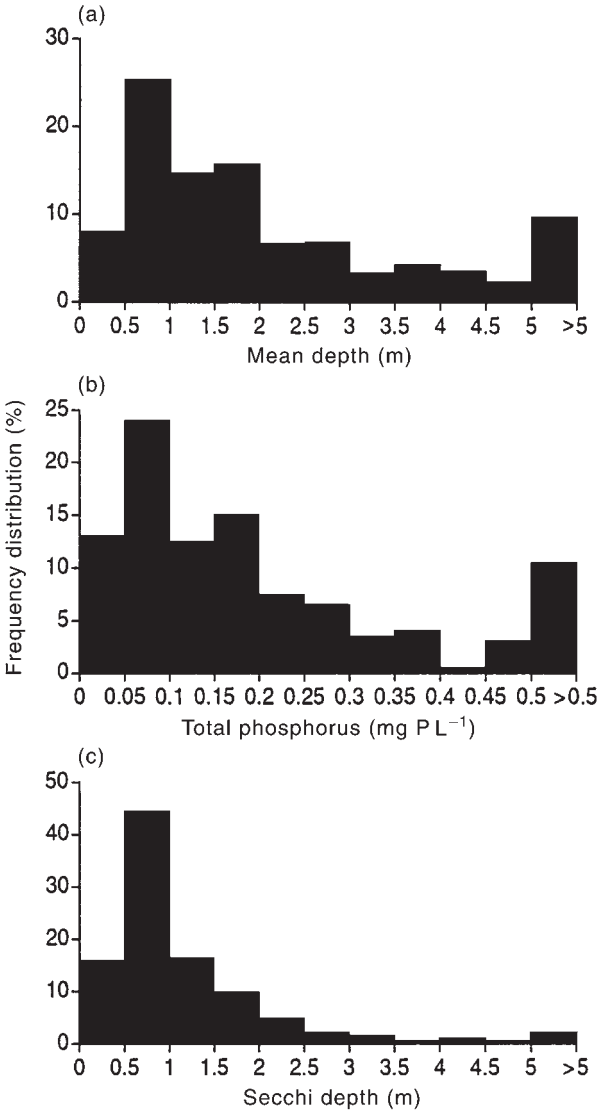



**Fig. 1.** Frequency distribution (%) of the (a) mean depth of Danish lakes ($n$ = 500), (b) total phosphorus concentration ($n$ = 200) and (c) Secchi depth ($n$ = 180).

limiting the internal loading of phosphorus from the lake sediment or by stimulating the grazing food chain by increasing the grazing pressure from zooplankton on phytoplankton.

Biomanipulation via fish stock manipulation has been undertaken in more than 20 Danish lakes and is thus the most frequently applied method. The manipulations have involved either the removal of zooplanktivorous fish (mainly roach and bream) or stocking of predatory fish (generally pike or, less frequently, perch), or both. The purpose of fish manipulation was to reduce the fish predation pressure on zooplankton and thus increase the growth potential of large-sized zooplankton and thereby its ability to limit the abundance of phytoplankton and to promote increased abundance of predatory fish (Jeppesen *et al.* 1990; Søndergaard *et al.* 1990). Selective removal of zooplanktivorous fish has usually been carried out by using pound nets or trawling, but fish traps, gill nets and electro-fishing have also been used. The extent of fish removal in the different lakes has varied significantly, from 10 to 80 g $m^{-2}$ and between 5 and 80% of the estimated total fish biomass. The stocking of predatory fish, the main purpose of which was to affect the recruitment and survival of YOY (young-of-the-year) planktivorous fish, has involved stocking of pike fingerlings in spring along the littoral zone (Berg *et al.* 1997; Søndergaard *et al.* 1997). Also, the stocking of pike

**Table 1.** Restoration measures used in Danish lakes > $5 \times 10^4$ $m^2$ during the past 15 years

| Restoration measures | No. restoration projects | Lake size, mean depth and phosphorus concentration | Main objectives of the restoration |
|---|---|---|---|
| Fish removal | 20–30 | 10–850 $\times$ $10^4$ $m^2$<br>1.1–4.3 m<br>0.08–0.70 mg P $L^{-1}$ | To reduce the number of zooplanktivorous and benthivorous fish in order to improve the conditions for large-sized zooplankton and piscivores.<br>To improve water clarity, enhance growth of submerged macrophytes, benthic algae and the abundance of benthic invertebrates. |
| Piscivore stocking | 10–20 | 10–850 $\times$ $10^4$ $m^2$<br>1.2–3.5 m<br>0.08–0.27 mg P $L^{-1}$ | To reduce the number of zooplanktivorous and benthivorous fish in order to improve the potentials of large-sized zooplankton<br>To improve water clarity, enhance growth of submerged macrophytes, benthic algae and the abundance of benthic invertebrates. |
| Macrophyte implantation | 5 | 13–150 $\times$ $10^4$ $m^2$<br>0.8–2.6 m<br>0.1–0.5 mg P $L^{-1}$ | To increase the dispersal potential and abundance of submerged macrophytes in order to stabilize the clear-water stage.<br>To enhance the day-time refuge for large-bodied zooplankton. |
| Sediment dredging | 1 | 150 $\times$ $10^4$ $m^2$<br>0.8 m<br>0.9 mg P $L^{-1}$ | To reduce the internal loading of phosphorus by removing phosphorus-rich sediment. |
| Hypolimnetic aeration | 2 | 8–340 $\times$ $10^4$ $m^2$<br>5.0–13.1 m<br>0.1–0.5 mg $\times$ $10^4$ P $L^{-1}$ | To reduce the internal loading of phosphorus by improving the redox conditions in the hypolimnion and surface sediment. |
| Hypolimnetic nitrate addition | 1 | 10 $\times$ $10^4$ $m^2$<br>2.4 m<br>0.5 mg P $L^{-1}$ | To reduce the internal loading of phosphorus by improving the redox conditions in the hypolimnion and surface sediment. |

has varied greatly from 0.005 to 0.36 individuals $m^{-2}$ $year^{-1}$.

Macrophyte implantation has been used in five lakes with the purpose of increasing the abundance and distribution potential of submerged macrophytes. Despite the shallowness, the natural stock of macrophytes has often disappeared because of the high nutrient loading and turbid water (Jeppesen *et al.* 1999). Re-establishment after improved transparency may be slow, possibly because of a low or complete lack of seed banks and lack of nearby locations from where plants may spread. Also, waterfowl grazing may be responsible for slow re-establisment (Søndergaard *et al.* 1998). Based on the assumption that plants are capable of long-term and long-range spreading, implantation has generally been used as a supplement to other restoration methods and has so far often been limited to a small part of the total lake area. The most comprehensive experiment was made in Lake Engelsholm, where 900 $m^2$ of macrophytes were established inside enclosures. Native and eutrophication-tolerant species, such as *Potamogeton pectinatus* and *Potamogeton crispus,* were usually used.

Large-scale sediment removal has only been undertaken in shallow Lake Brabrand, near the city of Aarhus (lake area $150 \times 10^4$ $m^2$, average depth 0.8 m). The objective was to reduce the sediment release of phosphorus by removing the upper nutrient-rich sediment layer because mass balance measurements showed that lake internal loading was high (Jørgensen 1998). Simultaneously, the sediment removal was aimed at preventing the filling in of this recreationally important lake, that has an annual sediment increase of about 1 cm. Prior to, or concurrently with, sediment removal, phosphorus removal was introduced at all major sewage plants in the lake catchment. Sediment was removed by using a dredge (Mudcat®, Ellicot Machine Corp), which was pulled in tracks over the parts of the lake from which sediment was to be removed. Thereafter, the nutrient-rich sediment was pumped to depositing basins near the lake. In total, approximately 500 000 $m^3$ mud was removed over a 7-year period.

Hypolimnetic oxidation has been undertaken in two Danish lakes. The most comprehensive restoration so far was made in deep, stratified, Lake Hald in central Jutland (lake area $340 \times 10^4$ $m^2$, average depth 13 m, maximum depth 31 m). For 12 years, pure oxygen was pumped into the hypolimnion in summer when the lake was stratified (Rasmussen 1998). Oxygen was dispersed in the hypolimnion via eight diffusers (each having approximately 50 000 holes at 1 mm in diameter) placed at four different locations on the lake bottom. The purpose was to increase the sediment's phosphorus-binding capacity via oxidation of iron and to enhance survival of animals (particularly the endangered chironomid larvae, *Chironomus anthracinus*) living in the profundal zone. On average, $210 \times 10^3$ kg $O_2$ were added yearly, corresponding to 140 mg $m^{-3}$ $day^{-1}$ during the period of stratification.

Hypolimnetic addition of calcium nitrate has been used in one lake only. In the $10 \times 10^4$ $m^2$ large and summer-stratified Lake Lyng, situated near the town of Silkeborg (mean depth 2.4 m and maximum depth 7.6 m), calcium nitrate ($Ca(NO_3)_2$) was added to the hypolimnion during two summer periods (Søndergaard *et al.* unpubl. data, 2000). The dosages were 8–10 g N $m^{-2}$ and the nitrate was added either in a dissolved or granulated form at 5-m depths in areas greater than 5 m. Dosing was undertaken approximately once a week from late June until late August. The purpose was to increase the capability of the sediment to retain phosphorus under the anaerobic conditions that developed shortly after the onset of stratification via the oxidation effects of nitrate, on iron in particular (Ripl 1978).

## RESULTS AND DISCUSSION

In spite of a significant variation in lake morphometry, nutrient loading, intensity and scale of intervention (Table 1), some general patterns seem to emerge from Danish restoration projects, especially regarding fish manipulation on which comprehensive sets of data exist. The data obtained primarily cover a brief post-restoration period and do not allow an adequate validation of long-term effects.

The impact of fish manipulation on both the fish community and the remaining trophic levels depends highly on the scope of the manipulation (Tables 2,3). No effects or very few were observed, while large-scale and intensive manipulation often had marked effects on several biological and chemical variables used as indicators of improved water quality (Table 2). The results suggest that at least 80% of the zooplanktivorous fish stock should be removed, if an impact on trophic levels other than fish is to be obtained. This observation supports the results obtained from several other international experiments (Perrow *et al.* 1997; Hansson *et al.* 1998; Meijer *et al.*, in press, 1999). If the effect is to cascade to lower trophic levels, the critical fish biomass seems to be approximately 10 g $m^{-2}$ in eutrophic lakes, a level also recorded elsewhere by Seda and Kubecka (1997). However, the removal of large amounts of fish does not necessarily mean that this manipulation has effectively improved the water quality. The time scale is an important factor. In the case of long-term, but less intensive, interventions, the remaining fish will largely compensate for the removal via increased growth and reproduction. Thus, during the restoration of a $270 \times 10^4$ $m^2$ large lake conducted over a 5-year period, twice as many fish were removed as estimated

**Table 2.** Effects on the fish stock after fish removal

| Intensive fish removal over a short-term period | Long-term but low intensity fish removal | Modest fish removal |
|---|---|---|
| The growth rate of the remaining fish, especially perch, increases following improved water transparency. | Gradual reduction of the biomass proportion of bream. The biomass, however, remains high compared with intensively fished lakes. | Poor or no effect |
| Perch often becomes the dominant predatory fish, whereas the response of pike remains unclear. | Occasionally increased percentages of piscivores, especially caused by increased biomass of perch. Usually, however, the share of piscivores remains below 20%. | |
| The proportion of predatory fish increases significantly during the first 1–2 years following fish removal because of the increased abundance of perch. | | |

**Table 3.** Effects on different trophic levels recorded after intensive fish removal

| Parameter | Effects |
|---|---|
| Zooplankton | Increased abundance of large-sized species |
| | Increased grazing pressure on phytoplankton |
| Phytoplankton | Reduced abundance |
| Invertebrates | Increased abundance |
| | Increased food supply for perch |
| Submerged macrophytes | Gradually higher distribution depending on i.a. depth conditions, waterfowl grazing and seed bank. |
| Waterfowl | Increased abundance – including species feeding on submerged macrophytes (e.g. mute swan (*Cygnus olor*) and coot (*Fulica atra*)) |
| Nutrient content | Declining concentrations caused by increased retention |
| Secchi depth | Increased Secchi depth |

prior to the intervention, without it having any apparent effects on the fish stock biomass (Mæhl 1998). Therefore, fish removal should preferably not last much longer than 1–2 years (Hansson *et al.* 1998). In northern temperate lakes, it is particularly important to reduce the abundance of bream, as bream, as well as reducing the abundance of zooplankton, also markedly reduces the number of benthic invertebrates (Andersson *et al.* 1978; Brönmark *et al.* 1997). The loss of benthic invertebrates may have a serious impact on perch, whose growth largely depends on and is positively correlated with the abundance of macroinvertebrates (Persson 1983; Diehl 1993). Thus, at high bream abundance, a competitive bottleneck at the macroinvertebrate feeding stage occurs (Persson & Greenberg 1990), preventing perch from reaching the predatory stage. By removing bream, the growth of perch increases. This has been illustrated by Danish experiments where perch, in only 2 years, reached the size usually obtained over a 5-year period in a typical Danish lake (Müller & Jensen unpubl. obs., 1998). Finally, when searching for food in the sediment, roach and bream may also have a direct, negative influence on suspended matter and lake turbidity because of the resuspension of sediment (Breukelaar *et al.* 1994; Tátrai *et al.* 1997) or enhanced nutrient release (Brabrand *et al.* 1990; Havens 1991).

Experience regarding the stocking of pike fry is less comprehensive and in most cases only relatively low proportions have been stocked. It appears though, that if stocked in large numbers, cascading effects on lower trophic levels can be achieved, primarily in the year of stocking. Experiments from Lake Lyng showed that pike abundance did not depend on the number of pike stocked the previous year (Berg *et al.* 1997; Søndergaard *et al.* 1997). The reason is probably, as also shown in the Netherlands (Grimm & Backx 1990), that the size of the pike stock primarily depends on the number of habitats, especially that of the littoral and macrophyte-covered zones. Restoration by pike stocking should therefore primarily be considered in lakes

where a few years of stocking and improved transparency can lead to a shift in the biological structure towards one that maintains a clear-water state (e.g. colonization of submerged macrophytes). Likewise, the results from Lake Lyng showed that stocking densities must be much higher than the natural stock if any impact on other trophic levels is to be achieved (Søndergaard *et al.* 1997). The results observed in Denmark and elsewhere (Meijer *et al.* 1995; Prejs *et al.* 1997) indicate that stocking of at least 0.1 individuals $m^{-2}$ $year^{-1}$ is required, but more information is needed to optimize stocking strategies (Skov & Berg unpubl. data, 1999). Also, the potential of using piscivores seems highest if used in association with fish removal to impede massive growth of YOY fish (Perrow *et al.* 1997; Hansson *et al.* 1998).

Lake water nutrient concentrations also seem susceptible to fish manipulation. Often, both in-lake nitrogen and phosphorus decrease markedly and retention increases if clear-water conditions are obtained (Søndergaard *et al.* 1990; Jeppesen *et al.* 1998), which has a positive effect on the water quality of downstream lakes or fjords. The reasons for the cause in higher retention have not yet been identified, but various factors may be involved (Wright & Shapiro 1984; Jeppesen *et al.* 1998), including improved light conditions and the fact that increased benthic primary production exceedingly impedes phosphorus release from the sediment (Hansson 1992; Van Luijn *et al.* 1995).

When evaluating the effects of macrophyte implantation, it must be taken into consideration that the implantation experiments usually only covered a very modest part of the total lake area. So far, no evident impact of implantation has been found in the study of lakes, but the results of various other Danish investigations suggest that plant-eating waterfowl may delay and/or impede macrophyte dispersal. In a number of exclosure experiments, Lauridsen *et al.* (1993, 1994) and Søndergaard *et al.* (1996, 1998) showed that the growth of unprotected plants was much lower than plants protected against waterfowl grazing. Lake Engelsholm, provided evidence that even when waterfowl densities are relatively low, macrophytes may be subjected to a considerable grazing pressure. There is thus ample reason to indicate that plant-eating birds may delay the re-establishment of submerged macrophytes. Furthermore, even when macrophytes are established, herbivory by birds may enhance the probability of a transition back to the phytoplankton-dominated stage by favouring inedible plant species with a shorter growing season and lower stabilizing effects on the clear-water stage (Janse *et al.* 1998). The impact of macrophytes on a stabilization of the clear-water state seems considerable, especially when the plant volume infested with submerged macrophytes exceeds approximately 20% (Schriver *et al.* 1995; Meijer *et al.* in press, 1999). However, in a study on diurnal horizontal migration of zooplankton in a $21 \times 10^4$ $m^2$ shallow lake, Lauridsen *et al.* (1996) estimated that the establishment of a 3% coverage with 2 m diameter patches of dense *Potamogeton pectinatus* would be sufficient to double the density of *Ceriodaphnia* spp. and *Bosmina longirostris* in open water at night, which would subsequently have a significant impact on zooplankton grazing on phytoplankton.

Danish experience with physicochemical methods is limited. The sediment removal in Lake Brabrand led to an increase in water depth and a reduction of phosphorus release from the lake bottom. Although the lake still suffers from internal loading, both the duration and extent of the internal phosphorus loading seemed to decline significantly after the intervention, compared with other lakes where the external loading has been reduced (Jørgensen 1998). The lake remains, however, in the turbid state because the external nutrient loading has not been reduced sufficiently. Oxidation of the hypolimnion in Lake Hald has had a marked effect on the oxidation level and internal phosphorus loading and together with a reduction of external loading occurring simultaneously with the oxidation, this has led to higher transparency. Recent results, however, indicate that further oxidation beyond the now finished 12-year period is necessary to avoid increased internal loading (K. Rasmussen, pers. comm., 1999). Hypolimnetic nitrate addition in Lake Lyng showed that it is possible to limit the internal release and accumulation of phosphorus in the hypolimnion, even when using relatively low doses of nitrate (Søndergaard *et al.* unpubl. data, 2000), but if permanent effects are to be obtained, the treatment should probably be continued.

## CONCLUSIONS

An important prerequisite for a successful and stable restoration intervention in northern-temperate shallow lakes seems to be that lake nutrient loading should be brought to a level of 0.05–0.1 mg P $L^{-1}$ under equilibrium conditions, as previously concluded (Jeppesen *et al.* 1990, 1999). The probability of a successful intervention is expected to increase with declining nutrient levels. Clear-water conditions may be obtained even at high nutrient concentrations, but the risk of a return to the turbid state is high if the intervention is not continued.

By using biomanipulation in northern-temperate lakes, approximately 80% of the prey fish should be removed over a 1–2-year period, if increased growth of the remaining fish stock is prevented and if significant effects are to be obtained. The stock of zooplanktivorous fish needs to be reduced below approximately 10 g $m^{-2}$. Stocking of pike fry

to combat the YOY of planktivorous fish must be extensive ($0.1\ m^{-2}$) if effects cascading to lower trophic levels are to be obtained. Also, pike fry stocking is only effective the year in which it is made and must therefore be repeated until stabilization is achieved. Generally, the long-term stability of biomanipulated restoration is still very poorly elucidated, locally and internationally. One of the future challenges within this field is to determine the stability of the clear-water state, taking into account the often significant interannual variations in fish recruitment and growth of submerged macrophytes mediated by, for instance, variations in climate.

Experience with implantation of submerged macrophytes indicates that protection against waterfowl grazing in the early phase of implantation can be useful. Danish experience with large-scale sediment removal and hypolimnetic oxygenation is limited. The results seem to confirm other findings showing that it is possible to reduce both the duration and size of internal nutrient loading, but that hypolimnetic oxygenation needs to be conducted for many years in order to gain permanent effects.

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# Internal phosphorus loading in shallow Danish lakes

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*Key words:* shallow lakes, phosphorus retention, internal loading, sediment

**Abstract**

High phosphorus concentrations due to internal loading from the sediment with a strongly negative impact on lake water quality, is often seen in shallow lakes after a reduction of external loading. To analyse the nature of internal loading we studied 1. the seasonal phosphorus concentrations of 265 Danish shallow, mainly eutrophic lakes; 2. seasonal phosphorus mass balances and retention for eight years in 16 eutrophic lakes, and 3. phosphorus mass balances and changing sediment phosphorus profiles for 15 years in one hypertrophic lake. Lake water, inlets and outlets were routinely sampled 10–26 times annually. Total phosphorus (TP) concentrations during summer were two–four times higher than winter values in lakes with a mean summer total phosphorus concentration ($TP_{sum}$ ) above 0.2 mg P $l^{-1}$. Annual phosphorus retention decreased with increasing $TP_{sum}$ and was lower than predicted from the Vollenweider model, particularly in lakes with $TP_{sum}$ above 0.2 mg P $l^{-1}$. The seasonal phosphorus retention in lakes with $TP_{sum}$ below 0.1 mg P $l^{-1}$ was positive during the whole season, except July and August when mean retention ranged from −10 to −30% of inlet loading. In lakes with $TP_{sum}$ above 0.1 mg P $l^{-1}$, the retention was positive during winter, but negative from April to September. The negative retention was most pronounced in lakes with the highest $TP_{sum}$, particularly in May and July when mean retention ranged from −50 to −68% in lakes with $TP_{sum}$ above 0.2 mg P $l^{-1}$. The retention was generally less negative in June, when a clearwater phase typically occurs and close to 0 also in lakes with a high $TP_{sum}$. Mass balances from the hypertrophic lake have now shown a 15-yr net annual negative retention following reduced external loading. Sediment profiles suggest phosphorus release from depths down to 25 cm and that net internal phosphorus loading may persist for another 15 yrs. It is concluded that internal loading of shallow eutrophic lakes may have a considerable and persistent impact on summer TP after reduced external loading.

## Introduction

Although some lakes may respond fast to changes in external phosphorus loading (Sas, 1989), measures introduced to reduce external loading have frequently not as expected led to a decrease in lake water phosphorus concentrations (Marsden 1989; Jeppesen et al., 1991; Van der Molen & Boers, 1994). The reason is internal loading of phosphorus released from a sediment pool which was created when external loading was high. The intensity and duration of internal loading may have a very significant impact on lake water phosphorus concentrations and subsequently on lake water quality (Jeppesen et al., 1991; Phillips et al., 1994).

Data on lake water seasonal TP, sediment concentrations of various phosphorus fractions and phosphate gradients in interstitial water as well as laboratory release experiments have been used to describe and evaluate the possible impact of internal loading following reduced external loading (Shaw & Prepas, 1990; Van der Molen, 1991; Jensen et al., 1992; Ignatieva 1996; Istvanovics & Petterson, 1998). Mass balance calculations determined by total input and output measurements are another and probably the most accurate, but usually very costly, approach if precise determination is required (Dillon & Evans, 1993). Due to inadequate knowledge about the mechanisms behind internal loading in shallow lakes (Phillips et al., 1994; Welch & Cooke, 1995), it has so far been difficult to establish general relationships between simple lake or sediment characteristics and the intensity and duration of internal loading. Consequently, models predicting lake water concentrations as a tool in lake

*Table 1.* Characteristics of the 265 shallow lakes

| | 25% fractile | Median | 75% fractile | Mean |
|---|---|---|---|---|
| Area, ha | 17 | 40 | 137 | 247 |
| Mean depth, m | 1.2 | 2.1 | 3.2 | 2.3 |
| Mean summer Secchi depth, m | 0.75 | 1.2 | 2.0 | 1.6 |
| Mean summer TP, mg P $l^{-1}$ | 0.15 | 0.30 | 0.58 | 0.47 |

*Table 2.* Characteristics of the 16 shallow lakes

| | Minimum | Median | Mean | Maximum |
|---|---|---|---|---|
| Area, ha | 5 | 34 | 91 | 662 |
| Mean depth | 0.9 | 1.9 | 2.5 | 9.9 |
| Water retention time, days | 7 | 30 | 70 | 266 |
| Mean summer TP, mg P $l^{-1}$ | 0.086 | 0.286 | 0.322 | 0.991 |
| Mean summer Secchi depth, m | 0.4 | 0.6 | 0.8 | 2.0 |

management are less valid for lakes suffering from internal loading.

In this study we analyse the internal phosphorus loading of shallow Danish lakes using:

1. seasonal variations in lake water phosphorus concentrations from 265 lakes;
2. 8 years of monthly mass balance calculations from 16 lakes; and
3. 15 years of mass balance measurements from hypertrophic Lake Søbygaard combined with contemporaneous measurements of sediment phosphorus profiles.

Our aim was to evaluate the seasonal dynamics of phosphorus concentrations and retention in shallow lakes along a phosphorus gradient in order to gain new insight into the nature of internal loading.

## Methods and study areas

### *Seasonal lake water phosphorus concentrations in 265 shallow lakes*

The lakes included in this analysis were mainly eutrophic, shallow and relatively small (Table 1). The lakes were sampled at least 10 times annually. Sampling was conducted from 1985 and onwards, but each lake is only represented once. If data were available from more than one year, the most comprehensive and most recent data set was used. Only epilimnic (surface) samples were included. Lake water total phosphorus (TP) was analyzed as molybdate reactive phosphorus following persulphate digestion according to Koroleff (1970). Mean summer total phosphorus ($TP_{sum}$) was calculated as mean values from 1 May to 1 October.

### *Mass balances in 16 shallow lakes for 8 years*

The 16 lakes used in the mass balance analyses are all included in the Danish Nationwide Monitoring Programme (Kronvang et al., 1993). The lakes are relatively small, turbid, eutrophic and retention time is short (Table 2). From 1989 to 1996, the main inlet of each lake was sampled 18–26 times annually, depending on seasonal variations in discharge, while the minor inlets were sampled less frequently, depending on their relative contribution to total loading. Outlet samples were collected twice monthly during summer and once monthly during winter, i.e. 19 times annually. TP was analysed and $TP_{sum}$ calculated as described above. From 1989 to 1996, external TP loading to four of the 16 lakes was significantly reduced ($p < 0.05$). In one of the lakes, $TP_{sum}$ was $<0.1$ mg P $l^{-1}$, in two lakes it ranged between 0.1 and 0.2 mg P $l^{-1}$ and in another lake $TP_{sum}$ was $>0.2$ mg P $l^{-1}$.

### *Lake Søbygaard*

For several decades Lake Søbygaard (area 40 ha, mean depth 1.2 m, water retention time 3–4 weeks) was heavily loaded with waste water from a nearby town (Jensen et al., 1992; Søndergaard et al., 1993; Jeppesen et al., 1998). In the 1970s and early 1980s, annual loading was about 30 g P $m^{-2}y^{-1}$. The lake, however, remained hypertrophic and in 1982, chemical treatment was implemented at the sewage treatment plant and external TP loading consequently decreased by 80–90%. The lake, however, remained hypertrophic and since 1982, mean summer TP has ranged from 0.4 to 1.0 mg P $l^{-1}$ and chlorophyll *a* between 130 and 840 $\mu$g $l^{-1}$ (Jeppesen et al., 1998).

The mass balance of Lake Søbygaard was determined as described above with minor modifications (see also Jeppesen et al., 1998).

Sediment cores were sampled in 1985, 1991 and 1998 from a central location and sectioned into 2 cm slices. Sediment from three different cores was pooled into one sample before analysis. A more detailed description is given by Søndergaard et al. (1993). Sediment total phosphorus ($TP_{sed}$) was analyzed spectrophotometrically as molybdate reactive phosphorus

after extraction of ash-free sediment with 1 M HCl (modified from Andersen, 1976). Dry weight (DW) was determined by drying at 105 °C for 24 h and loss on ignition (LOI) subsequently determined by drying to constant weight at 550 °C.

Sediment profiles were adjusted to 1985 level using a sedimentation rate of 0.6 cm $y^{-1}$ (Søndergaard et al., 1993). Volumetric phosphorus concentrations used to determine $TP_{sed}$ content per unit area were calculated using $TP_{sed}$, DW and LOI, assuming an inorganic matter density of 2.6 g $cm^{-3}$ and an organic matter density of 1.05 g $cm^{-3}$.

## Results

### *Seasonal phosphorus concentrations in 265 shallow lakes*

Seasonal TP variations depended on nutrient levels. In lakes with $TP_{sum}$ below 0.05 mg P $l^{-1}$, seasonal variation was small and summer concentrations did not differ much from winter values (Figure 1). In more eutrophic systems and particularly when $TP_{sum}$ was above 0.1 mg P $l^{-1}$, summer concentrations were significantly and typically 2–4 fold higher than winter values. The period with lake water summer concentrations more than twice as high as winter values increased from about 1 month in lakes with a $TP_{sum}$ between 0.1 and 0.2 mg P $l^{-1}$ to 4–6 months in lakes with $TP_{sum}$ above 0.2 mg P $l^{-1}$. Highest maximum summer TP relative to winter concentrations was reached in lakes with a mean summer TP ranging between 0.4 and 0.8 mg P $l^{-1}$ (Figure 2). At this TP level, maximum TP was 2.5 times higher than winter values in 50% and 4 times higher in 25% of the lakes. Mean summer TP showed a similar pattern. In lakes with $TP_{sum}$ above 0.2 mg P $l^{-1}$, half of the lakes had a mean summer phosphorus concentration which was 1.7–2.1 fold higher than winter values. Of lakes with a $TP_{sum}$ between 0.4 and 0.8 mg P $l^{-1}$, 25% had a 3.4 times higher mean summer TP.

### *Phosphorus mass balances during 8 years from 16 shallow lakes*

Annual phosphorus retention (as % of inlet) was highest in lakes with the lowest phosphorus concentrations, but decreased with increasing $TP_{sum}$ (Figure 2). More than 50% of the lakes with $TP_{sum}$ between 0.4 and 0.8 mg P $l^{-1}$ had a negative annual retention. At all TP levels, but particularly in lakes with

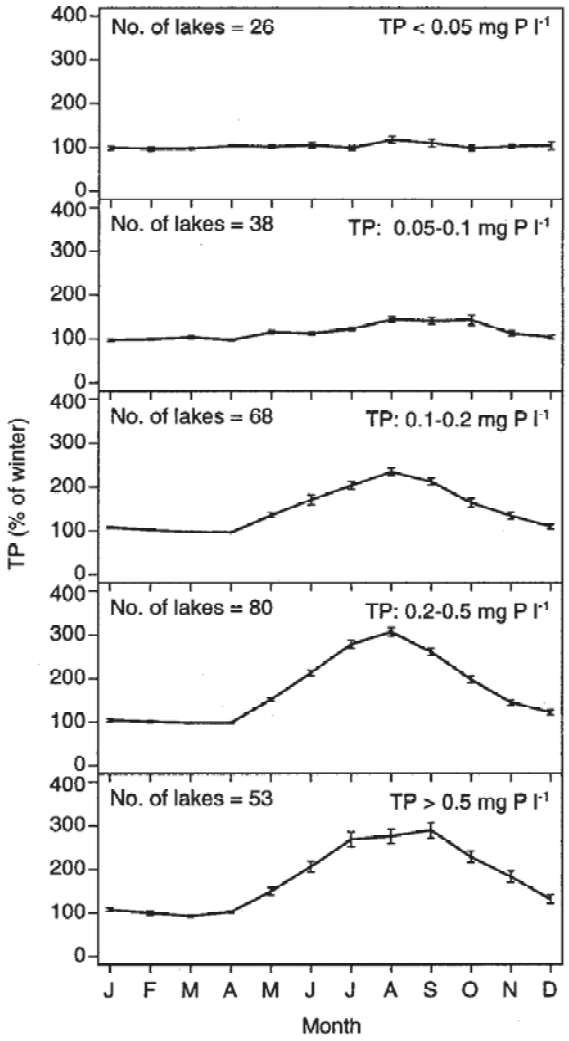



*Figure 1.* Seasonal variation in TP (monthly mean ± SD) as per cent of winter values (1 Jan. – 31 March) in different categories of $TP_{sum}$ (number of lakes = 265). Modified from Jeppesen et al. (1997).

$TP_{sum}$ above 0.2 mg P $l^{-1}$, median retention was considerably lower than predicted from the Vollenweider model ($TP_{lake} = TP_{inlet}/(1+Tw^{0.5})$), where $TP_{lake}$ is mean annual TP, $TP_{inlet}$ is the annual mean inlet TP and Tw is the hydraulic retention time (Vollenweider, 1976).

Seasonally, large differences in phosphorus retention were recorded between eutrophic and less eutrophic lakes (Figure 3). In lakes with $TP_{sum}$ below 0.1 mg P $l^{-1}$, mean phosphorus retention was positive throughout the year excepting July and August, while retention was negative from April to September in lakes with $TP_{sum}$ >0.1 mg P $l^{-1}$. Retention was

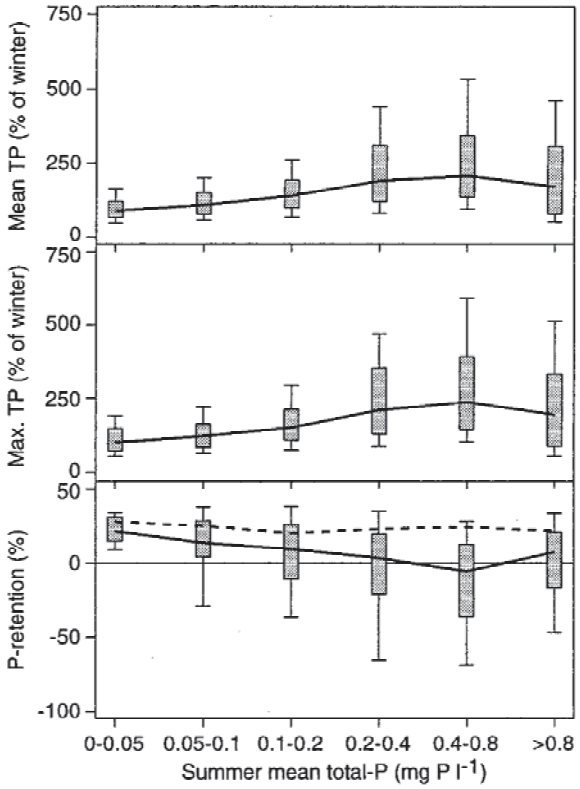



*Figure 2.* Upper: Mean summer TP as per cent of winter values (1 Jan. – 31 March) in 265 lakes. Middle: Maximum summer TP as per cent of winter values (1 Jan. – 31 March) in different categories of $TP_{sum}$ in 265 lakes. Lower: TP retention in three different categories of $TP_{sum}$ in 16 lakes during 8 years. Percentiles of 10, 25, 50 (median), 75 and 90% are shown. Median values of predicted P-retention (Vollenweider, 1976) are shown by - - - and SE by bars.

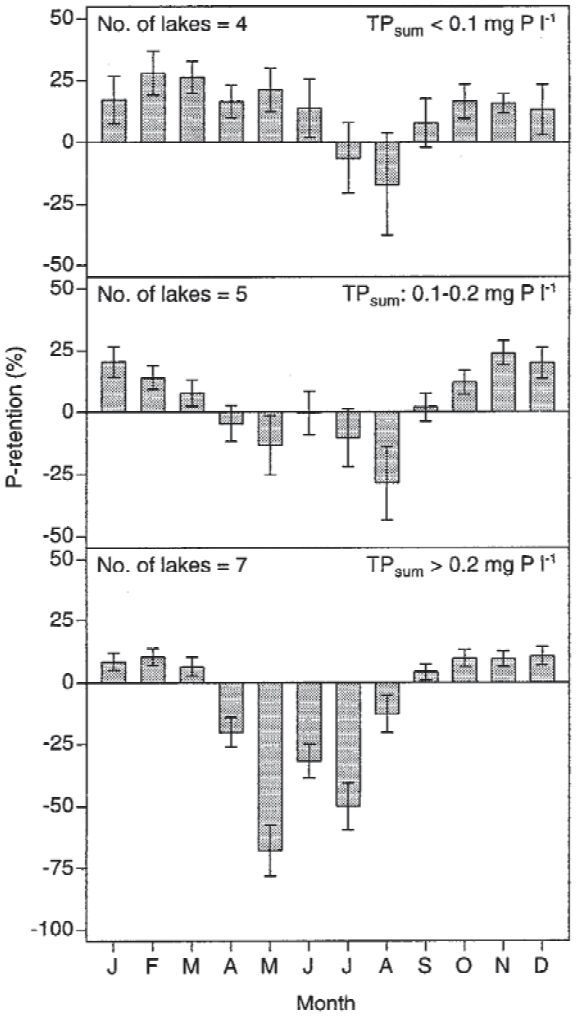



*Figure 3.* Seasonal TP-retention in three different categories of $TP_{sum}$ in 16 lakes during 8 years.

most negative in May and July (as high as 50-68% of external loading), while in June retention was often less negative, particularly in lakes with a $TP_{sum}$ between 0.1 and 0.2 mg P $l^{-1}$.

*Lake Søbygaard*

The phosphorus profile of the Lake Søbygaard sediment has changed markedly during the 13 yrs since the first profile was made in 1985 (Figure 4). In the upper 25–30 cm of the sediment $TP_{sed}$ has decreased at all depths. From 1985 to 1991, phosphorus was primarily released from the very high concentrations found at 15–20 cm depth, but from 1991 to 1998 $TP_{sed}$ decreased at all depths. At most depths down to 25–30 cm, $TP_{sed}$ has been reduced by 3–4 mg P $g^{-1}$ DW since 1985. Calculations based on comparisons of the 1985- and 1998-profiles show that a total of 57 g P $m^{-2}$ has been released from the upper 20 cm sediment (1985-level, Table 3). In the same period, mass balance measurements show a total release of approximately 40 g P $m^{-2}$.

## Discussion

As in many temperate shallow lakes (Sas 1989; Phillips et al., 1994; Welch & Cooke 1995; Ekholm et al., 1997), TP in most Danish lakes increases during summer. This increase may be due to increased inlet concentrations because waste water constitutes a larger proportion during summer at low-river discharge (Kristensen et al., 1990). However, in most cases the increase can only be attributed to increased loading

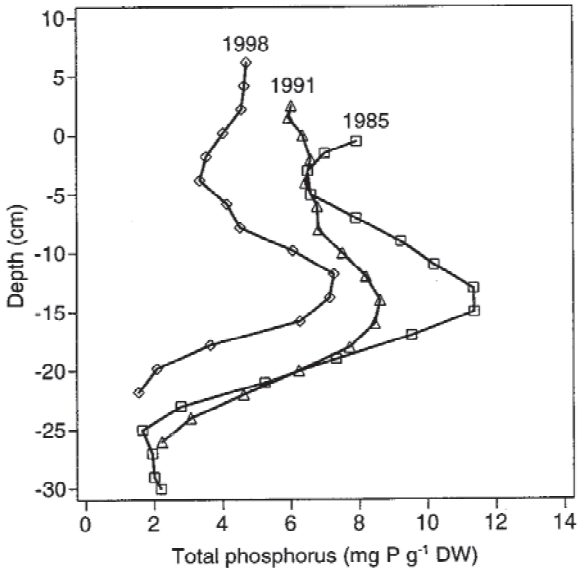



*Figure 4.* Sediment profiles of total P in Lake Søbygaard in 1985, 1991 and 1998. Sediment depth adjusted to 1985 level.

*Table 3.* Total phosphorus content in the sediment of Lake Søbygaard in 1985 and 1998. Calculation of mass balance measurements for the same period is also shown

| Sediment 1985 | depth, cm 1998 | 1985 g P $m^{-2}$ | 1998 g P $m^{-2}$ | Difference 1985–1998 g P $m^{-2}$ |
|---|---|---|---|---|
| –6 – – 8 | 0–2 | | 6.9 | + 6.9 |
| –4 – – 6 | 2–4 | | 9.2 | + 9.2 |
| –2 – – 4 | 4–6 | | 9.6 | + 9.6 |
| –2–0 | 6–8 | | 9.1 | + 9.1 |
| 0–2 | 8–10 | 9.0 | 8.5 | – 0.5 |
| 2–4 | 10–12 | 11.5 | 8.8 | – 2.7 |
| 4–6 | 12–14 | 14.2 | 11.4 | – 2.8 |
| 6–8 | 14–16 | 19.2 | 12.5 | – 6.7 |
| 8–10 | 16–18 | 25.2 | 18.7 | – 6.5 |
| 10–12 | 18–20 | 29.9 | 24.3 | – 5.6 |
| 12–14 | 20–22 | 36.2 | 24.8 | – 11.4 |
| 14–16 | 22–24 | 38.6 | 22.8 | – 15.8 |
| 16–18 | 24–26 | 33.4 | 12.0 | – 21.4 |
| 18–20 | 26–28 | 25.3 | 6.7 | – 18.6 |
| **–8–20** | **0–28** | **242.5** | **185.3** | **– 57.2** |
| Mass balance calculations | | | | **– 39** |

from the sediment. The importance of internal loading for determining lake phosphorus concentrations varies with $TP_{sum}$. The most pronounced impact was found in the most eutrophic lakes in which mean TP exceeded winter values by a factor 2–3 for several months. In these lakes, summer TP concentrations depend on internal rather than external loading.

A typical feature of the lakes was a major discrepancy between measured and calculated annual P retention, particularly in eutrophic lakes. This emphasises, as also found in Dutch lakes (Van der Molen et al., 1994), that the empirical relationship developed by Vollenweider (1976) markedly underestimates TP values for lakes in which external loading has been recently reduced and in which internal loading constitutes a considerable part of total loading.

Phosphorus retention exhibits a seasonal pattern that mimics the seasonal variation in lake water TP. During winter, retention is positive while it is negative during part of the summer. Even lakes with $TP_{sum}$ below 0.1 mg P $l^{-1}$ had a negative retention for two months (July–August), but the duration and magnitude of negative retention increase with increasing $TP_{sum}$ and last for 5 months (April-August) in the more eutrophic systems.

There may be several explanations for the seasonal variations in the capacity of the sediment to retain phosphorus and its dependency on the eutrophication level (Boström et al., 1982). The strong seasonal variations, however, indicate that changes in temperature and biological activity are key factors as previously demonstrated experimentally in Danish lakes (Jensen & Andersen, 1992). During winter, sedimentation of organic matter and the mineralization processes in the sediment are slow and the sediment has a relatively good P-sorption capacity because oxidisers like oxygen and nitrate penetrate the sediment (Andersen, 1982; Jensen & Andersen, 1992). During spring and summer when temperature, biological activity and sedimentation increase, the oxidised surface layer is diminished, implying that the sorption of P entering this zone from above (sedimentation) and below (transport upwards from deeper parts in P-rich sediments) or from P retained during winter is less sufficient. A similar seasonal pattern has been suggested for certain shallow freshwater (Søndergaard et al., 1993) and marine areas (Boers et al., 1998).

Enhanced temperatures also stimulate the mineralization of organic matter, thereby releasing inorganic phosphate to the interstitial water (Boström et al., 1982; Jensen & Andersen, 1992), and eventually then – depending on the sorption capacity of the sediment – to the overlying water. The mineralization may involve not only newly settled material, but also organic matter previously settled during winter. Furthermore, increasing temperatures and biological activity are likely to enhance phosphorus transport rates from deeper layers of the sediment as also indicated by the seasonality, which porewater profiles of phosphate and of various substances important to the mineralization processes

show (Søndergaard, 1990; Belzile et al., 1996; Urban et al., 1997). Finally, photosynthetically elevated pH in eutrophic lakes may increase release rates (Søndergaard, 1988; Welch & Cooke, 1995; Istvanovics & Petterson 1998), mediated through increased solubility of iron-phosphate compounds at increasing pH (Lijklema, 1976), but see Jensen & Andersen (1992).

Phosphorus retention is less negative in June, both in lakes with $TP_{sum}$ between 0.1–0.2 mg P $l^{-1}$ and $> 0.2$ mg P $l^{-1}$. Although we have no direct measurements evidencing the mechanisms behind this pattern, it is probably linked with the clearwater phase typically appearing in late May and early June. This clearwater period has been interpreted as a consequence of late-spring development of a high zooplankton biomass and its potential grazing on phytoplankton (Luecke et al., 1990; Jeppesen et al., 1997). The clearwater phase usually disappears abruptly when the young-of-the-year fish of zooplanktivorous roach and bream start foraging in the pelagic. The coupling between clearwater conditions and decreasing internal loading may involve several mechanisms including reduced sedimentation of organic matter as discussed above and enhanced benthic primary production taking up phosphorus and oxidizing the sediment surface (Van Luijn et al., 1995). Supporting the importance of clearwater conditions for decreasing internal phosphorus loading are observations from biomanipulation experiments where reduced biomass of zooplanktivorous fish and improved transparency often led to decreased TP (Søndergaard et al., 1990; Benndorf & Mierch, 1991; Nicholls et al., 1996; Jeppesen et al., 1998).

In the eutrophic lakes, TP retention was highly negative in May and in lakes with $TP_{sum}$ above 0.2 mg P $l^{-1}$ more negative than later in the year, indicating that seasonal retention not only relates to temperature-depending mechanisms. Several factors, which cannot be determined from our data, may be involved, but it can be anticipated that some of the winter-retained phosphorus is being rapidly released from the surface sediment at the onset of the increasing biological activity in spring, when the sedimentation of a phytoplankton spring maximum results in a diminished oxidized surface layer.

From a management point of view, a major problem in the interpretation of seasonal TP and retention data as described above is to determine whether the pattern seen represents equilibrium conditions or a recovery situation after reduced external loading. Unfortunately, long-term data records quantifying the loading history of lakes, are rare. In Denmark, we do have some information, however, based on measurements in rivers providing us with some indications. The annual median TP concentration in 36 Danish streams and rivers, which were sampled continuously from 1978 to 1988, showed a decrease from about 0.65 to 0.25 mg P $l^{-1}$ from 1978 to 1981, but no changes took place in the following yrs (Kronvang et al., 1997). These rivers were mainly relatively small and the reduced phosphorus concentrations were addressed to the improved treatment and diversion of waste water measures implemented in the 1970s and 1980s with a view to reducing lake loading (Kronvang et al., 1997). In another data set calculating annual mean phosphorus concentrations in mainly large rivers discharging into the sea or coastal areas, the phosphorus reduction was recorded somewhat later. During the 80s, the mean phosphorus concentration ranged between 0.6 and 0.65 mg P $l^{-1}$, but from 1990 to 1994 the concentration gradually decreased to 0.20–0.25 mg P $l^{-1}$ (Svendsen et al., 1997). Lakes presented in this study therefore represent various loading histories, but lakes to which loading has been reduced within the past 10–15 years constitute a significant part, particularly among the most eutrophic ones. This is supported by the finding that the retention in lakes with $TP_{sum}$ above 0.2 mg P $l^{-1}$ is much lower than predicted from the Vollenweider model. It is consequently most reasonable to consider most of the eutrophic lakes in this study as being in recovery after reduced external loading.

Hypertrophic Lake Søbygaard is an illustrative example showing the importance and longevity of internal loading after reduced external loading. For 15 years now, annual TP retention has been negative and no decreasing trend has yet been traced. From 1983 to 1995, retention ranged from $-1.9$ to $-5.4$ g P $m^{-2}$ $y^{-1}$ (Jeppesen et al., 1998) and since 1995 from $-2.1$ to $-3.3$ g P $m^{-2}$ $yr^{-1}$ (authors' unpubl. data). Net release depended on the biological structure and was positively related to chlorophyll *a* (Jeppesen et al., 1998).

It is usually difficult to discover long-term changes in the sediment of lakes and compare these with changes in net internal loading because the sediment P-pool is much higher than the annual net loading. In Lake Søbygaard, which has had a high negative retention for many years, however, a gradual decrease in sediment phosphorus concentrations may be observed. Earlier studies showed that phosphorus was mainly released from 5 to 15 cm depth during the

first five years and from 5 to 20 cm during the first eight years following the external loading reduction (Søndergaard et al., 1993). The 1998 profile seems to confirm the tendency that phosphorus is being released from deeper and deeper sediment layers and for the last 7 years phosphorus seems to originate from all sediment depths as low as approx. 25 cm. This is deep compared to assumptions for other lakes (Boström et al., 1982), and our findings suggest that deeper P-pools in lakes not necessarily and not eventually will be permanently buried in the sediment, but that a net release may occur from gradually deeper sediment parts if a mobile pool is present. This is an important aspect to consider when evaluating the potential duration of internal loading.

The differences between net release calculated from mass balance measurements (39 g P $m^{-2}$ released since 1985) and those calculated from changes in sediment profiles (approximately 57 g P $m^{-2}$ released from the upper 28 cm) are remarkably low, considering the often significant inter-lake variability in sediment TP (Downing & Rath, 1988) and coring difficulties (shortening of cores when sampling soft sediment) (Blomqvist, 1985). Bearing in mind the difficulties as to discerning between temporal and spatial variation when sampling sediment and assuming that the concentration level and the volumetric content, which is now rather constant in the upper 12 cm (Figure 4, Table 3), also are the levels expected to be reached in a future state of equilibrium in the parts of the sediment from which phosphorus is now being released, then approximately additionally 60 g P $m^{-2}$ is expected to be released from the Lake Søbygaard sediment. At the present release rate this means that another 15–25 years will pass before the lake will eventually be in equilibrium, implying that the transient phase after reduced external loading may last for more than 30 years.

We believe that the data presented in this study, which are based on data and mass balances from a large number of lakes, provide a general description of the importance of internal phosphorus loading in eutrophic temperate shallow lakes to which external loading has been reduced during or within the past 10–20 years. It may be concluded that phosphorus release from the sediment has a pronounced impact for many years on summer phosphorus concentrations in these lake types.

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# 6. Impact of Submerged Macrophytes on Phytoplankton in Shallow Freshwater Lakes

Martin Søndergaard and Brian Moss

## Introduction

The ubiquity of phytoplankton, and its fundamental importance as a primary producer and mediator of many major biological processes in lakes, has led to comprehensive research on its biology. Its importance for water quality and its increasing predominance as the main primary producer at the expense of submerged macrophytes in shallow lakes follow from increased nutrient loading. Often, macrophytes have completely disappeared, and nutrients are so abundant that it is difficult to conceive much bottom-up control of phytoplankton through especially phosphorus. Restriction of the nutrient loading to reduce the amount of phytoplankton and to increase water clarity and restore a more diverse biological structure has now started to reverse this process in many areas. Simultaneously, we can anticipate a renewed importance of submerged macrophytes in many lakes.

Most research has, however, been on phytoplankton in the pelagic environment, and few studies have focused on phytoplankton in relation to the presence of submerged macrophytes. Thus, the existence and relative importance of top-down control of phytoplankton by zooplankton and/or bottom-up control through nutrients, which is well known from the pelagic, is poorly documented from macrophyte beds. It is largely unknown whether the mechanisms in the macrophyte beds are similar or to what extent different densities of macrophytes may influence a trophic cascade. By their structuring effect, macrophytes create an environment that is fundamentally different from that of the open water and that potentially may

have great impact on the interactions between the different trophic levels (Jeppesen et al., this volume, Chapter 5).

In this chapter, we describe how submerged macrophytes may affect phytoplankton biomass and community structure. We focus on temperate shallow lakes with the potential for an extensive macrophyte cover. It is not our intention to give a comprehensive review but to highlight some of the clearest and, in our opinion, most important relationships. We supplement these with some of our own experience and results, particularly from large-scale enclosure experiments in the shallow and eutrophic Lake Stigsholm, Denmark, and Little Mere in the United Kingdom, where macrophyte and fish densities have been manipulated. We also try to create a framework that defines hypotheses about which phytoplankton biomass and species differences to expect along nutrient, zooplankton, and macrophyte gradients.

## Effect of Macrophytes on Phytoplankton Biomass

Assuming a given external flux of nutrients and fish-mediated grazing pressure from zooplankton, several mechanisms may be responsible for the general observation that a lower phytoplankton biomass is found in the presence compared with the absence of macrophytes. The most conspicuous, and probably one of the best documented mechanisms, is that of enhanced grazing pressure from pelagic zooplankton.

### Increased Grazing

Large-sized pelagic zooplankters generally occur in greater numbers inside or around the edges of macrophyte beds than outside, although this effect can be modified by fish (see Jeppesen et al., this volume, Chapter 5, for a comprehensive discussion). They appear to use the macrophyte beds as a daytime refuge against fish predation. The concept of this mechanism was first introduced by Timms and Moss (1984) and has been confirmed or indicated by several other studies (De Meester et al., 1993; Stansfield et al., 1995; Lauridsen et al., this volume, Chapter 13). It is discussed in detail by Jeppesen et al. (this volume, Chapter 5), who also suggested that there is much higher grazing pressure on phytoplankton inside than outside the macrophyte beds. An additional consequence of macrophyte presence for decreasing phytoplankton biomass is that their effects seem to extend beyond the border of the macrophyte beds. A horizontal diurnal migration of pelagic zooplankton from dense plant beds covering only 3% of the lake is enough to increase the grazing potential of zooplankton in the open water by 100% (Lauridsen et al., 1996).

Grazing by pelagic zooplankton may often be supplemented by grazing from plant-associated zooplankton species (Irvine et al., 1989; see also Jeppesen et al., this volume, Chapter 5) and other invertebrate filter and suspension feeders (e.g., insect larvae, mollusks). For obvious reasons, plant-associated zooplankters occur

in much higher densities in macrophyte beds than outside. For example, in experiments conducted in Little Mere in 1993, the number of grazers in mesocosms containing plants was much greater than in those without, with consequent reductions in chlorophyll *a* concentrations (Beklioglu and Moss, 1996). In addition, Schriver et al. (1995), using 100-$m^2$ enclosures in Lake Stigsholm, recorded a decline in phytoplankton biomass with increasing macrophyte density in terms of percentage volume infestation (PVI) even when planktonic cladocerans were absent. They argued that plant-associated zooplankton such as chydorids, *Pleuroxus* and *Eurycercus,* which usually were found in tenfold higher densities during both day and night in the vegetation, could be of significance. Although the effects on phytoplankton have been rarely documented directly, there are thus reasons to believe that plant-associated filter feeders are important in decreasing the phytoplankton biomass within macrophyte beds.

## Changes in Nutrient Cycling

The presence of macrophytes may in various ways influence nutrient cycling (see Barko and James, this volume, Chapter 10) and thus potentially also phytoplankton biomass and the growth and competition among different phytoplankters. Decreased availability of nitrogen and sometimes phosphorus may be expected in most cases.

Macrophytes, and their associated epiphytes, take up and store nutrients, which are then not available for the phytoplankton (Kufel and Ozimek, 1994; O'Dell et al., 1995). A rapid growth of *Elodea* in Lake Zwemlust was followed by limitation of nitrogen during summer (Ozimek et al., 1990), and it was calculated that 64% of the total lake nitrogen and 61% of that of phosphorus (excluding the sediment pool) were found in the macrophyte compartments (van Donk and Gulati, 1995). Indirect effects are also likely to explain the generally low nitrogen concentrations often seen in the presence of macrophytes, even when the uptake by macrophytes themselves is considered low. The structured environment inside macrophyte beds, which may include gradients leading to anoxic conditions, may increase denitrification (Weisner et al., 1994; Jeppesen et al., submitted a). Changes in nitrogen competition between phytoplankton and sediment-associated bacteria may also be important: when phytoplankton biomass is low (in the presence of macrophytes), the nitrate uptake by the bacteria in the sediment and the subsequent denitrification to $N_2$ is relatively high compared with a situation with a high phytoplankton biomass (when macrophytes are absent) in which more nitrate is taken up and stored as organic nitrogen by the phytoplankters. The overall nitrogen removal is therefore higher in the presence of macrophytes. The general importance of these mechanisms, however, still needs to be verified, although it is indicated by several whole-lake studies where the macrophyte density has changed markedly (van Donk et al., 1990; Jeppesen et al., 1997; Moss et al., 1997).

When phosphorus availability is concerned, the mechanisms are probably more diverse and the outcome more complicated. Brammer (1979) explained an ex-

clusion of phytoplankton in the proximity of a dense stand of *Stratiotes aloides* to be caused by competition for nutrients together with the co-precipitation of phosphorus with calcium carbonate on the leaf surfaces of the submerged plants. Very dense macrophyte beds, however, may also increase the availability of phosphorus (Stephen et al., 1997). Canopy-forming types, poorly rooted species such as *Elodea* and dense *Chara* beds, which create dense shade, may have more or less anoxic conditions in their bottom layers (Pokorny et al., 1984; Moss et al., 1986), thereby increasing the potential for phosphorus release from the sediment, particularly where the iron-bound and redox-sensitive phosphorus fractions constitute a significant part of the total P in the sediment. This is consistent with observations in Cockshoot Broad, United Kingdom, in which a recovery of macrophytes was associated with an increased release of phosphorus from the sediment (Moss et al., 1996). An analogous mechanism is an increased phosphorus release from the sediment due to enhanced pH, caused by high photosynthetic activity, which has been shown to affect the phosphorus sorption mechanisms in the surface sediment (Søndergaard, 1988). Plant beds frequently have high pH in the surface layers but lower values in the darker, deeper layers toward the sediment (Frodge et al., 1990). The ultimate effects of pH variation in plant beds are thus complex. Finally, release of organic carbon compounds from macrophytes may also stimulate bacterial production (Wetzel and Søndergaard, this volume, Chapter 7), and thus indirectly increase bacterial phosphorus uptake at the expense of phytoplankton.

Less dense macrophyte beds and macrophytes having a more developed root system may, however, affect sediment release rate in a converse way by oxidizing the surface sediment (Andersen and Olsen, 1994; Flessa, 1994; Christensen and Andersen, 1996), thereby reducing the redox-sensitive phosphorus release. The final outcome for phosphorus availability in macrophyte beds may therefore depend on both macrophyte species and density. In general, however, submerged macrophytes may be regarded as phosphorus sinks during their active growth and only as a potential phosphorus source during the relatively short periods of their senescence (Carpenter and Lodge, 1986; Rørslett et al., 1986; Jones, 1990).

Although change in nutrient concentrations is probably related to macrophyte density, it is often difficult to show its effects on phytoplankton biomass because of masking by other mechanisms acting simultaneously. In their enclosure experiments, Schriver et al. (1995) noted that the concentrations of orthophosphate and ammonium were strongly and positively correlated with the potential grazing pressure from zooplankton. They suggested that the shift to low phytoplankton biomass at high grazing pressure from zooplankton should be attributed to enhanced grazing rather than nutrient control of algal growth. In that particular case, nutrient changes did not seem to be important. In enclosure experiments in Little Mere, phosphate and ammonium concentrations were also unaffected by the presence or absence of macrophytes but strongly influenced by the numbers of zooplankters, responding to changes in fish predation (Beklioglu and Moss, 1996). The concentrations of chlorophyll *a* in the phytoplankton, however, were significantly and positively related to the number of fish both in the presence or absence of macrophytes.

## Allelopathy

Release of organic compounds from macrophytes, in particular from *Chara*, with allelopathic effects on phytoplankton has also been suggested to reduce phytoplankton production and biomass (Wium-Andersen, 1982; Gross, 1995; van Donk and Gulati, 1995; Gross et al., 1996) or cause changes in the phytoplankton community structure (Jasser, 1995). However, in field experiments using whole plants and in laboratory experiments using macrophyte extracts, Jasser (1995) only recorded a minor effect on phytoplankton biomass in the presence of macrophytes or their extracts, although major effects were seen in the community structure. Similarly, Forsberg et al. (1990) did not find any allelopathic effects of *Chara* on chlorophyll *a* concentrations when comparing phosphorus/chlorophyll *a* relationships in *Chara*-dominated and non-*Chara* lakes. Furthermore, they also noticed a dense epiphytic growth on the *Chara* plants, which they interpreted as an absence of allelopathy by *Chara* in situ. The importance of allelopathic substances, although undoubtedly they exist and can be shown to act in laboratory situations, remains still to be fully documented at the ecosystem level.

## Increased Sedimentation

Increased sedimentation and reduced resuspension, due to the less turbulent and more quiescent waters among the macrophytes, is yet another factor that may negatively influence the phytoplankton community (see also Barko and James, this volume, Chapter 10). In Chesapeake Bay, with semidiurnal tides, Kemp et al. (1984) recorded a lower phytoplankton biomass in a littoral weedbed area compared with nearby open water. They ascribed this to increased sedimentation within the weedbed. Similarly, Jones (1990) recorded a consistently lower phytoplankton biomass in the vegetated areas of the freshwater Potomac River than in the adjacent unvegetated reaches. Van den Berg et al. (this volume, Chapter 25) recorded a 30-fold lower phytoplankton density inside dense *Chara* beds than outside and attributed it to stiller conditions. In other situations, for example Balls et al. (1989), increased sedimentation of phytoplankton in macrophyte beds in ponds was not found to be important because the phytoplankton community became dominated by flagellates. The importance of this mechanism may also be questioned because of the very high phytoplankton biomass found in nutrient-rich brackish lakes, even when macrophyte coverage is substantial (Baks et al., 1993; Jeppesen et al., 1994; Jeppesen et al., this volume, Chapter 28).

## Reduced Light Conditions

Finally, there may be shading effects of macrophytes on phytoplankton (Sand-Jensen, 1989; Ozimek et al., 1990). The shading effect is highly related to macrophyte biomass, but macrophyte surface area also depends on plant species (Sher-Kaul et al., 1995), and light extinction coefficients of macrophyte canopies show a fourfold variation among canopy species (review Carpenter and Lodge, 1986). Dense beds are undoubtedly self-shading (Nielsen and Sand-Jensen, 1989; Frodge et al.,

1990). However, quite dense periphyton communities may persist within them, and epipelic algae can be found on the bottom sediment among the plants. The strategies of buoyancy and motility are available to the phytoplankton community to overcome this potential problem.

## Macrophyte Threshold Effects

In most of the examples mentioned above, the conclusions have been drawn from comparisons between macrophyte-free and macrophyte-rich areas. An important aspect, not least from a management point of view, is whether macrophyte density must attain a certain threshold before they have a significant effect on the phytoplankton. Several studies indicate that this is the case. Canfield et al. (1984) noted that water transparency was generally high in lakes with a macrophyte PVI exceeding 30%, and Schriver et al. (1995) found a threshold level of 15–20% PVI (*Potamogeton pectinatus* and *Potamogeton pusillus*), above which a sudden decrease from 30 $mm^3/L$ to 10 $mm^3/L$ in phytoplankton biovolume was recorded (Fig. 6.1A). Later experiments in Lake Stigsholm suggested a relatively low refuge effect for zooplankton at a PVI below 20–24%, but a significant effect at a PVI of 50% (Jeppesen et al., this volume, Chapter 5). The presence of a threshold and a nonlinear relation with macrophytes is also proposed with regard to the impact of mollusks (Brönmark and Vermaat, this volume, Chapter 3) and the refuge efficacy for zooplankton (Jeppesen et al., this volume, Chapter 5).

## Consensus

Although it may be difficult or even futile to attempt to determine which of the mechanisms mentioned above is decisive, for several may act together, it is becoming increasingly indisputable that strong antagonistic forces exist between macrophytes and phytoplankton and that the presence of macrophytes generally is tantamount to low phytoplankton biomass in freshwater lakes. This statement is supported by numerous studies comparing phytoplankton in the presence and absence of macrophytes, including examples from this book (Blindow et al., this volume, Chapter 26; Faafeng and Mjelde, this volume, Chapter 27; Jeppesen et al., this volume, Chapter 28; Van den Berg et al., this volume, Chapter 25). Balls et al. (1989) also concluded that in experimental ponds where the submerged plants remained intact, plants were able to buffer strongly the effects of added nutrients. Despite large loadings, nutrient concentrations in the presence of plants could not build up sufficiently to support large algal growths. By contrast, where the plants were artificially removed, in a set of replicate but cleared ponds, sufficiently large concentrations did establish and support large algal crops. In 20-$m^2$ experimental enclosures in the Danish Lake Stigsholm, a fourfold higher chlorophyll *a* concentration and a 10–25-fold higher phytoplankton biomass were recorded in macrophyte-free enclosures compared with enclosures with a 50% PVI (Table 6.1). The statement is also supported by empirical studies of more than 100 Danish lakes, in which Secchi disk transparency was, in general, much higher in

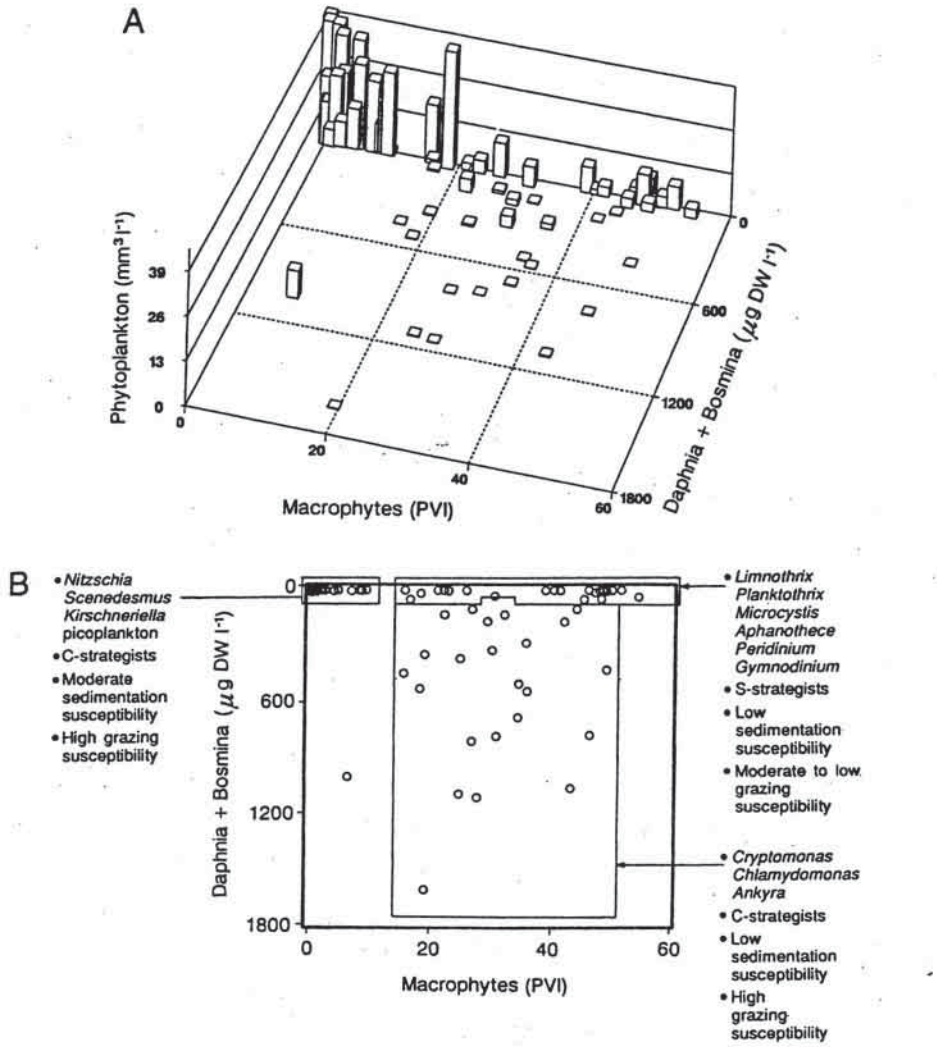



**Figure 6.1.** Phytoplankton biovolume (A) and community structure (B) in relation to macrophyte PVI (percentage volume infestation) and *Daphnia* + *Bosmina* biomass. (Modified from Schriver et al., 1995.)

lakes with a high macrophyte coverage compared with those without plants at comparable nutrient concentrations (Jeppesen et al., 1990).

Despite a number of reservations and huge differences among lakes and environmental conditions, we have attempted to summarize the effects and make suggestions concerning the relative importance of different factors associated with macrophytes on phytoplankton biomass in Figure 6.2.

**Table 6.1.** Chlorophyll *a* Concentrations (Mean of 1 Daytime and 1 Night-Time Sampling in 3 Enclosures, ± SD) and Phytoplankton Biomass (Mean of 3 Night-Time and 3 Daytime Samplings in 3 Enclosures ± SD) in Macrophyte-Free ($n = 3$) and Macrophyte-Rich (50% PVI, $n = 3$) 20-$m^3$ Enclosures from July 25 to 27, 1994, in Lake Stigsholm

| | With macrophytes ($n = 3$) | | Without macrophytes ($n = 3$) | |
|---|---|---|---|---|
| | Day | Night | Day | Night |
| Chlorophyll *a* (µg/L) | 17 (19) | 8.3 (4.0) | 64 (10) | 37 (7.9) |
| Phytoplankton biomass ($mm^3$/L) | 0.80 (0.78) | 0.31 (0.21) | 8.1 (1.4) | 7.6 (2.1) |

All observations comparing enclosures with and without macrophytes are significantly diferent ($P < .01$, *t* test unequal variance). Only night and day observations of chlorophyll *a* in macrophyte-free enclosures are significantly different ($P = .02$, *t* test unequal variance).

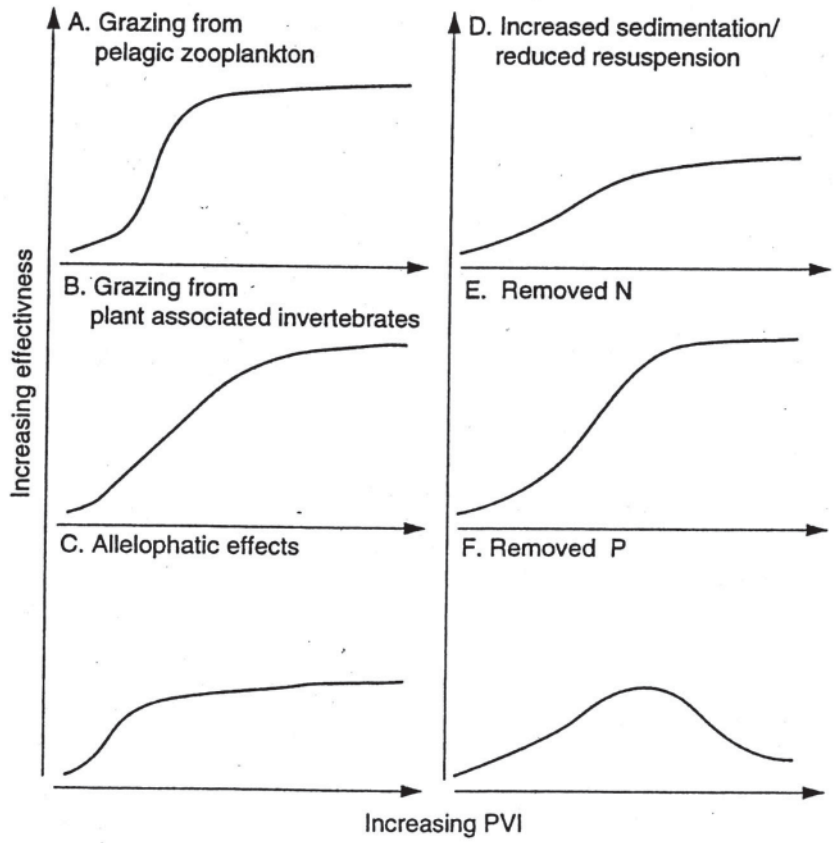



**Figure 6.2.** The relative importance of different mechanisms potentially affecting the phytoplankton biomass at increasing macrophyte PVI (percentage volume infestation).

The importance of grazing by pelagic zooplankton species is probably not linearly proportional to macrophyte density because of a need for macrophytes to attain a threshold density before they have a refuge effect (Fig. 6.2A). Furthermore, the macrophyte density required depends on fish density, the refuge effect being diminished at increasing fish density (Jeppesen et al., submitted b; Jeppesen et al., this volume, Chapter 5). It is also likely that the importance of grazing by pelagic zooplankton diminishes in very dense macrophyte beds. *Daphnia,* in particular, are frequently most abundant at the edges of macrophyte beds rather than in their interiors (Lauridsen et al., 1996). As beds cover the lake, edge habitat is diminished. We think the importance of grazing overall is likely to be high in most situations, however.

Having exceeded a certain minimum macrophyte density, grazing from plant-associated invertebrates is expected to increase linearly with macrophyte density (Fig. 6.2B). The effect, however, again may depend on fish predation, although in mesocosm experiments in Little Mere (Beklioglu and Moss, 1996), there was no effect of increasing perch (*Perca fluviatilis*) numbers on densities of plant-associated *Cladocera* (*Chydorus, Eurycercus, Simocephalus, Sida*), although severe reductions of *Daphnia* numbers, despite the presence of plant effects. At present, it is difficult to assess the relative importance of plant-associated grazers as there are few quantitative data and conditions in which such grazing is likely to be high (in dense plant beds) are also those in which nutrient effects may be strong.

Allelopathic effects, if present (Fig 6.2C), may be predicted to be proportionately more important at low plant densities, where the plants are growing vigorously before self-shading and perhaps nitrogen limitation sets in, and to reach a plateau of effectiveness at moderate macrophyte densities. Bacterial decomposition of the substances is likely to increase as the beds become dense. The effectiveness of allelopathy is also likely to be dependent on macrophyte species. Based on the present evidence, we suggest that allelopathy is generally of lesser importance in the ecosystem context than other factors.

Effects of stilling of the water and increased sedimentation and/or reduced resuspension are expected to increase proportionally with macrophyte density (Fig. 6.2D), but their overall importance is not well documented. We consider their importance moderate compared with other factors under most circumstances.

Reduction of nitrogen availability is likely to follow a pattern of increased effect after attainment of a threshold macrophyte density sufficient to account for considerable uptake and to create conditions appropriate to denitrification (Fig. 6.2E). This effect is probably an important one overall, especially in dense macrophyte beds and in lakes having a relatively low nitrogen loading and consequently the highest possibility of nitrogen limitation of phytoplankton.

The importance of effects on phosphorus will depend more on macrophyte type and density than most of the other factors. If the macrophyte beds are not too dense, the effect is likely to be one of reduction in concentration; but if macrophytes become very dense, the effect may be the opposite, with the risk of an enhanced release from the sediment (Fig. 6.2F). We think that in terms of

mechanisms limiting the growth of phytoplankton in weedy, eutrophic lakes, phosphorus effects are unlikely to be very important.

## Fish- and Nutrient-Mediated Effects of Macrophytes on Phytoplankton Biomass

We have not yet fully considered the influence of fish and nutrients. There are reasons to believe, however, that fish modify the influence of macrophytes on phytoplankton. In Chapter 5, it was shown that the behavior of fish and zooplankton and their interactions were highly dependent on macrophyte density. We must therefore anticipate consequential effects on phytoplankton. Based on the threshold level of macrophyte density of 15–20% PVI influencing fish-mediated impact on zooplankton in the Lake Stigsholm experiments (Schriver et al., 1995) and on other results, we have summarized, in Table 6.2, phytoplankton changes along a nutrient, macrophyte, and fish gradient. Schriver et al. (1995) argued also for the existence of a fish threshold. They concluded that the effects on phytoplankton biomass of a high macrophyte coverage were only important as long as the fish density did not exceed approximately 2 fry/$m^2$. If it was higher, the refuge effect seemed low and even a high macrophyte density would be unable to have any major zooplankton-mediated effects on the phytoplankton. These generalizations are probably modified by the actual nutrient level, and it may, for instance, be possible that nitrogen limitation effects may become more important under such circumstances because high fish densities can frequently be found in association with dense plant beds.

In Table 6.2, we also suggest how some phytoplankton variables may change along nutrient, macrophyte, and fish gradients. Such a generalization is complicated by the fact that the macrophyte community itself will also change along a nutrient gradient. Small vascular plants and *Charophyte*-dominated communities are most common at low nutrient concentrations (say, <50 µg total P/L). Small, shallow, and very fertile waters (say, >200 µg total P/L) tend to become dominated by nymphaeids (lilies), with their partly emergent foliage, perhaps because periphyton infestation inhibits maintenance of submerged leaves. Intermediately fertile waters have mixed communities of tall submerged vascular plants, although charophytes and small vascular plants may still be present and lilies usually are. Clear water is maintained at all combinations of nutrient concentrations and macrophyte densities, except low macrophyte density/high nutrients and high fish densities, by the mechanisms discussed above. Grazing, either by daphnids or by plant-associated animals, is likely to be important in all circumstances in which plants persist, although the balance of these two grazer groups will differ. Daphnids may be less important in very dense submerged macrophyte stands.

The relative importance of the availability of nitrogen and phosphorus is likely to vary along the gradients (Table 6.2), with phosphorus becoming less limiting and nitrogen more so as both nutrient loading and macrophyte density increase.

**Table 6.2.** Generalized Matrix Showing Proposed Effects of Nutrient Concentration, Fish Density, and Macrophyte Density on Phytoplankton Biomass, Phytoplankton Community Structure, Phytoplankton Cell Size, Transparency, N/P Limitation, and Dominant Macrophtyes[a]

| | Low nutrient | | High nutrient | |
|---|---|---|---|---|
| | Low fish | High fish | Low fish | High fish |
| *Phytoplankton biomass* | | | | |
| Low Macrophytes | Low | Low/medium | Medium/high | High |
| High macrophytes | Low | Low | Low/medium | Medium/high |
| *Phytoplankton community* | | | | |
| Low macrophytes | Flagellates, zyg | Flagellates, zyg | Diatoms, chlor Flagellates | Diatoms, cyano, chlor |
| High macrophytes | Flagellates, zyg | Flagellates, zyg | | Cyano, flagellates |
| *Phytoplankton cell size* | | | | |
| Low macrophytes | Small/large | Small/large | Medium/small | Large/medium |
| High macrophytes | Small/large | Small/large | Small | Small/large |
| *Transparency* | | | | |
| Low macrophytes | High | High/medium | Medium/low | Low |
| High macrophytes | High | High | High/medium | Low/medium |
| *N/P limitation* | | | | |
| Low macrophytes | P | P | N/P | N/P |
| High macrophytes | N/P | N/P | N | N |
| *Dominant macrophytes* | | | | |
| Low macrophytes | s. vas/t. vas | s. vas/t. vas | t. vas/lilies | t. vas/lilies |
| High macrophytes | s. vas/t. vas | s. vas/t. vas | t. vas/lilies | t. vas/lilies |

Abbreviations: cyano, cyanophytes; chlor, chlorococcales; zyg, zygnemetales; s. vas, small vasculars and charophytes; t. vas, tall vasculars.

[a]Approximate definitions used in this context: low nutrient: $P < 25$ µg/L and N/P high; high nutrient: $P > 100$ µg/L and N/P low. Low fish density: <2 fry/m$^2$; high fish density: >2 fry/m$^2$. Low macrophyte density: PVI < 15%; high macrophyte density: PVI > 30%.

This is because the nitrogen to phosphorus ratios in inflowing waters modified by human activities, especially those concerning sewage and stock effluents, tend to be relatively richer in phosphorus than is required for nutrient sufficiency by algae and because phosphorus is mobilized in dense plant beds, whereas combined nitrogen is denitrified. Finally, the nature of the phytoplankton is likely to change along the gradients, with large cells or colonies being favored by increasing nutrient concentrations, large nitrogen fixers becoming more abundant as nitrogen stress increases, but small and often flagellate cells and cyanophytes being favored at high densities of submerged plants due to sedimentation of large nonbuoyant cells under such conditions.

## Indirect Effects of Macrophytes on Phytoplankton Composition and Size

The impact of macrophytes on phytoplankton extends not only to biomass. Macrophytes may also have marked effects on the species and size of individuals forming the community. As with biomass, several mechanisms may be involved in determining species composition. Change in the nutrient regime, as mentioned above, is probably one of the most important. But other factors such as differential tolerance to shading (Sand-Jensen, 1989), susceptibility to sedimentation in the darker and quiescent water inside the macrophyte beds, allelopathic effects (Jasser, 1995), and increased input of particulate dissolved organic carbon from the macrophyte–epiphyte complex, altering the planktonic food chain and thereby also influencing the phytoplankton, may also be important.

One general tendency is that of increased importance of flagellates in the presence of macrophytes and/or large-sized zooplankton. For example, Balls et al. (1989), in fish-free pond experiments, observed no systematic pattern of change with nutrient loading in either plant-dominated or cleared ponds and a great deal of variation within the limits of communities all dominated by small and often flagellated organisms. They attributed this to a stronger selection for small organisms with low sinking rates in the very short water column, compared with any possible selection for large phytoplankters through grazing. Godmaire and Planas (1986) in enclosure experiments also recorded a prevalence of flagellates during the entire season in the presence of *Myriophyllum spicatum* and suggested that neither light nor nutrient conditions seemed responsible for this particular community composition. And in Lake Veluwemeer, Van den Berg et al. (this volume, Chapter 25) recorded a shift in species composition from cyanobacteria to flagellates over a transect from no vegetation to a dense vegetation of *Chara*.

A tendency toward a greater dominance of small flagellates at increasing macrophyte abundance, despite an increase in large-sized daphnids, was also observed by van Donk et al. (1990) in a whole-lake experiment in the biomanipulated Lake Zwemlust (The Netherlands). In this lake, submerged macrophyte density increased from 0 to 80% within 3 years, and apart from a much lower phytoplankton biomass, this was accompanied by a shift toward small edible species. Larger phytoplankton species such as *Aphanizomenon*, which is considered more resistant to zooplankton grazing, did not develop. This was explained by dependence of phytoplankton growth on several factors operating simultaneously or successively. Thus, zooplankton grazing in spring, nitrogen limitation caused by the macrophytes and grazing in summer, and temperature and light availability in winter controlled phytoplankton growth. It was concluded that large-sized inedible phytoplankton taxa were unable to oust the small-sized and edible but fast-growing C strategists, which were more effective in taking up nutrients during nutrient limitation due to their high surface-to-volume ratio.

A shift to edible cryptophyte flagellates has also been observed in the clear open water following a fish kill (Sarnelle, 1993) and in biomanipulated lakes where the fish stock has been reduced (Leah et al., 1980; Reinertsen et al., 1990;

Søndergaard et al., 1990). Thus the effects of herbivory on algal succession were not predictable from the relative susceptibilities of these algal species to grazing mortality.

In a series of 100-$m^2$ enclosures, Schriver et al. (1995) investigated the importance of zooplankton and macrophyte densities on the phytoplankton composition. Among their findings was that when *Daphnia* and *Bosmina* were abundant, fast-growing small flagellates such as *Cryptomonas* and *Chlamydomonas* dominated. Although cryptophytes (as percentage of biomass) were positively related to both zooplankton biomass and the interaction between macrophyte PVI and zooplankton biomass, *Chlamydomonas* was significantly related only to zooplankton biomass, indicating that *Cryptomonas* is more dependent on the presence of macrophytes. In the absence of planktonic cladocerans but in the presence of macrophytes, cyanophytes (mainly represented by *Planktothrix, Limnothrix, Microcystis,* and *Aphanothece*) and dinoflagellates (mainly *Gymnodinium* and *Peridinium*) were dominant. Cyanophytes were negatively related to the interaction between PVI and zooplankton biomass. In the absence of both planktonic cladocerans and macrophytes, the community was dominated by fast-growing diatoms (mainly *Nitzschia*) and chlorophytes (mainly *Scenedesmus* and *Kirchneriella*). Chlorococccales were therefore negatively related to PVI*zooplankton biomass. Basically, these findings are in agreement with the life history strategies described by Reynolds (1987). Fast-growing algae are moderately susceptible to loss by sedimentation and are dominant in the turbulent environment of the macrophyte-free areas, whereas slow-growing buoyant cyanophytes are dominant in the more quiescent water among the plants, where the risk of sedimentation is higher.

However, flagellates such as *Cryptomonas,* with a high grazing susceptibility, were present at high grazing pressure from zooplankton, and cyanophytes, with low susceptibility, were present at low grazing pressure, contrary to expectation, but also often seen in whole-lake studies. On the basis of these findings, Schriver et al. (1995) argued that the term *grazing susceptible* should maybe be replaced by *grazing resistant* and *grazing tolerant,* the implication being that grazing-resistant algae such as *Microcystis* are only favored until grazing pressure increases to a certain limit whereafter grazing-tolerant algae such as *Cryptomonas* take over (see also Table 6.2). The results of Schriver et al. (1995) have also been summarized in Figure 6.1B, which defines the three groups of phytoplankton.

It may be asked why small flagellates, vulnerable to zooplankton grazing, seem to be such successful competitors in the presence of macrophytes, with the considerable grazer pressure associated with them. A possible explanation (Sommer, 1988) is that flagellates, due to their motility, are better adapted to exploit an environment that is heterogeneous and structured with respect to nutrient distribution, such as that created by macrophytes. Furthermore, many flagellates are mixotrophic and able to use DOC excreted from macrophytes and their associated epiphytic environment. A combination of these advantages and their high growth rates could be a major reason to their success.

There may also be effects of macrophytes on species composition as well as changes in the balance of groups such as diatoms, cyanophytes, or flagellates. In

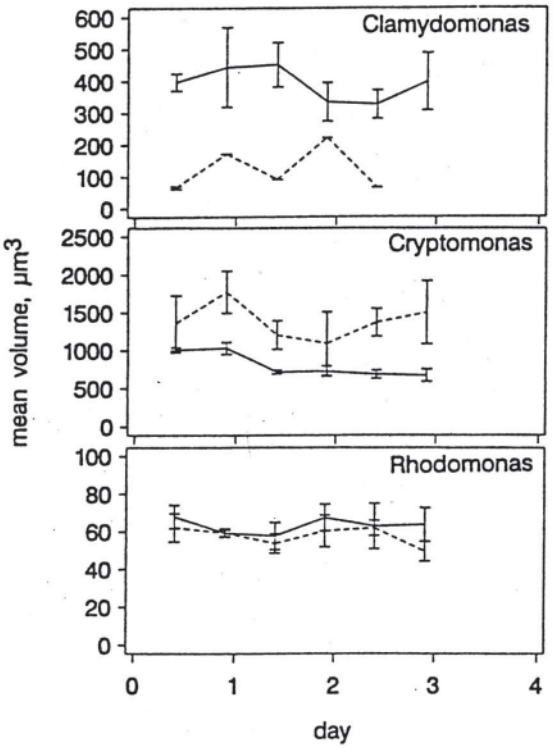



**Figure 6.3.** Biovolume ($\mu m^3$) of *Chlamydomonas* sp., *Cryptomonas* spp. (*C. reflexa, C. curvata,* and *C. marssonii*), and *Rhodomonas* sp. in 20-$m^2$ enclosure experiments in Lake Stigsholm from July 25 (12 AM) to 27 (12 PM), 1994 (sampling at 12 AM and 12 PM), using macrophyte-free (solid line) and macrophyte-dense (broken line) compartments. Mean values and standard error of three enclosures with and three enclosures without macrophytes. For total phytoplankton biomass, see Table 7.1.

the Little Mere mesocosm experiments, in which the phytoplankton community was studied in relation to different densities of fish (Beklioglu and Moss 1996), the presence or absence of macrophytes had no effect on the total biomass of diatoms but very significant effects on particular diatom species. *Aulacoseira granulata* was much less abundant and *Nitzschia palea* much more abundant when macrophytes were present.

Results from the Lake Stigsholm experiments, furthermore, indicate that macrophytes can also affect the size distribution at the phytoplankton genus/species level (Søndergaard et al., unpubl.). As an example, marked differences in the size of the flagellates *Chlamydomonas* and *Cryptomonas* were recorded between macrophyte-rich and nonmacrophyte areas, whereas no differences were recorded for *Rhodomonas* (Fig. 6.3). Whereas *Chlamydomonas* sp. was much smaller inside the macrophyte beds, the mean size of *Cryptomonas* spp. (three different species)

during the whole experimental period was significantly higher in the presence of macrophytes (1,100 to 1,800 $\mu m^3$) than in their absence (700–1,000 $\mu m^3$). Differences in grazing susceptibility and size-specific mechanisms to avoid grazing among different species within macrophyte beds could reinforce the suggestion that grazing is a major agent affecting the phytoplankton in and around macrophyte beds. The difference in size may also be related to differences in nutrient concentrations, which are known to influence the size of phytoplankton (Watson et al., 1992; Hansson and Carpenter, 1993; Larocque et al., 1996). No matter which mechanism lies behind the results, however, they indicate that macrophytes can have a structuring effect also on phytoplankton size distribution.

## Concluding Remarks

The presence of macrophytes clearly has a negative effect on phytoplankton biomass. There are indications that along a macrophyte density gradient, the response is nonlinear, a threshold being recognizable at a macrophyte density of approximately 15–30% PVI, depending on fish abundance. Several mechanisms contribute to maintenance of low phytoplankton crops in association with macrophyte beds. These include direct mechanisms associated with the plants themselves, including creation of a still water environment, poor light climate, and secretion of allelopathic substances as well as mechanisms indirectly linked with the plants, such as provision of refuges or habitat for grazers on algae, and modification of the ambient nutrient regime by the metabolic activity of the plants. In turn, these mechanisms are affected by a second level of influences—predation by fish on the grazers, imposition of nutrient loads from the catchment, and hydrodynamic conditions that determine the degree of water movement through the plant beds.

All these mechanisms interact. The hydrodynamics must determine the extent to which anoxic gradients that promote denitrification can be established; excretion by grazers modifies the nutrient regime. Furthermore, the influence of specific composition of the macrophyte community is likely to be important. There are metabolic as well as structural differences among different species of submerged plants, let alone those between, for example, nymphaeids and charophytes. The degree of grazing on algae will also be affected by the composition of the grazer community, which in turn, is determined by the size structure and composition of the fish community. The macrophyte growth will also be subject to grazing by vertebrates so that the structure and composition of the beds will vary.

Major gaps exist concerning the differential effects of different macrophyte species and of those of mixed fish communities. Most experiments have used monospecific or simple communities because of the need to balance adequate replication against the considerable costs of ecosystem experiments, even in mesocosms. A further gap is that of determining whether allelopathy is important in the complex ecosystem context. All in all, the mutual interaction of all these factors makes it highly unlikely that anything other than a very general model of

how the macrophyte and phytoplankton communities interact can be constructed. The importance of different mechanisms and processes will, to some extent, be lake-specific. Much progress has been made in this area through the use of mesocosm experiments, and this should continue. In addition, accumulating experience from a variety of different lakes may also begin to reveal greater generalities than we have been able to discern here. The comparative approach coupled with whole-lake and mesocosm experiments of increasing scope are likely to be the way forward.

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# PHOSPHORUS FRACTIONS AND PROFILES IN THE SEDIMENT OF SHALLOW DANISH LAKES AS RELATED TO PHOSPHORUS LOAD, SEDIMENT COMPOSITION AND LAKE CHEMISTRY

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**Abstract**—Depth profiles of sediment dry weight (DW), loss on ignition (LOI), total iron (Fe), total nitrogen (TN), calcium (Ca), total phosphorus (TP), $NH_4Cl$-P, NaOH-P, HCl-P and Res-P were measured in 32 shallow meso- to hypertrophic Danish lakes and the data compared with that for external phosphorus load (Pex), lake water phosphorus and phytoplankton biovolume. Surface sediment TP was highly and positively correlated to Pex (g P $m^{-2}$ $y^{-1}$) and Fe (mg Fe $g^{-1}$ DW) (TP = 0.92 + 0.23 Pex + 0.031 Fe, $r^2 = 0.68$, $P < 0.0001$), but not or only weakly related to the other parameters analyzed. NaOH-P in the sediment was significantly and positively correlated to Fe ($r^2 = 0.43$) and TP ($r^2 = 0.75$), while HCl-P was correlated to Pex ($r^2 = 0.68$), TP ($r^2 = 0.28$) and to phytoplankton biovolume ($r^2 = 0.35$), but not to Ca. On average 4% of the surface sediment TP was $NH_4Cl$-extractable, 35% NaOH-extractable and 20% HCl-extractable, with the remaining 41% being residual phosphorus (Res-P). At sediment depths below 50 cm the Res-P fraction was greater (63% of TP), while there was still considerable NaOH-P (21% of TP), thus indicating a significant permanent deposition of iron-bound phosphorus. In most sediments a higher surface Res-P and NaOH-P were recorded as compared to depths below 20–30 cm. The presence of a NaOH-P gradient, even in lakes in which P-lake and anthropogenic impact were low, suggests that post-depositional mobility of phosphorus may occur. The Res-P gradient indicates that a major part of the organically bound phosphorus in the upper 10–20 cm of the sediment eventually is mineralized and dissolved.

*Key words*—shallow lakes, sediment, phosphorus fractionation, phosphorus load, iron

## INTRODUCTION

The importance of lake sediment as both a nutrient sink and a nutrient source has long been recognized. Thus, during the recovery phase following a reduction in external phosphorus loading, excessive internal loading from a phosphorus-rich sediment may prevent or delay improvement in lake water quality for several years (Boström *et al.*, 1985; Marsden, 1989; Jeppesen *et al.*, 1991; Søndergaard *et al.*, 1993).

Fractionation schemes as a method to characterize phosphorus binding to a variety of organic and inorganic sediment components have been widely used (Williams *et al.*, 1971; Hieltjes and Lijklema, 1980; Psenner *et al.*, 1988; Golterman and Booman, 1988). As opposed to, for instance, release experiments, phosphorus fractionation has the advantage of giving a time-integrated expression that is largely independent of seasonal and year to year variation. Although it is an open question which type of sediment phosphorus the different fractionation methods actually measure (Pettersson *et al.*, 1988; Jáuregui and Sánchez, 1993), phosphorus fractionation usually yields operationally defined fractions (Pettersson *et al.*, 1988). However, as the relationships between the various fractions and internal loading are largely unknown, their potential usefulness in predicting sediment–water interactions is presently limited. A first step to make sediment fractionation schemes more operationally useful could be to determine the general relationships between the different phosphorus fractions and other relevant parameters.

The present study is based on the sediment and water characteristics of 32 shallow Danish lakes, the aim being to establish relationships between different phosphorus forms and other sediment or lake characteristics, in particular the factors which determine phosphorus sediment composition in terms of concentration, depth profile and phosphorus fractions. The study was restricted to shallow lakes in which sediment–water nutrient recycling is expected to play a relatively more important role than in deep lakes because of the high sediment surface:water volume ratio.

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## MATERIALS AND METHODS

### *The lakes*

The data used in the following analyses are based on investigations undertaken by local county authorities in 32 shallow lakes (mean depth <5 m) comprising 34 separate lake basins. The lakes represent the various Danish lake types and are all included in the Danish lake monitoring programme set up under the government's Action Plan on the Aquatic Environment in 1987 (Jeppesen *et al.*, 1991; Kronvang *et al.*, 1993). All but two of the lakes are freshwater lakes, the salinity of the other two being 1–9‰.

The lakes are in general relatively small and meso- to hypertrophic. A brief description of their main characteristics is given in Table 1. While the shallowness of most of the lakes means that they are not usually stratified, six of the lakes have a maximum depth >7 m, and temporary stratification may therefore be observed in their deeper parts during calm summer periods.

Calculations of phosphorus loading and mass balance were based on data for 23–25 of the lakes for the period 1989–92 (Windolf *et al.*, in press). Mean inlet phosphorus concentration (P-inlet) and hydraulic load were established for each lake using data from the main inlet (18–26 annual samples, the number depending on the seasonal variation in discharge). Sampling in minor inlets was less frequent and adjusted according to their relative contribution to the total hydraulic and nutrient load. From these values the annual mean phosphorus load, Pex (g P $m^{-2}$ $y^{-1}$), was calculated. Because external loading has reduced or changed during the last decades, many of the lakes were not in a stable state and internal P release from the sediment was significant. Thus in 18 out of 24 lakes, summer mean lake water P concentration (P-lake) was more than 50% higher than the values predicted by the Vollenweider equation (Vollenweider, 1975). The importance of internal loading is further emphasized by the mass balance calculations as these showed that annual net phosphorus retention was negative in nearly 50% of the lakes (Table 1).

Submersed vegetation was absent or sparse in most of the lakes. This was particular so with the 23–25 relatively fast-flushed lakes for which it was possible to establish mass balances.

### *Sampling and analyses*

In 25 of the 34 lakes/lake basins sediment was sampled at three different locations with a Kajak or piston corer (i.d. = 5.2 cm). In the remaining nine lakes, sampling was undertaken at five locations in one lake, at four locations in one lake, at two locations in five lakes and at one location in two lakes. In all but one lake at least the upper 20 cm sediment was sampled and in 31 of the lakes at least the upper 30 cm sediment was sampled. If possible, the cores were sectioned into seven depths (0–2, 2–5, 5–10, 10–20, 20–30, 30–50 and 50–70 cm), each of which was analyzed separately. Thus for most lakes, a total of 21 sediment analyses were made. The sediment was sampled at randomly selected stations located in that area of each lake lying between 70 and 90% on the hypsograph measured from the shore to the deepest part. The analyses from each location and depth were based on a composite sample of three cores. In all but six lakes the sediment was sampled in November or December between 1988 and 1992. Of the remaining six lakes, two were sampled in April and one in August, while three were sampled prior to 1988—one in 1981 and two in 1986 or 1987.

Table 1. Characteristics of the lakes studied ($n = 32$). Twenty-five lakes were used in the hydraulic retention time and P retention calculations. Data are from 1992

| | Mean | 25% | Median | 75% |
|---|---|---|---|---|
| Area ($km^2$) | 2.6 | 0.24 | 0.45 | 1.53 |
| Mean depth (m) | 3.5 | 1.2 | 2.6 | 3.7 |
| Max. depth (m) | 7.7 | 2.6 | 4.5 | 8.6 |
| Hydraulic retention time (y) | 2.0 | 0.1 | 0.5 | 2.5 |
| Summer Secchi depth (m) | 1.4 | 0.5 | 1.0 | 1.8 |
| Total P (mg P $l^{-1}$) | 0.17 | 0.07 | 0.11 | 0.20 |
| P retention (% of load) | 2 | −18 | 2 | 28 |

Sediment was analyzed for dry weight (DW), loss on ignition (LOI), total iron (Fe), total nitrogen (TN), calcium (Ca), total phosphorus (TP), $NH_4Cl$-P, NaOH-P, HCl-P, and residual phosphorus (Res-P) fractioned according to the sequential extraction technique of Hieltjes and Lijklema (1980). Res-P was calculated as the difference between TP and the sum of $NH_4Cl$-P, NaOH-P and HCl-P. As only molybdate reactive phosphorus was determined on the extracts, Res-P also included any dissolved but nonmolybdate reactive organic phosphorus that may have been released during extraction. DW was determined by drying at 105°C for 24 h, and LOI was subsequently determined by drying to constant weight at 550°C. Ca was determined using atomic absorption spectrophotometry and TN by the Kjeldahl method. TP was analyzed spectrophotometrically as molybdate reactive phosphorus after extraction of ash-free sediment with 1 M HCl (modified from Andersen, 1976). Fe was measured spectrophotometrically after bipiridyl complex formation in the same solution. In two of the lakes only TP was determined.

P-lake and phytoplankton biovolume were calculated on the basis of fortnightly sampling from 1 May to 1 October. The P-inlet and P-lake data presented are mean values for the 4-year-period (1989–92, $n = 72$–104). Phytoplankton was fixed in Lugols solution and counted using an inverted microscope until the maximal error on biovolume of the dominant species was less than ±20%. Phytoplankton was determined to genus or species level and biovolume expressed as $mm^3$ $l^{-1}$ calculated by fitting each genus/species to simple geometric figures.

Statistical analyses including regressions were made using SAS (1990). Significance was accepted at the 1% level unless otherwise indicated. When conducting regression analyses between sediment and non-sediment parameters (e.g. depth, Pex, phytoplankton biovolume and P-lake), a mean value of the sediment sampling locations was used. When comparing sediment parameters, all sampling locations were used.

## RESULTS

The sediment surface (0–2 cm) variables analyzed showed marked inter-lake variability (Fig. 1), TP ranging for example from 0.2 to 8.5 mg P $g^{-1}$ DW. The DW of most surface sediments was low, lying between 4 and 10%. LOI ranged from 10 to 40%, median LOI being 25%. Both Fe and Ca differed greatly from lake to lake, Fe ranging from 2 to 150 mg Fe $g^{-1}$ DW and Ca from 0.6 to 296 mg Ca $g^{-1}$ DW. All phosphorus fractions varied 10-fold or more (Fig. 1). Median $NH_4Cl$-P, NaOH-P, HCl-P and Res-P were 0.036, 0.38, 0.32 and 0.64 mg P $g^{-1}$ DW, respectively. Mean Fe:TP ratio was approx. 10 (by weight) in most lakes, with the range being from 2 to 45.

The phosphorus in the surface sediment mainly consisted of NaOH-P and Res-P, which accounted for 35 and 42% of TP, respectively (Fig. 2). The remainder was mainly HCl-P, while $NH_4Cl$-P only constituted a minor part (range <1% to 11%,

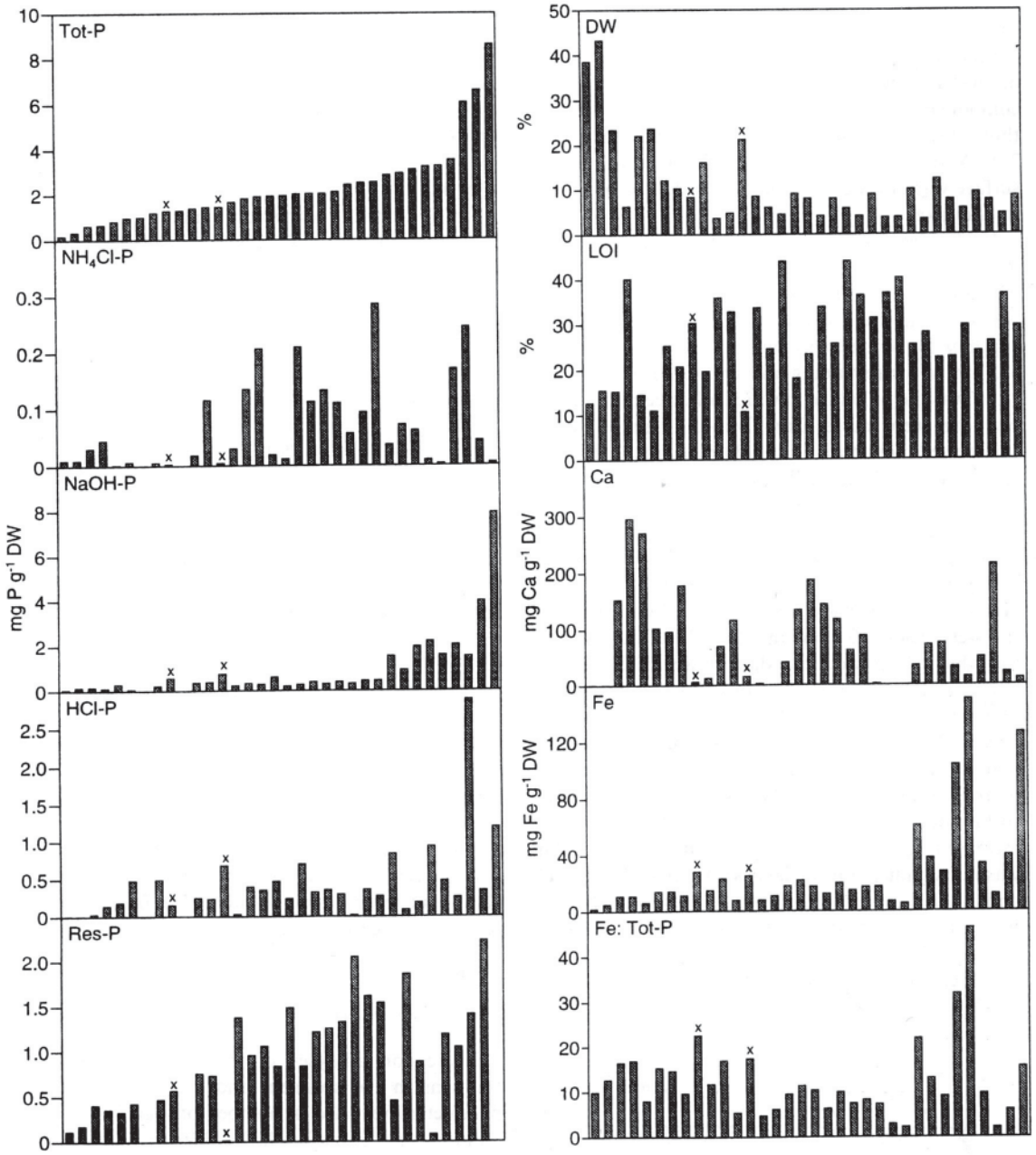



Fig. 1. Surface sediment characteristics of each lake investigated ranked in order of increasing TP. The two brackish lakes are indicated by an x.

average 4%). Composition varied only slightly with increasing depth and was independent of TP concentration. However, in very deep sediment the proportion of Res-P tended to be higher and that of NaOH-P lower. At a depth of 50–70 cm NaOH-P averaged 21% of TP, while Res-P constituted 63%. Fractional composition of the two brackish lakes was not significantly different from that of the freshwater lakes, $NH_4Cl$-P, NaOH-P, HCl-P and Res-P comprising $<1\%$, 42–52%, 12–47% and 1–44%, respectively (Fig. 1).

TP concentration in the surface sediment was higher than that in deeper-lying sediment, independent of whether P-lake was low or high. Thus surface sediment TP in the four lakes in which P-lake was lowest (19–35 $\mu g\ P\ l^{-1}$) was more than twice as high as that at a depth of 20–30 cm (Fig. 3). In these lakes the only nutrient input is atmospheric deposition or diffusive inflow, i.e. there are no surface inlets and no known point-source inputs, either current or previous. The lake catchment areas are mainly non-agricultural, with agricultural land only

accounting for 10–50% of the total catchment area. In each of the four lakes, the higher surface concentrations coincided with a high concentration of Res-P; the latter was the main phosphorus component of the sediment, comprising 60–80% in three of the lakes and 20–40% in the lake with lowest TP. A distinct increase of NaOH-P towards the surface was also observed in two lakes and of HCl-P in one lake.

In the five lakes with the highest P-lake (annual mean ranging from 255 to 1130 $\mu$g P $l^{-1}$) TP increased moderately towards the surface in four of the lakes and markedly, from 0.5 mg P $g^{-1}$ DW at a depth of 15–20 cm to 6.1 mg P $g^{-1}$ DW at a depth of 0–2 cm, in the remaining lake (No. 370) (Fig. 4). The latter was the only lake in which all three phosphorus fractions were more than four-fold higher in the surface sediment than in the deep sediment (Fig. 5). Res-P increased markedly towards the surface in all lakes, while the increase in HCl-P and NaOH-P was small except in lake 370.

In all but one of the remaining 21 lakes for which sediment profiles exist, TP concentrations were higher in the surface sediment (0–2 cm) than in sediment below 20 cm (Fig. 5). In 18 of the 30 lakes for which deep sediment data is available, surface sediment TP was more than twice as high as in deep sediment. Moreover, in 28 out of the 30 lakes, the NaOH-P content of the surface sediment was higher than in the deep sediment, and in 21 of the lakes was more than twice as high in the surface sediment. The surface:deep sediment ratio of both TP and NaOH-P increased with increasing P-lake, but the relationship was not significant if the lake with extremely high TP and NaOH-P ratios (13.1 and 72, respectively) was omitted ($P = 0.10$ and 0.02, respectively). The HCl-P content of surface sediment was higher than that of deep sediment in 21 lakes, the difference being as much as two-fold in 12 of the lakes. Finally, Res-P was higher in the surface than deep sediment of all lakes but four, and in 15 of the lakes was more than twice that at depths of more than 20 cm. The surface sediment Fe concentration was only higher than that of deep sediment in 17 of the 30 lakes, the difference being two-fold in only five cases. There was no correlation between high Fe ratios and P-lake ($P > 0.20$).

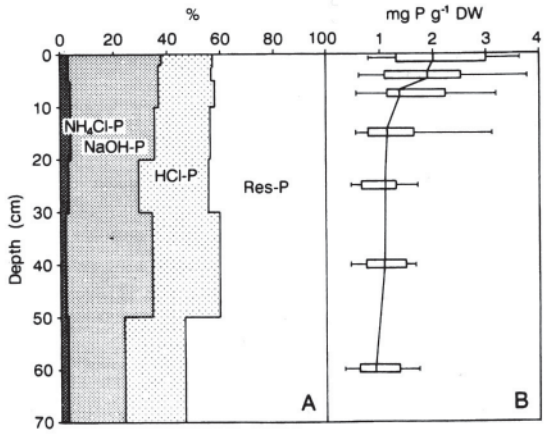



Fig. 2. Phosphorus depth profiles of 32 mainly shallow Danish lakes (30–50 cm, $n = 21$, 50–70 cm, $n = 9$). (A) Mean fractional composition in % of TP. (B) 10, 25, 50, 75 and 90% TP fractiles.

Simple linear regression analysis of surface (0–2 cm) sediment TP (mg P $g^{-1}$ DW) revealed a significant positive correlation to Pex (g P $m^{-2}$ $y^{-1}$)

$$TP = 1.72 + 0.28\ Pex\ (r^2 = 0.46,\ n = 21,\ P < 0.0001), \quad (1)$$

as well as to surface sediment Fe (mg Fe $g^{-1}$ DW):

$$TP = 1.50 + 0.031\ Fe\ (r^2 = 0.36,\ n = 99,\ P < 0.0001) \quad (2)$$

(Table 2 and Fig. 6). The Fe:TP ratio ranged from 1.6 to 50 (by weight).

The coefficient was improved when using both Pex and Fe as dependent variables

$$TP = 0.92 + 0.23\ Pex + 0.031\ Fe\ (r^2 = 0.68,\ n = 21,\ P < 0.0001). \quad (3)$$

Surface sediment TP was also significantly, albeit weakly, related to DW and TN ($r^2 = 0.11$ and 0.09, respectively), but not to any of the other variables, e.g. P-inlet, P-lake or hydraulic retention time (Table 2). However, multiple stepwise regression analysis revealed that LOI (%) and mean depth ($z$) also contributed significantly ($P < 0.01$) and increased the regression coefficient:

$$TP = -2.46 + 0.26\ Pex + 0.024\ Fe + 0.090 \cdot LOI + 0.58\ z\ (r^2 = 0.91,\ n = 21,\ P < 0.0001. \quad (4)$$

Regression analyses of the various phosphorus fractions in the sediment and other non-fractional parameters revealed a number of significant linear relationships (Table 3). NaOH-P (mg P $g^{-1}$ DW) was closely correlated to TP at all depths ($r^2 = 0.75$, Figure 7), and the relationship was improved when restricted to the surface sediment

$$NaOH\text{-}P = -0.89 + 0.80\ TP\ (r^2 = 0.80,\ n = 95,\ P < 0.0001) \quad (5)$$

or to deep (> 20 cm) sediment

$$NaOH\text{-}P = -0.69 + 1.07\ TP\ (r^2 = 0.85,\ n = 178,\ P < 0.0001). \quad (6)$$

NaOH-P was also related to Fe at all depths, although the relationship was less pronounced ($r^2 = 0.43$, Fig. 7). Multiple regression analysis of the relationship between NaOH-P, TP and Fe in

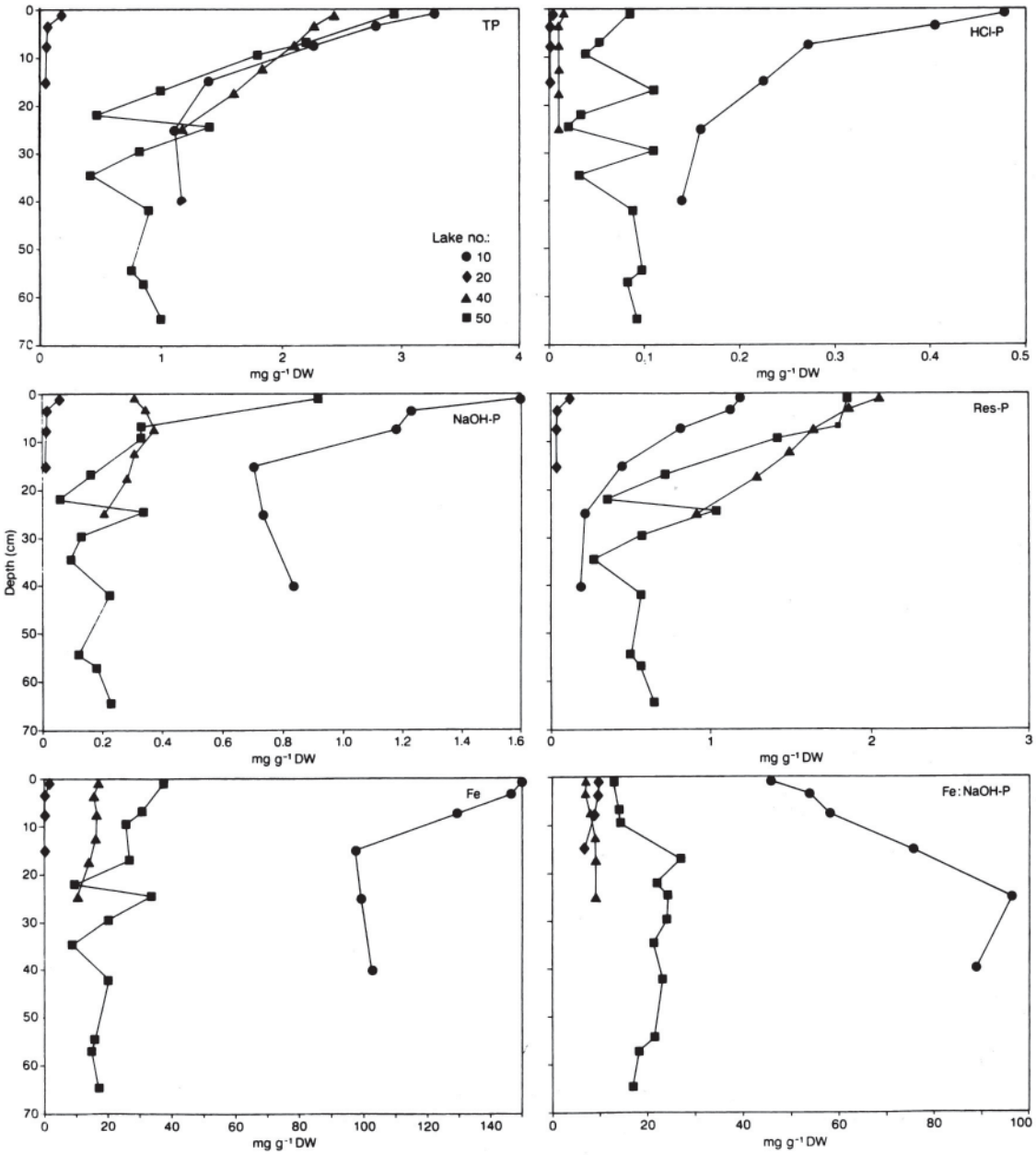



Fig. 3. Mean sediment TP, NaOH-P, Fe, HCl-P, Res-P and Fe:NaOH-P profiles in the four lakes (Nos 10, 20, 40 and 50) in which P-lake was lowest. Mean annual P-lake was 19, 20, 35 and 25 $\mu$g P $l^{-1}$, respectively. The lakes receive no external nutrient input other than atmospheric deposition and diffusive inflow.

surface sediment (0–2 cm) improved the correlation only slightly

$$\text{NaOH-P} = -0.91 + 0.67\,\text{TP} + 0.011\,\text{Fe} \quad (r^2 = 0.83,\ n = 95,\ P < 0.0001). \quad (7)$$

By considering Pex and Fe in a multiple regression, surface NaOH-P could be described as

$$\text{NaOH-P} = -0.37 + 0.041\,\text{Fe} + 0.11\,\text{Pex} \quad (r^2 = 0.77,\ n = 19,\ P < 0.0001). \quad (8)$$

The NaOH-P fraction tended to be relatively higher at high TP [Fig. 7(B)]. Thus while NaOH-P only constituted 10–20% of TP at concentrations below 2–3 mg P $g^{-1}$ DW in the surface sediment, it constituted 40–50% at higher concentrations, and approached 100% in some of the lakes. The same tendency was observed for the NaOH-P and Fe relationship [Fig. 7(A)], i.e. NaOH-P was mainly high (>1 mg NaOH-P mg $g^{-1}$ DW) when Fe was high (>approx. 30 mg Fe $g^{-1}$ DW). The minimum Fe:NaOH-P ratio recorded was 6 (by weight).

HCl-P (mg P $g^{-1}$ DW) correlated to TP

$$\text{HCl-P} = 0.10 + 0.12\ \text{TP}\ (r^2 = 0.28,\ n = 583,\ P < 0.0001) \quad (9)$$

with the regression coefficient increasing to 0.35 if only surface sediment was analyzed (Table 3). HCl-P in the surface sediment was also highly correlated to Pex

$$\text{HCl-P} = 0.21 + 0.10\ \text{Pex}\ (r^2 = 0.68,\ n = 19,\ P < 0.0001) \quad (10)$$

but the relationship was not improved by multiple regression including both Pex and TP.

Surface sediment HCl-P correlated to the mean phytoplankton biovolume (phyt), in particular when excluding blue-greens ($r^2$ = 0.35). A multiple stepwise regression relating HCl-P to Pex, TP and phyt ($mm^3\ l^{-1}$) was only significant for Pex and phyt

$$\text{HCl-P} = -0.17 + 0.10\ \text{Pex} + 0.026\ \text{phyt}\ (r^2 = 0.80,\ n = 19,\ P < 0.0001). \quad (11)$$

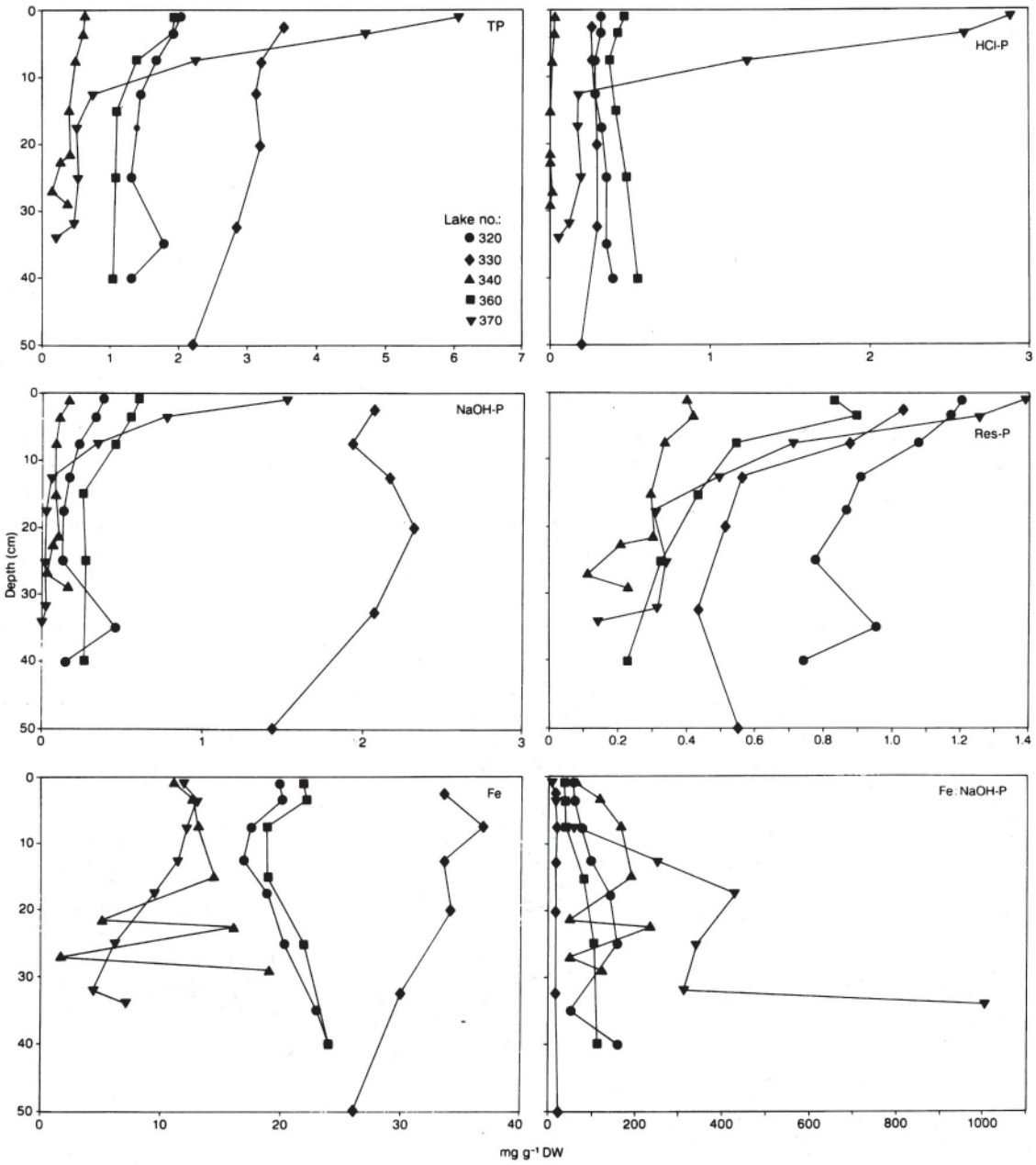



Fig. 4. Mean sediment TP-profiles in the five lakes (Nos 320, 330, 340, 360 and 370) in which lake water P was highest. The mean annual P-lake was 279, 434, 255, 271 and 1131 $\mu g\ P\ l^{-1}$, respectively. Pex ranged from 1.6 to 20.7 g P $m^{-2}\ y^{-1}$.

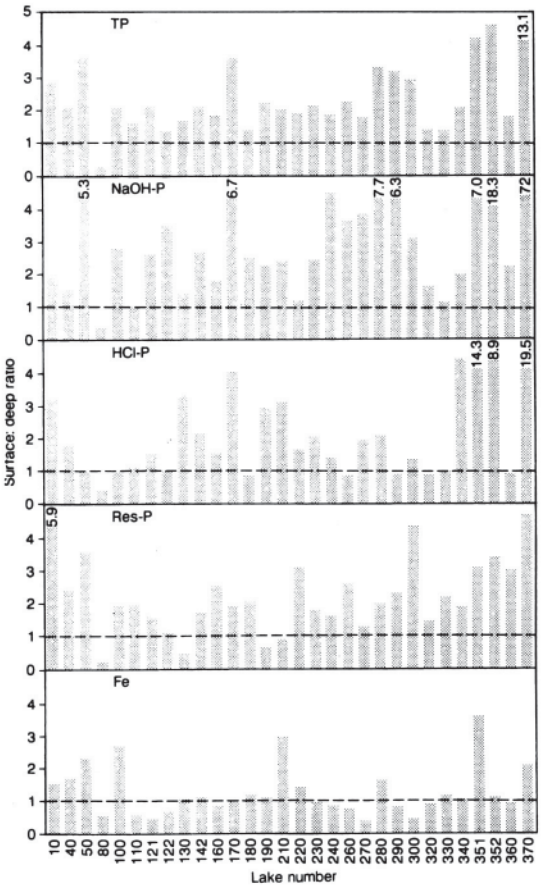



Fig. 5. The ratio between phosphorus fraction concentrations at a depth of 0–2 cm (average of all positions sampled) and below 20 cm (as a mean of all sampling depths and positions). The lakes are ranked in order of increasing P-lake.

The relationship was not improved by excluding blue-greens.

There was no significant correlation between HCl-P and Ca, either in surface or deep sediments ($P > 0.19$).

In the case of $NH_4Cl$-P (mg P $g^{-1}$ DW), surface sediment correlated strongest to LOI

$$NH_4Cl\text{-}P = -0.043 + 0.0043\ LOI\ (r^2 = 0.23,\ n = 93,\ P < 0.0001) \quad (12)$$

Table 2. Linear regression of TP (mg $g^{-1}$ DW) in surface sediment (0–2 cm) and different lake and sediment parameters. * = significant at 5% level, NS = not significant, $n$ = number of data

| | $r^2$ | $n$ |
|---|---|---|
| Mean depth (m) | 0.13* | 34 |
| Pex (g P $m^{-2}$ $y^{-1}$) | 0.46 | 20 |
| Mean inlet P (mg P $l^{-1}$) | NS | 20 |
| Hydraulic retention time (y) | NS | 20 |
| Mean lake P (mg P $l^{-1}$) | NS | 34 |
| Sediment iron (mg Fe $g^{-1}$ DW) | 0.36 | 98 |
| Calcium (mg Ca $g^{-1}$ DW) | NS | 90 |
| Total nitrogen (mg N $g^{-1}$ DW) | 0.09 | 95 |
| Dry weight (% of wet weight) | 0.11 | 98 |
| Loss on ignition (% of dry weight) | 0.06* | 96 |

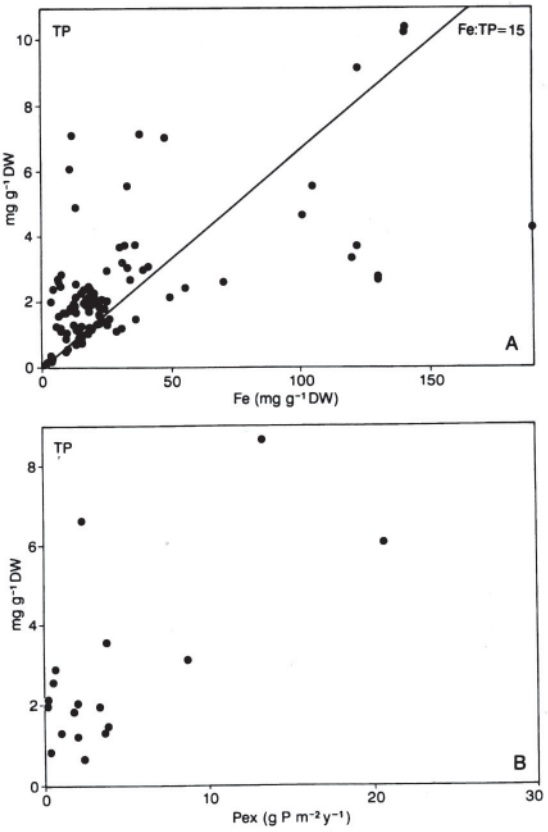



Fig. 6. TP (mg $g^{-1}$ DW) in the surface sediment (0–2 cm) vs: (A) Fe, (total iron in surface sediment, g Fe $g^{-1}$ DW), (TP = 1.50 + 0.031 Fe, $r^2 = 0.36$, $n = 99$); (B) Pex (external load, g P $m^{-2}$ $y^{-1}$), (TP = 1.72 + 0.28 Pex, $r^2 = 0.46$, $n = 21$).

and to phytoplankton (excl. blue-greens) ($r^2 = 0.16$, $P = 0.02$), there being no correlation to TP and only a weak negative correlation to Fe. By including Pex in a multiple regression, $NH_4Cl$-P could be described by the equation

$$NH_4Cl\text{-}P = -0.10 + 0.0064\ LOI + 0.0045\ Pex (r^2 = 0.35,\ n = 19,\ P < 0.0001) \quad (13)$$

Despite the fact that Res-P accounted for the greatest proportion of TP (Fig. 1), no significant relationship between TP and Res-P was detectable in surface sediments and the relationship in sediments deeper than 20 cm was weak and negative ($r^2 = 0.06$). Surface sediment Res-P correlated to LOI ($r^2 = 0.21$), while in deep sediments the correlation was weak ($r^2 = 0.03$, $P = 0.02$).

## DISCUSSION

Even though the data presented in this study cover a relatively broad range of lake types and sediments, the fact that it has been possible to establish relatively strong relationships between sediment phosphorus fractions and a number of parameters suggests the existence of rather general determining factors over a

Table 3. Linear regression ($r^2$) of phosphorus fractions and sediment total phosphorus (TP), iron (Fe), calcium (Ca), loss on ignition (LOI), external load (Pex), mean annual phytoplankton biovolume (phyt) and mean annual phytoplankton biovolume excl. blue-greens (phytn). A negative slope is indicated by −. When including Pex, phyt and phytn $n$ = 19–32

| | TP | Fe | Ca | LOI | Pex | phyt | phytn |
|---|---|---|---|---|---|---|---|
| All sediment depths, $n$ = 528–582 | | | | | | | |
| $NH_4Cl$-P | 0.01 | −0.06 | 0.01* | 0.07 | | | |
| NaOH-P | 0.75 | 0.43 | −0.07 | NS | | | |
| HCl-P | 0.28 | 0.08 | 0.01 | NS | | | |
| Res-P | NS | −0.11 | NS | 0.06 | | | |
| Depth = 0–2 cm, $n$=86–95 | | | | | | | |
| $NH_4Cl$-P | NS | −0.10 | NS | 0.23 | NS | NS | 0.16* |
| NaOH-P | 0.80 | 0.46 | −0.08 | NS | 0.25* | NS | NS |
| HCl-P | 0.35 | 0.12 | NS | NS | 0.68 | 0.22 | 0.35 |
| Res-P | NS | −0.07 | NS | 0.21 | NS | NS | NS |
| Depth > 20 cm, $n$=158–178 | | | | | | | |
| $NH_4Cl$-P | NS | −0.04 | NS | NS | | | |
| NaOH-P | 0.85 | 0.39 | −0.05 | NS | | | |
| HCl-P | 0.15 | 0.04* | NS | −0.03* | | | |
| Res-P | −0.06 | −0.16 | NS | 0.03* | | | |

*Significant at 5% level only.

wide range of environmental conditions. Fe was a key factor in many of the established relationships. Thus sediment TP was strongly and positively related to sediment Fe, this being in concert with the classical iron theory (Einsele, 1936; Mortimer, 1941) and in accordance with more recent investigations, e.g. Ostrofsky (1987), Jeppesen *et al.* (1991), Jensen *et al.* (1992) and Moutin *et al.* (1993). However, despite the fact that redox potential in the deep sediment of the majority of the lakes presented here presumably was very low, these sediments also exhibited a strong relationship between TP and Fe and between Fe and NaOH-P. Moreover, the relative proportion of NaOH-P was more or less independent of depth, averaging 20% of TP at depths below 50 cm. Assuming that NaOH-P represents iron-bound phosphorus, these findings indicate that phosphorus in the deep sediment is still present as a particulate iron phosphorus complex, either protected from reduction by a coating of $FePO_4$, or in the reduced form as vivianite (Golterman, 1984; De Groot, 1991). Although phosphorus has been shown to be released at depths up to 20–23 cm (Søndergaard *et al.*, 1993), the occurrence of NaOH-P in the deeper parts of the sediment (>50 cm), where interaction with lake water should not be possible, suggests that phosphorus can be permanently immobilized in an iron-bound form. Adding iron to sediment following a reduction in external P load may, by increasing phosphorus retention, thus prove to be a valuable restoration measure (Boers *et al.*, 1994; Phillips *et al.*, 1994).

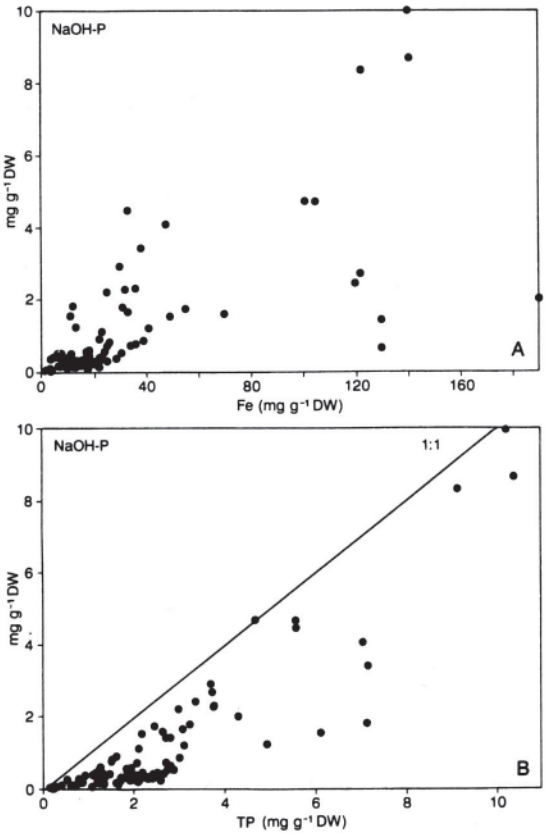



Fig. 7. NaOH-P in the surface sediment (0–2 cm) vs: (A) iron (Fe), (NaOH-P = 0.13 + 0.031 Fe, $r^2$ = 0.46, $n$ = 95); (B) total phosphorus (TP), (NaOH-P = −0.89 + 0.80 TP, $r^2$ = 0.80, $n$ = 95).

The importance of iron with regard to phosphorus retention is supported by the observation that TP concentrations were mainly high (>3 mg P $g^{-1}$ DW) when Fe exceeded 20–30 mg Fe $g^{-1}$ DW (Fig. 6), the only exception being lake 370, which had an extremely high Pex. NaOH-P usually comprised a large proportion of TP at high Fe (Fig. 7) and even at low TP and Fe always comprised at least 10–20% of TP; this could be attributable to very strong adsorption of the first phosphate ions on iron-hydroxide particles (Lijklema, 1976), while the adsorption of subsequent P may be more dependent on concentration and equilibrium conditions. Part of the NaOH-P may also represent phosphorus bound to aluminium and clay minerals.

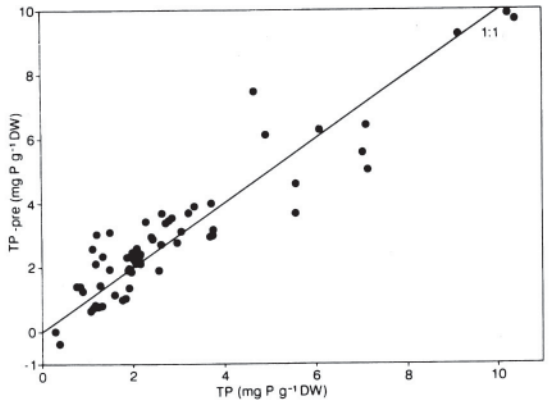



Fig. 8. Measured TP and predicted TP (TP-pre) in surface sediment based on equation (4).

In addition to Fe, TP was highly related to Pex; in combination these two factors explained 68% of the variation in surface sediment phosphorus concentration. This implies that a reduction in Pex will eventually lead to a reduction in TP, provided that Fe loading remains unchanged.

A further implication is that the current Pex may be estimated from measurements of surface sediment TP and Fe - an estimation which can be improved to account for 91% of the variability by including LOI and mean depth (Fig. 8). Thereby past Pex levels may be predicted using sediment dating in combination with measurements of TP and Fe (Dillon and Evans, 1993). However, post-depositional phosphorus mobility in sediment (Carignan and Flett, 1981) raises the question of whether such predictions are always possible, as also indicated by the profiles in the four low P lakes showing higher concentrations in the surface than in the deeper sediment. Because the catchment area of these lakes is partly agricultural, the possibility cannot be excluded that some changes in Pex have occurred during the period represented by the P profiles. However, it seems unlikely that the very distinct P profile found in the four lakes without any known anthropogenic load or past changes in load should be attributed to historical changes in Pex. Recently developed techniques to reconstruct past sediment TP using palaeolimnological records in the sediment, e.g. diatoms (Anderson *et al.*, 1993), might be used in combination with the relationships established here to gain new insight into determining past Pex.

In all four low P lakes the sediment TP gradient was partly due to a distinct Res-P gradient, which does not have to be due to migration of dissolved phosphorus. Thus a Res-P gradient was also found in most of the other lakes studied, Res-P often being 2–6 times higher in the surface than in the deep sediments. This is attributed to contemporary sedimentation of organic bound phosphorus as phytoplankton or macrophytes and its accumulation in the upper sediment. The presence of a Res-P gradient also indicates that only a relatively small organic P fraction (approx. 20–50% of the original organic P content of the surface sediment and probably even less of the sedimenting organic material) will be permanently deposited in shallow lake sediments. Surface sediment Res-P (<20 cm) must thus be regarded as highly dynamic, comprising not only refractory organic phosphorus, but also a major organic fraction which will eventually be decomposed and released.

Two of the four low P lakes also showed a distinct NaOH-P gradient towards the surface sediment (Fig. 3). These were the lakes having the highest Fe and an iron profile resembling that of NaOH-P. In both lakes the Fe:NaOH-P ratio decreased towards the surface, however, thus indicating net post-depositional phosphorus mobility relative to iron.

Many of the lakes investigated, including both low and high P lakes, also had distinct HCl-P profiles. In the high P lakes, which became increasingly eutrophic in the past, this may be related to the increased calcite precipitation (see below) known to accompany increasing eutrophication (Schelske *et al.*, 1988). The HCl-P gradients were frequently accompanied by steep NaOH-P gradients (Fig. 5), and variations in NaOH-P did not seem to be compensated for by variations in HCl-P, as has otherwise been found by Moutin *et al.* (1993). Instead a possible linkage between NaOH-P and HCl-P gradients is the occurrence of high porewater P concentrations in the presence of high sediment NaOH-P concentrations, a combination which may lead to phosphate adsorption by $CaCO_3$ and the formation of apatite or other Ca complexes (Søndergaard *et al.*, 1993).

In contradiction to Fe, there was no correlation between Ca and any of the phosphorus fractions, including HCl-P, at any depth. This is consistent with the findings of others (e.g. Jensen *et al.*, 1992) and confirms that the amount of Ca present in the sediment has no impact on the distribution of the various phosphorus pools (De Groot, 1991). Instead, surface sediment HCl-P seemed highly correlated to mean annual phytoplankton biovolume, particularly when excluding the slowly settling blue-greens. This indicates that the HCl-P sediment phosphorus fraction depends on the input of organic matter; however, as HCl-P did not correlate to LOI, the coupling cannot be direct. A possible explanation may be that HCl-P depends on co-precipitation of phosphate with calcite (Kleiner, 1988; House, 1990; Moutin *et al.*, 1993) and hence is more strongly related to biological processes than usually considered. This is in concert with our finding that surface sediment HCl-P and Pex were strongly correlated. Although it could be argued that calcite-precipitated phosphate would be dissolved at the lower pH in the sediment (Golterman, 1988), the elevation of sediment pH by high photosynthetic activity as observed in hypertrophic lakes may to

some extent counterbalance this effect (Søndergaard, 1988), particularly in shallow areas allowing benthic primary production.

Both Res-P and $NH_4Cl$-P correlated to LOI in the surface sediment, the correlations to Res-P being in agreement with the finding in 66 eastern North American lakes that Res-P may be used as an indicator of the organically bound phosphorus fraction (Ostrofsky, 1987). The correlation between $NH_4Cl$-P and LOI emphasizes that the $NH_4Cl$-P phosphorus fraction is very dynamic and may therefore be a useful indicator of the processes that determine sediment water interactions, despite the fact that it usually constitutes only a few per cent of the TP pool (Pettersson *et al.*, 1988; Søndergaard, 1988; Jensen *et al.*, 1992).

## CONCLUSION

The phosphorus concentration of lake surface sediment depends both on the external P load and on the sediment iron concentration, high sediment P concentrations only being reached when both factors are high.

The always higher concentration of Res-P in surface sediments than in deeper sediment shows that in the upper 10–20 cm of the sediment most organically bound phosphorus is only stored temporarily. NaOH-P seems to be mainly related to chemical parameters (mainly Fe), while the other fractions (HCl-P, $NH_4Cl$-P and Res-P) seem to be more directly coupled to biological parameters.

The presence of NaOH-P also in deep parts of the sediment suggests that phosphorus can be permanently immobilized in an iron-bound form..

That there were only minor changes in the relative contribution of different P fractions at increasing depth suggests that the relationship between the various phosphorus forms is relatively stable despite variations in sediment composition, aging and redox conditions.

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# Eight years of internal phosphorus loading and changes in the sediment phosphorus profile of Lake Søbygaard, Denmark

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*Key words:* phosphorus, retention, sediment, release, fractionation, iron:phosphorus ratio

## Abstract

During each of the first 8 years following an 80–90% reduction in external phosphorus loading of shallow, hypertrophic Lake Søbygaard, Denmark in 1982, phosphorus retention was found to be negative. Phosphorus release mainly occurred from April to October, net retention being close to zero during winter. Net internal phosphorus loading was 8 g P $m^{-2}$ $y^{-1}$ in 1983 and slowly decreased to 2 g P $m^{-2}$ $y^{-1}$ in 1990, mainly because of decreasing sediment phosphorus release during late summer and autumn. The high net release of phosphorus from Lake Søbygaard sediment is attributable to a very high phosphorus concentration and to a high transport rate in the sediment caused by bioturbation and gas ebullition. Sediment phosphorus concentration mainly decreased at a depth of 5 to 20 cm, involving sediment layers down to 23 cm. Maximum sediment phosphorus concentration, which was 11.3 mg P $g^{-1}$ dw at a depth of 14–16 cm in 1985, decreased to 8.6 mg P $g^{-1}$ dw at a depth of 16–18 cm in 1991. Phosphorus fractionation revealed that phosphorus release was accompanied by a decrease in $NH_4Cl$-P + NaOH-P and organic phosphorus fractions. HCl-P increased at all sediment depths. The Fe:P ratio in the superficial layer stabilized at approximately 10. Net phosphorus release can be expected to continue for another decade at the present release rate, before an Fe:P ratio of 10 will be reached in the sediment layers from which phosphorus is now being released.

## Introduction

Recovery of eutrophic lakes following a reduction in external phosphorus loading can be severely delayed by the release of phosphorus from sediment (Marsden, 1989; Sas, 1989; Jeppesen *et al.*, 1991). The duration of net annual sediment release is usually considered to be relatively short, i.e. not more than 5 years (Sas, 1989). However, shallow, highly polluted lakes tend to have a longer recovery period (Boström *et al.*, 1985; Cullen & Forsberg, 1988; Jeppesen *et al.*, 1991), this being mainly attributable to high loading (relative to the lake area) prior to the reduction in external phosphorus loading. Recent results furthermore indicate that the iron phosphorus ratio may play an important role in the control of phosphorus release from sediments, and that iron can suppress phosphorus release when the ratio (by weight) is higher than 15 to 20 (Jensen *et al.*, 1992a).

The present article describes persistent internal loading, expressed as negative phosphorus retention, in shallow, hypertrophic Lake Søbygaard, Denmark. Changes in phosphorus retention during the first 8 years of recovery following an 80–

90% reduction in external phosphorus loading were determined from the monthly mass balance calculated by means of inlet-outlet measurements. In addition we describe how the internal loading of Lake Søbygaard is reflected by changes of sediment phosphorus profile and composition in the upper 30 cm of the sediment. Finally we demonstrate that phosphorus and iron profiles may give valuable information concerning the role of the sediment in future internal loading of the lake.

## Methods

### *Lake description*

Lake Søbygaard is a small (40 ha) and shallow (mean depth 1.0 m) hypertrophic lake situated in the central part of Jutland, Denmark. 80–90% of the total hydraulic loading is derived from one main inlet, the remainder coming from groundwater springs along the lake shore. The lake has two outlets. Mean hydraulic residence time varies from 15–20 days in winter to 25–30 days in summer.

Prior to 1982, when phosphorus reduction was implemented at the sewage plant, Lake Søbygaard received large amounts of poorly treated waste water which caused accumulation of a phosphorus-rich sediment (Søndergaard, 1989; Søndergaard, 1990). From 1976 (when biological treatment of sewage was started) to 1982 most of the phosphorus entering the lake was inorganic. Since 1982, the total phosphorus concentration at the inlet has usually been between 0.1 and 0.3 mg P $l^{-1}$. Outlet phosphorus concentration, which was alike lake water concentration, varied with the season, winter concentration being roughly the same as inlet concentration, and mean summer concentration being 0.5 to 1.0 mg $l^{-1}$ (Jeppesen *et al.*, 1990; Søndergaard *et al.*, 1990). Mean summer chlorophyll *a* concentration, which was about 800 $\mu g\, l^{-1}$ in 1984 and 1985, has declined, and has been 200–300 $\mu g\, l^{-1}$ since 1987 (see also Jeppesen *et al.*, 1990; Jeppesen *et al.*, 1992).

### *Mass balance*

Hydraulic flow in main inlet and outlet was determined from stage discharge rating curves. Stage was measured continuously and discharge by a propeller as a minimum with monthly intervals. Daily flow rates were calculated.

Phosphorus concentration in the inlet and outlet was determined weekly or biweekly in 1983 and 1984, and by automatic sampling every 6 hours since 1985. Point samples from all the main springs along the lake shore, which represent approx. 20% of the hydraulic loading, were collected 2–15 times during the investigation. The latter displayed low variability in phosphorus concentration (mean phosphorus loading was 0.28 g P $m^{-2}\, y^{-1}$), and were included in the mass balance calculations. The tot-P concentration of inlet and outlet water was analyzed as molybdate reactive phosphorus following persulphate digestion (Koroleff, 1970).

Total phosphorus input and output were calculated by linear interpolation of the measured concentrations between sampling dates and by multiplying the concentration by the daily discharge volume. This was done on an yearly or quarterly basis.

### *Sediment*

Data concerning sediment in 1985 are based on samples collected on 6 occasions during the period July 1985 to February 1986. Subsequently sediment was only sampled once every two years: in March 1987, in September 1989 and in April 1991. The sediment cores were sampled with a Kajak bottom sampler (Kajak *et al.*, 1965) or a modification of this (piston-corer). Inner core diameter was 52 mm and core length 50 cm. Sampling was each time conducted at the same position ($\pm$ 20 m) in the center of the lake.

Each analysis was made on a composite sample of three cores sectioned into 1 or 2 cm thick slices for determination of the sediment profiles. Dry weight (DW) was determined by drying at 105 °C for 24 h, and loss on ignition (LI) was

subsequently determined by drying to constant weight at 550 °C. Total phosphorus (tot-P) was analyzed spectrophotometrically as molybdate reactive phosphorus after extraction of ash-free sediment with 1 M HCl (modified from Andersen, 1976). Total iron (tot-Fe) was measured after bipiridyl complex formation in the same solution. Phosphorus fractionation, carried out (with minor modifications) according to the method of Hieltjes & Lijklema (1980), gave the following fractions: $NH_4Cl$-P, NaOH-P and HCl-P. Residual phosphorus is expressed both as org-P (determined by analysis of total phosphorus in the sediment remaining after the HCl extraction) and res-P (calculated as tot-P ($NH_4Cl$-P + NaOH-P + HCl-P). Org-P is considered to represent refractory organic bound phosphorus, while res-P includes easily dissolvable organic phosphorus that is released during the preceding extractions. Sediment phosphorus fractionation was only carried out in 1985 and 1991.

As Lake Søbygaard is characterized by a distinct seasonal variation in $NH_4Cl$-P in the upper sediment (Søndergaard, 1988), and as the sediment cores were sampled at different times of the year, $NH_4Cl$-P has been included in the NaOH-fraction when presenting the data.

Volumetric phosphorus concentration was calculated by means of tot-P, DW and LI, assuming an inorganic matter density of 2.6 g $cm^{-3}$ and an organic matter density of 1.05 g $cm^{-3}$.

Sediment profiles from 1985, 1987 and 1989 were adjusted, so as to take into account a net sediment increase of 0.6 cm $y^{-1}$. Sediment profiles shown have thus been adjusted supposing that the level of the sediment surface was zero in 1991; accordingly, the 1985 sediment surface is located at 3 cm on figures. The net sedimentation rate was calculated on the basis of a mean iron retention since 1984 of 36 g Fe $m^{-2}$ $y^{-1}$ (S.D. = 7.4); a mean iron concentration of 60 mg Fe $g^{-1}$ dw and a mean DW of 10% in the upper 20 cm of the sediment. Due to compaction, however, especially the uppermost 5 to 10 cm of the sediment are believed to undergo changes in DW over time. This complicates the direct comparison of sediment profiles of different years in the surface sediment. Changes in sediment DW below 20 cm depth are believed to have been negligible during the period from 1985 to 1991.

## Results

Towards the end of 1982 external phosphorus loading of Lake Søbygaard was reduced from about 30 g P $m^{-2}$ $y^{-1}$ to less than 5 g P $m^{-2}$ $y^{-1}$ (Fig. 1). Phosphorus retention had been high (and positive) for decades prior to this reduction; afterwards it immediately became negative due to sediment phosphorus release, and has remained so, although the amount released has gradually decreased (Fig. 1). During the last 4 years retention has largely remained unchanged between −2

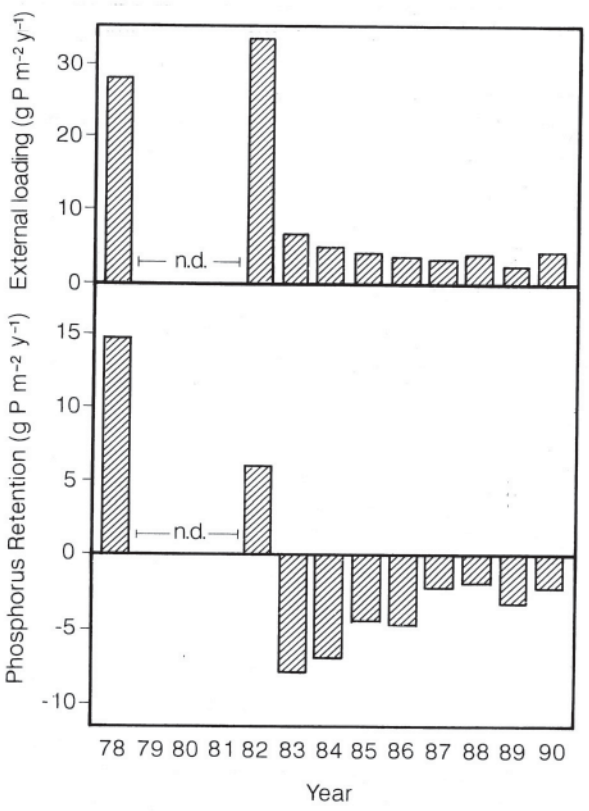



*Fig. 1.* External phosphorus loading (upper) and phosphorus retention (lower) in Lake Søbygaard from 1978 to 1990 (g P $m^{-2}$ $y^{-1}$) Measures to severely reduce external phosphorus loading were implemented from the end of 1982. Data from 1978 are from Aarhus County (1979). n.d. = no data.

and $-3.3$ g P $m^{-2}$ $y^{-1}$, which corresponds to a net annual phosphorus release from the lake of 0.8 to 1.3 tons phosphorus.

Quarterly phosphorus retention was usually close to zero during the 4th and 1st quarters (i.e. autumn/winter), whereas during the 2nd and 3rd quarters (i.e. spring/summer) retention was highly negative, varying between -0.7 and -2.7 g P $m^{-2}$ $quarter^{-1}$ (Fig. 2). This corresponds to a net internal phosphorus loading of 8–30 mg P $m^{-2}$ $d^{-1}$.

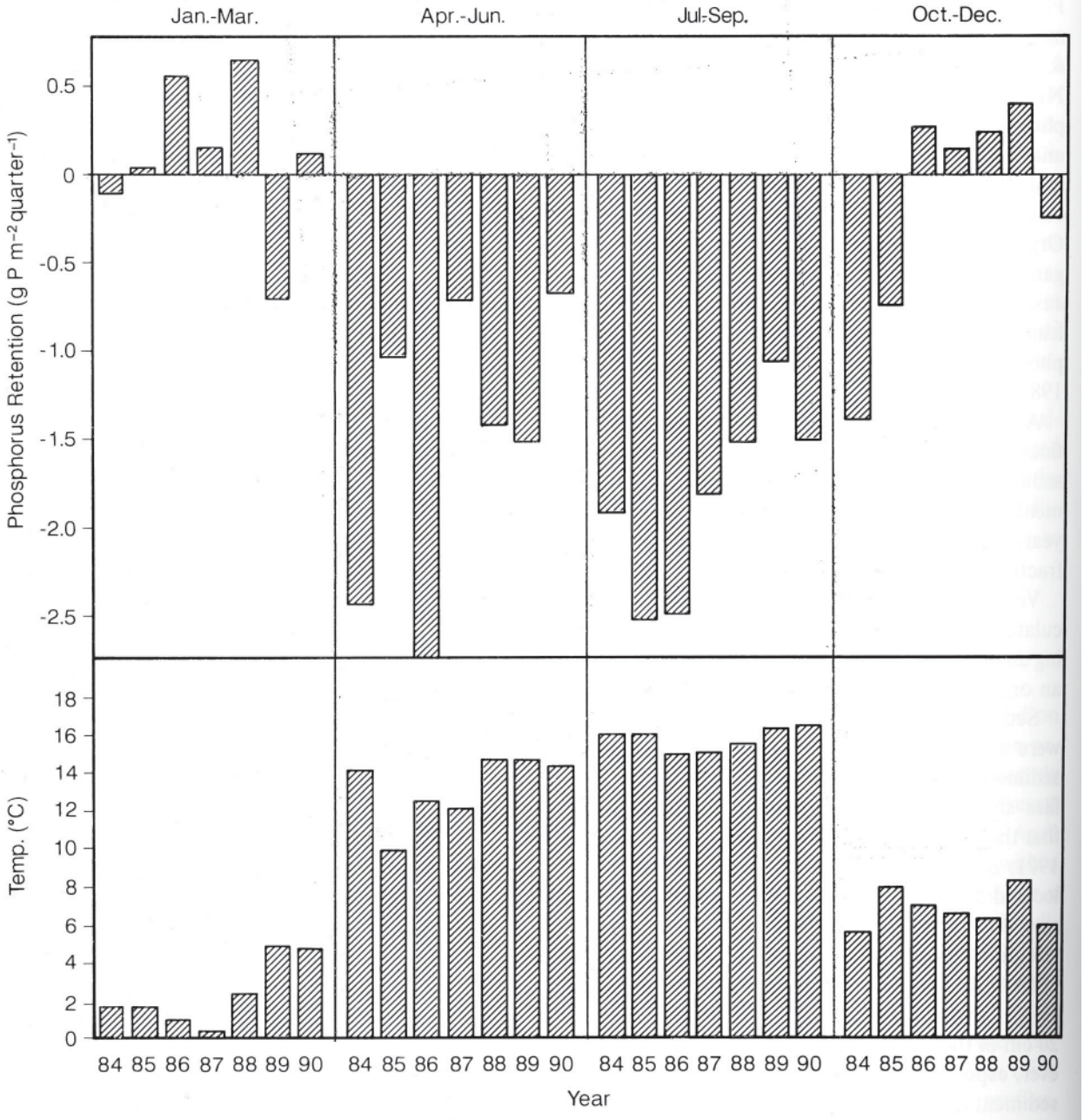



*Fig. 2.* Quarterly variation in phosphorus retention in Lake Søbygaard from 1984 to 1990 (g P $m^{-2}$ $quarter^{-1}$). The calculations are based on mass balances corrected for changes in the phosphorus pool accumulated in the lake water.

In 1984 and 1985 retention was also markedly negative during the 4th quarter. Inter-annual variation in retention was most pronounced in the 2nd quarter from April to June. Retention in the 3rd quarter has tended to become less negative since 1986 ($p < 0.02$). Inter-annual variation in quarterly mean lake water temperature was up to 5 °C (Fig. 2). Apart from general seasonal changes from winter to summer, no clear relationship could be established between different temperature and different net phosphorus retention recognized within the quarters.

Sediment dry weight (DW) and loss on ignition (LI) changed only slightly during the period 1985 to 1991 when comparing corresponding sediment depths (relative to sediment surface). DW was

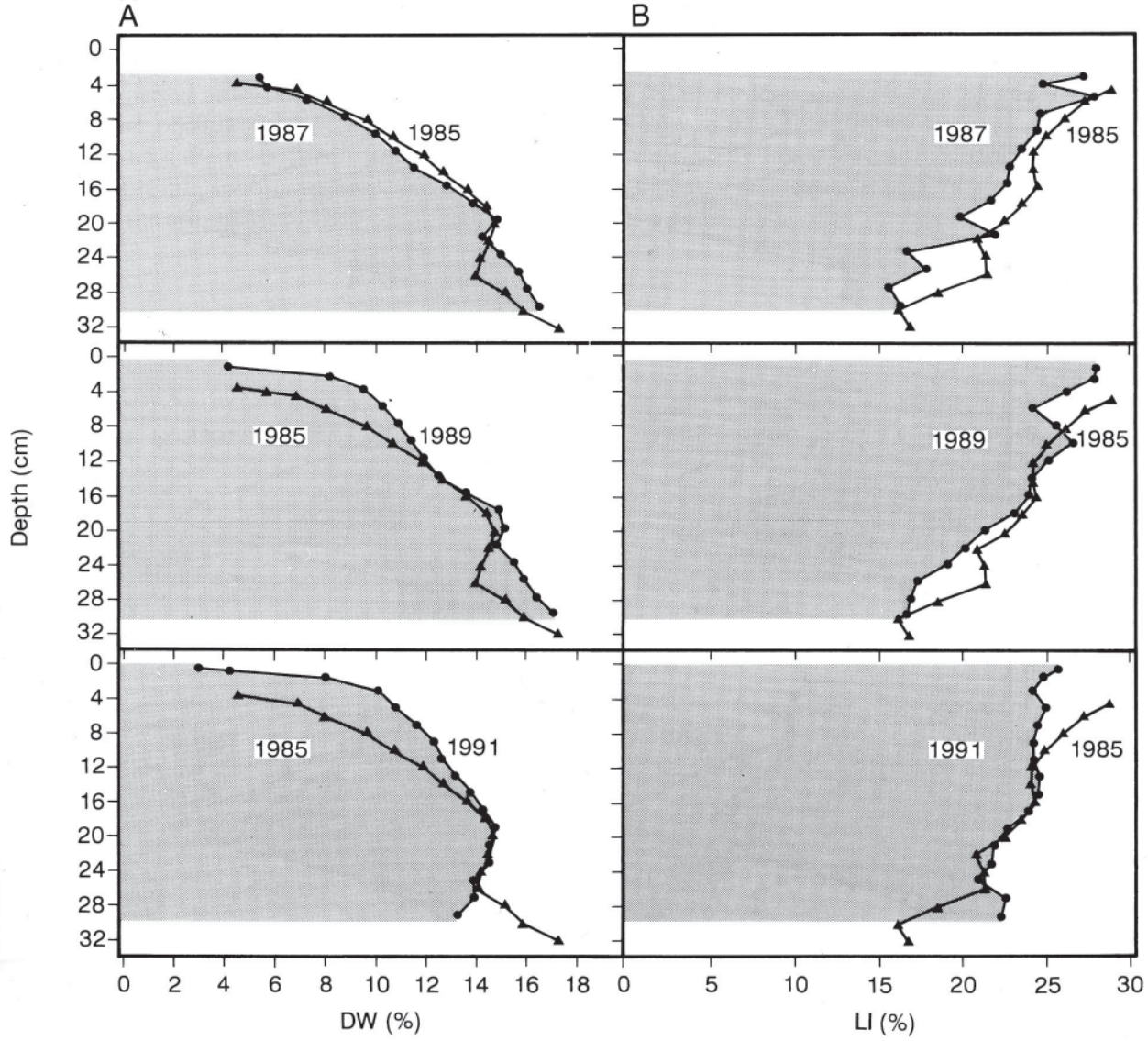



*Fig. 3.* Profiles of Lake Søbygaard sediment in 1985, 1987, 1989, and 1991 of A: dry weight (DW) and B: loss on ignition (LI). All profiles have been adjusted relative to the surface sediment being zero in 1991 using a net sediment increase of 0.6 cm $y^{-1}$. 1985 surface is 3 cm.

3–5% in the superficial sediment, increasing to 15% at a depth of 20 cm. LI was 24–27% in the superficial sediment, decreasing to 20% at a depth of 20 cm (Fig. 3). Tot-Fe was almost homogeneously distributed throughout the sediment, the concentration ranging from 58–65 mg Fe $g^{-1}$ dw in the upper 30 cm of the sediment in 1991, and from 50–70 mg Fe $g^{-1}$ dw in 1985.

Sediment tot-P per g dry sediment decreased markedly during the period 1985 to 1991 (Fig. 4), the decrease occurring in surface sediment and at depths from 10 to 23 cm, but in particular at depths from 12 to 20 cm (sediment surface = 0 cm in 1991). Maximum tot-P, which was 11.4 mg P $g^{-1}$ dw at a depth of 16–20 cm in 1985 decreased to 10.0 mg P $g^{-1}$ dw in 1987, and to 8.5 mg P $g^{-1}$ dw in 1989 and 1991 (Fig. 4A). This corresponds to a reduction in volumetric tot-P from 1.9 to 1.5 mg P $cm^{-3}$ between 1985 and 1991 (Fig. 4B). At sediment depths from

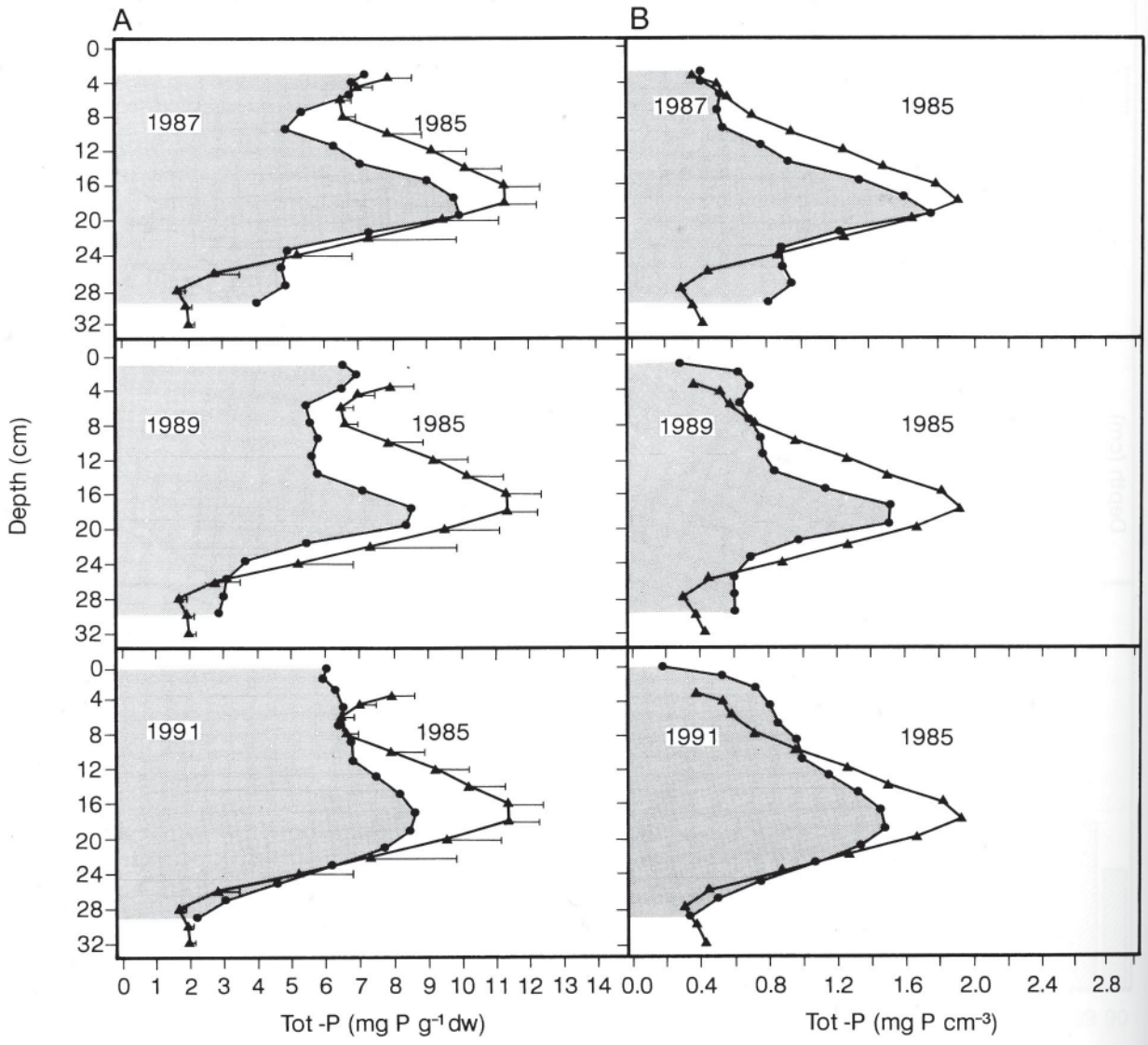



*Fig. 4.* Profiles of Lake Søbygaard sediment in 1985, 1987, 1989, and 1991 of total phosphorus concentration. A: on a dry weight basis (mg P $g^{-1}$ dw) and B: on a volume basis (mg P $cm^{-3}$). 1985 error bars indicate 1 SD of mean value of 6 samples. Sediment profiles adjusted as in Fig. 3.

0–10 cm the volumetric tot-P content increased between 1985 and 1991, especially in the newly settled surface sediment (Fig. 4B). Maximum tot-P in the surface sediment decreased from 7.8 mg P $g^{-1}$ dw in 1985 to 6.1 mg P $g^{-1}$ dw in 1991.

According to an almost unchanged and distinct phosphorus gradient from approx. 23 to 26 cm depth, the upper 23 cm of the sediment seems to represent the sediment part subject to changes in phosphorus concentration. This part contained a total of 242 g P $m^{-2}$ in 1985 that was reduced by 3 g P $m^{-2}$ to 239 g P $m^{-2}$ in 1991. In the same period inlet-outlet measurements revealed a net phosphorus release from the sediment of 14.5 g P $m^{-2}$. Inconsistency between measured net phosphorus release and recorded changes in the sediment concentrations should be attributed to local sediment variations and to difficulties in obtaining exact sediment profiles.

Changes in tot-P are primarily reflected in a decrease in $NH_4Cl$ + NaOH-P, which constituted between 60 and 80% of tot-P in 1985, and in the organic fractions. From 1985 to 1991 $NH_4Cl$ + NaOH-P decreased from a maximum of 9.2 mg P $g^{-1}$ dw to 7.2 mg P $g^{-1}$ dw (Fig. 5). Both org-P and res-P decreased at all sediment depths, but especially so in the superficial layer in which org-P decreased from 0.8 to 0.5 mg P $g^{-1}$ dw and res-P decreased from 2.4 to 1.1 mg P $g^{-1}$ dw between 1985 and 1991. HCl-P increased at all sediment depths by between 0.1 and 0.4 mg P $g^{-1}$ dw in the same period.

Calcium is homogeneously distributed in the sediment with a concentration of approx. 150 mg Ca $g^{-1}$ dw.

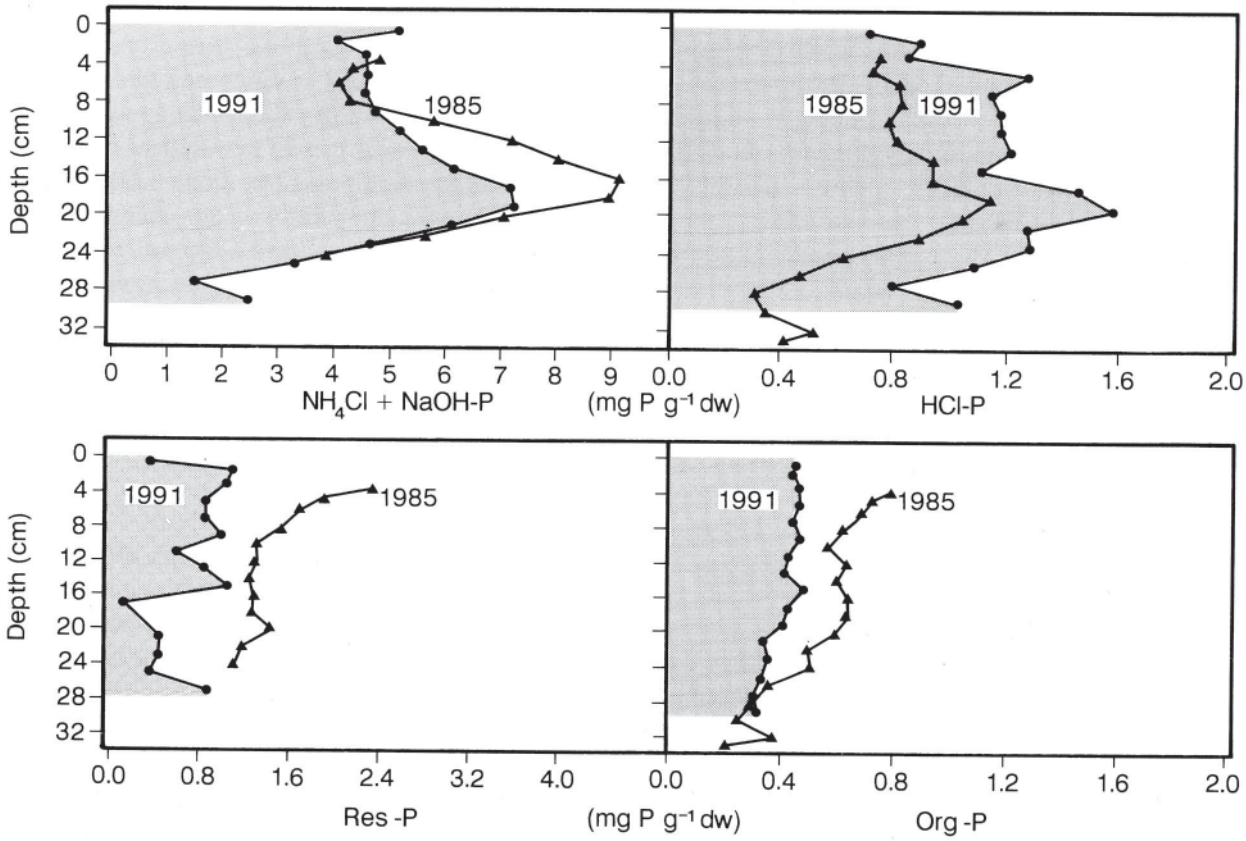



*Fig. 5.* Profiles of Lake Søbygaard sediment showing changes in $NH_4Cl$ + NaOH-P, HCl-P, org-P and res-P (calculated as tot-P – ($NH_4Cl$-P + NaOH-P + HCl-P)) from 1985 to 1991. Sediment profiles adjusted as in Fig. 3.

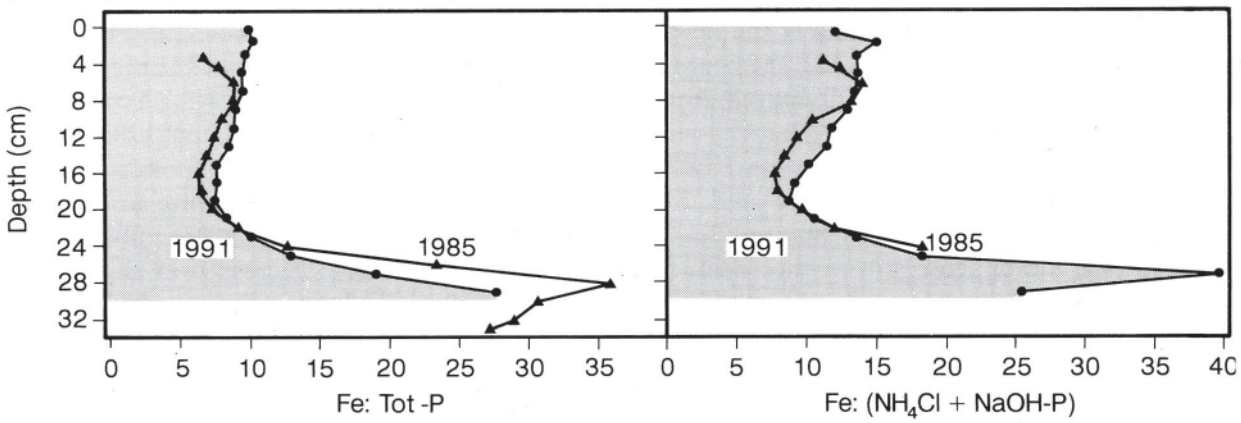



*Fig. 6.* Profiles of Lake Søbygaard sediment showing iron phosphorus ratio (Fe:P) and iron $NH_4Cl + NaOH$-P ratio (Fe:$NH_4Cl + NaOH$-P) by weight in the upper 30 cm in 1985 and 1991. Sediment profiles adjusted as in Fig. 3.

While the Fe:tot-P ratio remained relatively constant from 6 to 10 cm at a level of approx. 10 (by weight), the ratio in the surface sediment and at sediment depths from 10 to 22 cm increased. A minimum of 6.4 at 16 cm depth in 1985 increased to 7.6 in 1991 (Fig. 6). In the deep sediment layers (>28 cm), the Fe:tot-P ratio exceeded 25. The minimum Fe:$NH_4Cl + NaOH$-P ratio increased from 8.0 in 1985 to 9.0 in 1991, but remained at about 13 in the upper 8 cm of sediment. Mean (1984 to 1990) Fe:P ratio in the inlet to the lake was 12.6 (S.D. = 2.0).

The standard deviation of the sediment data generally remained within 10%, but was higher at sediment depths where the P-gradient was steep (due to local variation at the sampling site and minor inaccuracies during sampling and slicing of sediment).

## Discussion

In contrast to most other lakes that have been reported to suffer from internal phosphorus loading (Sas, 1989), P-retention in Lake Søbygaard continues to remain negative 8 years after a major decrease in external phosphorus loading (Fig. 1). The basic reason for the persistence of internal loading is the large phosphorus pool that has accumulated in the lake sediment as a consequence of previously excessive and prolonged external phosphorus loading, and concomitant high iron loading. The high phosphorus affinity of iron explains why, when comparing lakes with the same phosphorus loading, those with iron rich-sediments generally have the highest sediment phosphorus concentration (Jeppesen *et al.*, 1991). Lake Søbygaard thus seems to reflect the general pattern that rapidly flushed Danish lakes, which due to a large phosphorus pool accumulated in the sediment, seem to have a longer recovery period than slowly flushed lakes, even when recovery is expressed in terms of hydraulic residence time (Jeppesen *et al.*, 1991).

The net summer phosphorus release from the sediment of Lake Søbygaard is the net result of a gross release rate of 100–200 mg P $m^{-2}$ $d^{-1}$ that is more or less counterbalanced by a sedimentation rate of 100–150 mg P $m^{-2}$ $d^{-1}$ (Søndergaard, 1989; Søndergaard *et al.*, 1990). Thus, net phosphorus release, which was found to be 8–30 mg P $m^{-2}$ $d^{-1}$ on a quarterly basis during summer in the present study, would be expected to be very sensitive to changes in sedimentation rate. This has been confirmed by the finding that during periods of phytoplankton collapse and subsequently low sedimentation rate, net release increases to 100–200 mg P $m^{-2}$ $d^{-1}$ due to a sustained high gross release rate (Søndergaard *et al.*, 1990).

Although the composition of the phytoplankton community changed towards dominance by larger species during the study period, thus increasing the species-specific sedimentation rate, there was a simultaneous 4-fold decrease in primary production and enhanced algal grazing by zooplankton (Jeppesen *et al.*, 1992a). Thus, the overall sedimentation rate probably did not increase and hence would not have influenced net phosphorus release. The effect of reduced sedimentation is most likely restricted to the relatively short-lasting periods of phytoplankton collapse. The decrease in yearly phosphorus release from Lake Søbygaard sediment since 1983 is therefore unlikely to be attributable to an increase in sedimentation, although the differences in duration and frequency of phytoplankton collapse may account for some of the inter-annual variability.

Instead, the decreasing net phosphorus release in Lake Søbygaard must be due to a decrease in the gross release from sediment that can be expected to occur as the pool of releasable phosphorus in the upper sediment becomes exhausted, and the distance to the lake water increases. The reduction in sediment phosphorus concentration (Fig. 4) indicates that phosphorus was mainly released from the upper 6–18 cm in the period 1985 to 1987, whereas deeper layers down to 23 cm were included from 1989 to 1991, in turn indicating that the increasing transport distance may be partly responsible for the decreasing net release of phosphorus from the sediment.

*Factors influencing phosphorus release from sediment*

Phosphorus release from lake sediments is facilitated by a variety of mechanisms (Boström *et al.*, 1988; Marsden, 1989), some of which are discussed below. The distinct seasonal variation in P-retention in Lake Søbygaard, which was highly negative in spring/summer and approximately zero in autumn/winter, implies that phosphorus release from the sediment is strongly associated with biological activity, either directly through mineralization, or indirectly through an increase in upwards transport. Mineralization processes are believed to be crucial for nutrient cycling especially in the upper sediment layers, to which large amounts of sedimenting algae are constantly being added (Jensen & Kristensen, 1987). Changes in the fractionation profile i.e. the decrease in organically bound phosphorus in the surface sediment emphasize the importance of mineralization. However, the net phosphorus release is probably to some extent only indirectly coupled to mineralization in the deeper parts of the sediment since up to 85% of the decrease recorded in total phosphorus could be referred to a decrease in the inorganic NaOH-P fraction. In contrast, the reduction of org-P only accounts for 10–20% of phosphorus lost from the sediment at a depth of 10–20 cm, i.e. from sediment layers in which the reduction in phosphorus is greatest. These indirect effects can probably be ascribed to bioturbation and gas ebullition, both of which are known to enhance phosphorus release (Skogheim & Erlandsen, 1979; Wisniewski & Planter, 1985; Löfgren & Boström, 1989). The existence of such an indirect, biologically-mediated transport mechanism is supported by the finding that simple chemical diffusion driven solely by a phosphate gradient is not rapid enough to explain the high gross release rates recorded from deeper parts of Lake Søbygaard sediment (Søndergaard, 1990).

Sulphate reduction and subsequent $H_2S$ production lead to inactivation of iron through the formation of FeS and $FeS_2$, thereby disabling or reducing its phosphorus sorption capacity (Schindler, 1985; Boström *et al.*, 1988). Sulphate reduction activity was not measured in the present study. But a distinct summer sulphate gradient in the porewater from 70 mg $SO_4$ $l^{-1}$ in the surface sediment to 5 mg $SO_4$ $l^{-1}$ a depth of 20 cm, and an almost complete absence ($<0.3$ mg $l^{-1}$) of $Fe^{+2}$ in the porewater in the upper 12 cm of the sediment due possibly to the precipitation of iron sulfide (Søndergaard, 1990), indicate that sulphate reduction might be important. The coupling between sulphate reduction and phosphorus release would be expected to be of particular importance in sediments like Søbygaard that have a low Fe:P ratio and a large proportion of iron-bound phosphorus.

The presence of nitrate can both enhance and reduce phosphorus release from sediments due to either increased decomposition through denitrification or improved redox conditions in the presence of nitrate (Boström *et al.*, 1982). The summer period during which the nitrate concentration of Lake Søbygaard was less than 1 mg $NO_3$ $l^{-1}$ increased from 2.5–3.0 months in 1984–1986 to 4.7–5.6 months in 1987–1990 (unpublished observation), probably as a result of a 36% decrease in the external nitrogen loading in August 1987 (Jensen *et al.*, 1992b). Although it is difficult to assess the effect of this change in nitrate concentration, it seems that nitrate depletion does not affect phosphorus release.

Finally, resuspension and high pH are also able to enhance sediment release rates (Boström *et al.*, 1982; Hamilton & Mitchell, 1988; Søndergaard, 1991). However, the influence of these two factors in Lake Søbygaard seems to be restricted to the upper 5 to 10 cm of the sediment. Thus, pH has been shown to increase to between 9 and 10 during summer in the upper 5 cm of the sediment, while wind-induced resuspension seems to provoke major changes in porewater profiles down to 6–8 cm depth (Søndergaard, 1990). Furthermore, pH and resuspension are mainly believed to be important at the beginning of the spring/summer season. At this time phosphorus derived from the lake water and deeper parts of the sediment, and which has been retained in a relatively thick oxidized sediment layer during the winter period, is released when the thickness of this layer is reduced by increasing decomposition. The phosphorus release pattern, expressed in terms of mass balance, can thus be interpreted as being determined by two events: early seasonal release from the superficial sediment which does not lead to any inter-annual differences in the superficial sediment phosphorus concentration, and late seasonal release which can be attributed to net phosphorus transport from deeper sediment layers. This would explain why phosphorus release in the 2nd quarter has not tended to decrease since 1983, whereas the release rate in the 3rd quarter has been diminishing.

*Future perspectives*

Taking into account that the sediment profiles shown in Figs 3–6 represent only one location in the lake, and that the amount of phosphorus accumulated in the sediment (240 g P $m^{-2}$ in the upper 23 cm) is much higher than the net release of phosphorus (2–8 g P $m^{-2}$ $y^{-1}$), then the sediment P-budget is in reasonable agreement with the mass balance calculated from inlet-outlet measurements. We thus consider it justifiable to make use of the changes in P-profile from 1985 to 1991 to clarify the overall transport and mobility of phosphorus in the sediment.

In comparison with most other investigations (Shapiro *et al.*, 1971; Tessenow, 1972; Boström, 1982), the sediment layers subject to changes in phosphorus concentration in Lake Søbygaard are very deep. This is probably due to the very loose nature of the sediment, and the presence of easily-degradable organic material in deeper parts of the sediment, which enables decomposition to enhance phosphorus release and transport. Most of the phosphorus released seems to be derived from the $NH_4Cl$ + NaOH-P and organic fractions.

That there was considerable organic decomposition is supported by the fact that organically bound phosphorus (both org-P and res-P) decreased at all sediment depths during the period 1985-1991. The decrease was most pronounced in the superficial sediment which would be expected to contain the largest amount of easily-degradable organic material. However, part of the decrease in organically bound phosphorus in the surface sediment may have resulted from seasonal variation in the organic P fraction (Boström *et al.*, 1985). This may cause an artifact since most of the 1985 samples were collected during phytoplankton growth season, whereas the 1991 sample was collected in February. Org-P had also decreased, which indicates that this fraction cannot be totally considered as refractory and non-releasable.

HCl-P increased at all sediment depths between 1985 and 1991, hence confirming the assumption that HCl-P is generally immobile, and

suggesting the occurrence of diagenetic and transformation processes which involve HCl-P. A possible explanation is that the continuously high porewater phosphate concentration, combined with a high calcium content, leads to steady adsorption of phosphate by $CaCO_3$, and the formation of apatite or other Ca complexes. This would in turn lead to a general increase in the calcium-bound phosphorus fraction at increasing sediment depth as a consequence of early diagenetic processes (Boström *et al.*, 1988). In addition, increased pH, such as the high levels found in the surface sediment of Lake Søbygaard (Søndergaard, 1990), may also stimulate apatite formation (Stumm & Leckie, 1971).

That the very deep sediment layers take part in the sediment–water interactions is important when considering the future role of the sediment as a net source of phosphorus. In this respect the sediment iron:phosphorus ratio may be important in the control of phosphorus release (Jensen *et al.*, 1992a).

The Fe:tot-P ratio in the upper 6 to 10 cm of sediment changed only slightly between 1985 and 1991. If this stability is interpreted as reflecting a steady-state condition in which the Fe:tot-P ratio is such that no further phosphorus release can take place, then the Fe:tot-P ratio in deeper parts of the sediment would have to increase to 9–10 (by weight) before phosphorus release would cease. This is about the same as the ratio in the uppermost 3 cm of the sediment, which has been formed during the last 5 years. For the sediment at a depth of 7 to 22 cm to reach a Fe:tot-P ratio of 9.6, a total of 26 g P $m^{-2}$ would have to be released (Fig. 4). Thus, as the net release rate in recent years has been 2–3 g P $m^{-2}$ $y^{-1}$, it can be concluded that at the present release rate, the sediment contains enough phosphorus to support net release for another 10 years or so. The total duration of net internal phosphorus loading in Lake Søbygaard would then be approximately 20 years. The steady decrease in annual net release, which is probably due to the increased transportation distance, has not been taken into account.

## Conclusion

Phosphorus mass balance and sediment profiles indicate that the net internal phosphorus loading in Lake Søbygaard, at present continuing for the 9th year, is likely to continue for another decade. The decrease in net phosphorus release from the sediment since cessation of external loading is mainly attributable to a decrease in the gross release rate caused by an increase in the transport distance within the sediment. Indirect effects of biological activity, such as bioturbation and gas ebullition, are believed to be the main factors controlling the rate of net phosphorus release. Net phosphorus release mainly occurs at a sediment depth of 10–23 cm. However, decomposition in the superficial sediment causes a high flux of phosphorus from sediment to water i.e. a high gross release, and occasionally, when sedimentation rate is low, a high net release. The majority of the phosphorus lost from the sediment was derived from the NaOH-P fraction, and to some extent, from the organically-bound fraction. The HCl-P fraction increased in all sediment depths during the years of internal phosphorus loading.

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# Phosphorus release from resuspended sediment in the shallow and wind-exposed Lake Arresø, Denmark

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*Key words:* shallow lakes, resuspension, internal P-loading, equilibrium concentration

## Abstract

Wind-induced sediment resuspension occurs frequently in the shallow and eutrophic Lake Arresø, Denmark. The impact of resuspension on internal phosphorus loading was investigated by laboratory experiments studying P-release from the undisturbed sediment surface and by experiments simulating resuspension events.

Phosphorus release from undisturbed sediment sampled in May and August was 12 mg and 4 mg $m^{-2}$ $d^{-1}$, respectively. During experimental simulation of resuspension, soluble reactive phosphate (SRP) increased by 20–80 $\mu g$ $l^{-1}$, which indicates that a typical resuspension event in the lake would be accompanied by the release of 150 mg SRP $m^{-2}$. The internal P loading induced by resuspension is estimated to be 60–70 mg $m^{-2}$ $d^{-1}$, or 20–30 times greater than the release from undisturbed sediment.

SRP release during simulation of resuspension was mainly dependent on the equilibrium conditions in the water column and was basically independent of the increase in suspended solids and the duration of resuspension. A second simulation of resuspension conducted 26 hours later, did not result in any further release of SRP from sediment sampled in May. In contrast, there was an additional SRP release from sediment sampled in August, indicating that an exchangable P pool, capable of altering equilibrium conditions, is built up between resuspension events.

It is concluded that resuspension, by increasing the P flux between sediment and water, plays a major role in the maintenance of the high nutrient level in Lake Arresø. A relatively high release rate is maintained during resuspension because of the low Fe:P ratio and the high concentration of $NH_4Cl$-extractable P in the sediment.

## Introduction

Wind-induced sediment resuspension, which occurs frequently in shallow water (Andersen & Lastein, 1981; Aalderink *et al.*, 1984; Peters & Cattaneo, 1984), may influence the chemical and biological processes in the lake through effects on turbidity (Gippel, 1989), benthic communities (Jonasson & Lindegård, 1979; Demers *et al.*, 1987), redistribution of sediment (Bengtsson *et al.*, 1990), and nutrient concentration (Hamilton & Mitchell, 1988).

The present article discusses the importance of resuspension to internal P loading in the shallow and eutrophic Lake Arresø, Denmark, and is based on *in situ* observations of suspended solids

and Tot-P concentration during windy periods in 1988, and on laboratory studies of o-P release during simulation of sediment resuspension. A related paper (Kristensen *et al.*, this volume) discusses the frequency of resuspension in Lake Arresø and develops an empirical model which relates suspended solid and Tot-P concentration to wind velocity.

## Materials and methods

### *Lake description*

Lake Arresø, the largest lake in Denmark (41 $km^2$), is eutrophic, shallow and unstratified, with water depth mainly between 2 and 4 m. Some basic characteristics are given in Table 1. In 1988, $NO_3^-$ and $NO_2^-$ concentrations were $<0.01$ mg $N\,l^{-1}$ from June to December. SRP concentration was $<0.02$ mg $P\,l^{-1}$ from May to July, increasing to 0.14 mg $P\,l^{-1}$ at the end of August. Phytoplankton composition was dominated by chlorococcal green algae in 1988.

Suspended solids (SS) and Tot-P concentration in the lake water varied considerably during windy periods. For example, during an increase in wind velocity (WV) from 4 to 10 m $s^{-1}$ in August 1988, SS increased from 50 to 190 mg $l^{-1}$ and Tot-P from 0.3 to 0.8 mg $l^{-1}$ within 9 hours (see also Kristensen *et al.*, 1992). The following days, when WV decreased to about 4 m $s^{-1}$, SS and Tot-P decreased correspondingly. Later in the autumn several other wind-induced resuspension events were recorded (Kristensen *et al.*, 1992).

Sediment composition in Lake Arresø has previously been determined at 36 sampling locations (Hovedstadsrådet, 1989a). The sediment was found to be very homogeneous in the accumulation areas, i.e. areas deeper than approximately 2 m which constitute about 80% of the lake area. In the upper 5 cm of sediment dry weights (DW), loss on ignition (LI) and Tot-P generally differed less than 10% between locations and hence the average figures are given. Sediment DW was low, ranging from only 3% in the upper 2 cm to 7% at a depth of 5–10 cm (Table 2). LI was relatively high, ranging from 42% in the uppermost 2 cm to 33% at a depth of 5–10 cm. Tot-P decreased from 2.7 mg $P\,g\,DW^{-1}$ in the upper 2 cm to 1.4 mg $P\,g\,DW^{-1}$ in 20–30 cm depth (Table 2). Most of the P was in organic form, but loosely adsorbed $NH_4Cl$-extractable phosphorus ($NH_4Cl$-P) constituted a relatively large proportion.

*Table 1.* Characteristics of Lake Arresø 1988.

| | |
|---|---|
| Area ($km^2$) | 41 |
| Max depth (m) | 5.5 |
| Mean depth (m) | 2.9 |
| Mean hydraulic ret. time (yr) | 2.4 |
| Mean summer Tot-P (mg $P\,l^{-1}$) | 0.43 |
| Mean summer Chl a (mg $l^{-1}$) | 0.23 |
| Mean summer Secchi depth (m) | 0.49 |

### *Laboratory studies*

Three 1 m sediment cores consisting of approximately 20 cm sediment and 80 cm lake water were sampled from each of three different locations in the soft-sedimented accumulation area. The cores were incubated in triplicate in darkness at 18 °C and the water column was mixed and oxidized with atmospheric air supplied to the center of the water column.

Water samples were analysed for Tot-P according to Koroleff (1970), SS according to APHA (1980) and SRP according to Murphy & Riley (1962).

Sediment was samples in May and August 1988 and two sets of experiments conducted on each sediment. P release rates from the surface of undisturbed sediment were determined in the first set of experiments and P release during simulation of resuspension was followed in the second set. Both sets of experiments were conducted using the same cores.

During the study of P release from undisturbed sediment, the cores were placed in a continuous flow system and P release determined from the difference between the P concentration of inlet and outlet water (Fig. 1). In order to simulate lake water, the inlet SRP and $NO_3$-N concentration

*Table 2.* Sediment characteristics in the accumulation area (deeper than approx. 2 m, 80% of the lake area) in May 1988. Dry weight (DW), loss on ignition (LI) Tot-Fe, Tot-P and Tot-Fe/Tot-P are means of 36 sampling locations. P-fractionation was conducted in accordance with Hieltjes and Lijklema (1980) and is a mean of 4 sampling locations. Org-P is calculated as Tot-P -($NH_4Cl$-P + NaOH-P + HCl-P).

| Depth (cm) | DW | LI | Fe | Tot-P | $NH_4Cl$-P | NaOH-P | HCl-P | Org-P | Fe:P |
|---|---|---|---|---|---|---|---|---|---|
| | (%) | | (mg $DW^{-1}$) | | | | | | |
| 0–2 | 3.2 | 42 | 5.7 | 2.65 | 0.32 | 0.47 | 0.24 | 1.62 | 2.2 |
| 2–5 | 5.0 | 40 | 6.8 | 2.63 | 0.35 | 0.50 | 0.27 | 1.51 | 2.6 |
| 5–10 | 6.9 | 36 | 7.6 | 2.36 | 0.44 | 0.44 | 0.25 | 1.23 | 3.2 |
| 10–20 | 10.3 | 29 | 8.7 | 1.78 | 0.51 | 0.26 | 0.29 | 0.74 | 4.9 |
| 20–30 | 14.3 | 23 | 10.8 | 1.40 | 0.37 | 0.21 | 0.25 | 0.58 | 7.7 |

was 0 $\mu g\,l^{-1}$ and 500 $\mu g\,l^{-1}$ respectively for the experiments with sediment sampled in May and 100 $\mu g\,l^{-1}$ and 0 $\mu g\,l^{-1}$ respectively for the experiments with sediment sampled in August. Water renewal time in the cores was between 1 and 6 days. Further details of the experimental design are given in Søndergaard (1989). P release from the undisturbed sediment cores was measured both before and after simulation of resuspension.

P release during resuspension was determined by simulating resuspension in the same cores as were used for the study of P release from undisturbed sediment. Resuspension was simulated by means of a piston which continuously moved up and down the water column in close proximity to the sediment surface (Fig. 1). Different SS concentrations could be simulated by varying the distance between the lowest position of the piston and the sediment. Resuspension was simulated twice, the duration of the first resuspension being 4 hours, and that of the second resuspension, which started 26 hours later, being 17 hours for the experiments with sediment sampled in May

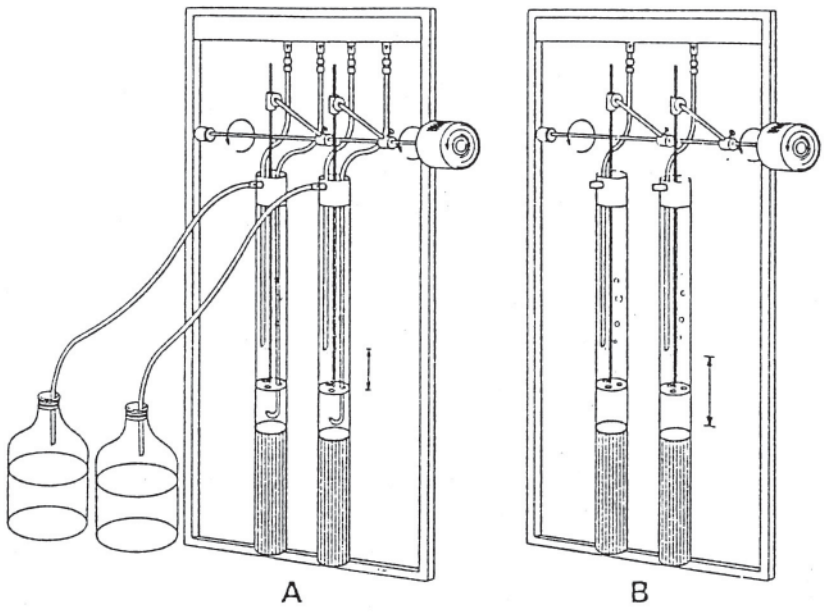



*Fig. 1.* Experimental design used in measuring P release from undisturbed (A) and resuspended (B) sediment.

and 42 hours for the experiments with sediment sampled in August. SS concentration varied between 20 and 300 mg SS $l^{-1}$, which approximates the range determined *in situ* (Kristensen *et al.*, 1992). Between 1 and 8 mm of the sediment were resuspended, which, although 3–4 times less than that calculated from *in situ* observations (20–30 mm), corresponds to the *in situ* situation given the difference in water column. In the individual cores, SS concentration was not necessarily increased to the same level during both simulations of resuspension, and resuspension depth therefore differed. SS, Tot-P and SRP were measured on cessation of resuspension and several times during the subsequent 24 hours. In the case of the first resuspension, carried out for 4 hours, SS, Tot-P and SRP were also measured after 2 hours of resuspension.

P release was calculated as the increase in SRP during the resuspension period. Except for the second resuspension with sediment sampled in May, SRP typically increased 50–70 $\mu g\, l^{-1}$ during resuspension (Table 3). The initial SRP concentration was higher with sediment sampled in August because of the higher SRP inlet concentration in the experiments with undisturbed sediment.

## Results

In the laboratory studies, analysis of variance revealed that the variation in nutrient release between the triplicates from one location was always within the variation recorded between the three different locations. Thus the results obtained with the cores from the 3 locations were analysed together, and most of the values given are the mean of the 9 samples.

There was no significant different in the P release rate from undisturbed sediment prior to or following simulation of resuspension (Table 4). In the experiments with sediment sampled in May, release rates for SRP and Tot-P averaged 3–4 mg and 11–13 mg $m^{-2}\, d^{-1}$, respectively, whereas in the experiments with sediment sampled in August release rates were lower, being about −2 mg and 5 mg $m^{-2}\, d^{-1}$ of Tot-P, respectively.

After resuspension termination, the percentage of SS and Tot-P remaining in suspension declined exponentially (Fig. 2) and within 24 hours, SS had declined to approximately 10%. On the basis of a 50% decline in SS, the settling velocity of the resuspended sediment was calculated to be between 0.4 and 0.5 m $h^{-1}$. Whereas the first resuspension increased SRP in the experiment with sediment sampled in May and August by 40–80 $\mu g$ SRP $l^{-1}$ (Fig. 3B), the second resuspension had no effect on SRP in the experiments with sediment sampled in May but caused a significant increase ($p < 0.002$) in the experiments with sediment sampled in August. SRP remained more or less constant after termination of resuspension, corresponding to a steady increase in the SRP/Tot-P ratio (Fig. 2).

With the exception of two cores in which initial SRP was high due to a high release rate from undisturbed sediment, SRP reached equilibrium at between 20 and 70 $\mu g\, l^{-1}$ of SRP during resuspension of sediment sampled in May (Fig. 3A). Higher SS did not result in higher SRP.

*Table 3.* SRP concentrations ($\mu g\, l^{-1}$) prior to and following the resuspension of May- and August-sediment ($n = 9$).

| | Before | | After | |
|---|---|---|---|---|
| | mean | (min–max) | mean | (min–max) |
| May-sediment | | | | |
| First resusp. | 32 | (1–211) | 80 | (22–252) |
| Second resusp. | 54 | (9–205) | 52 | (12–225) |
| August-sediment | | | | |
| First resusp. | 100 | (73–148) | 172 | (131–202) |
| Second resusp. | 164 | (126–192) | 214 | (146–349) |

*Table 4.* P release from the undisturbed sediment surface (mg P $m^{-2}\, d^{-1}$). Standard deviation is shown in parentheses ($n = 9$).

| | Before resuspension | | After resuspension | |
|---|---|---|---|---|
| | SRP | Tot-P | SRP | Tot-P |
| May | 3.9 (8.0) | 12.7 (11.5) | 3.2 (7.9) | 10.9 (9.8) |
| August | − 1.9 (4.0) | 4.5 (4.2) | − 2.7 (3.0) | 3.5 (3.7) |

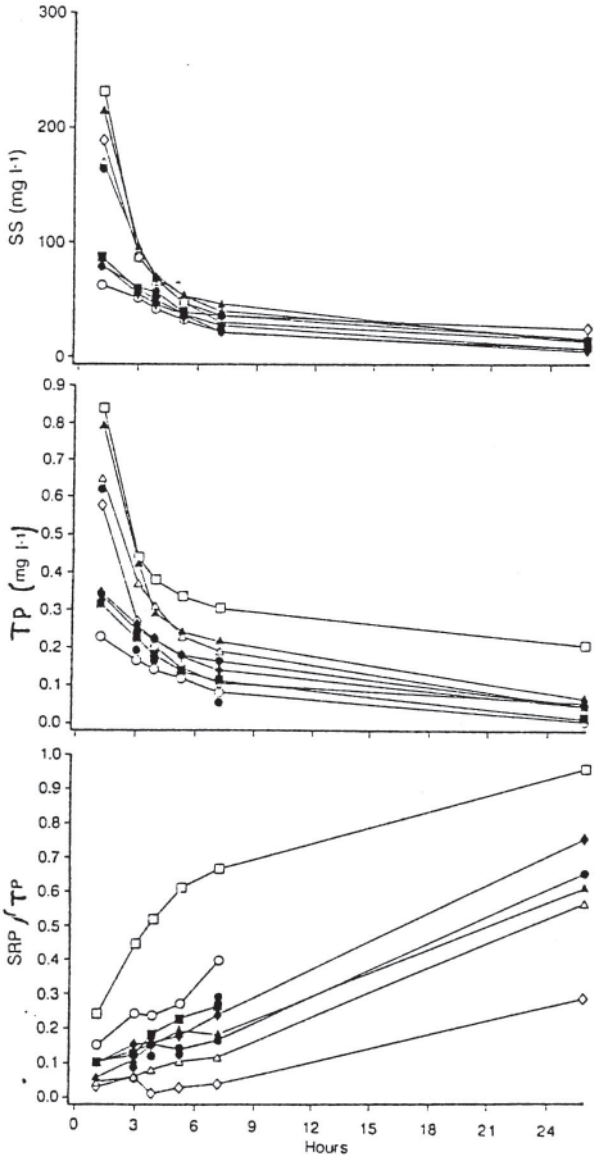



*Fig. 2.* SS, Tot-P and SRP/Tot-P after the second resuspension simulation in August. Resuspension was stopped at hour = 1. The concentrations are the increase caused by resuspension and thus corrected for initial concentrations. Different symbols refer to the 9 cores incubated.

In the case of sediment sampled in August, SRP increased from an initial level of about 100 $\mu g\,l^{-1}$ of SRP to 160–200 $\mu g\,l^{-1}$ of SRP in the first resuspension, and to 180–220 $\mu g\,l^{-1}$ of SRP in the second resuspension (Fig. 3A).

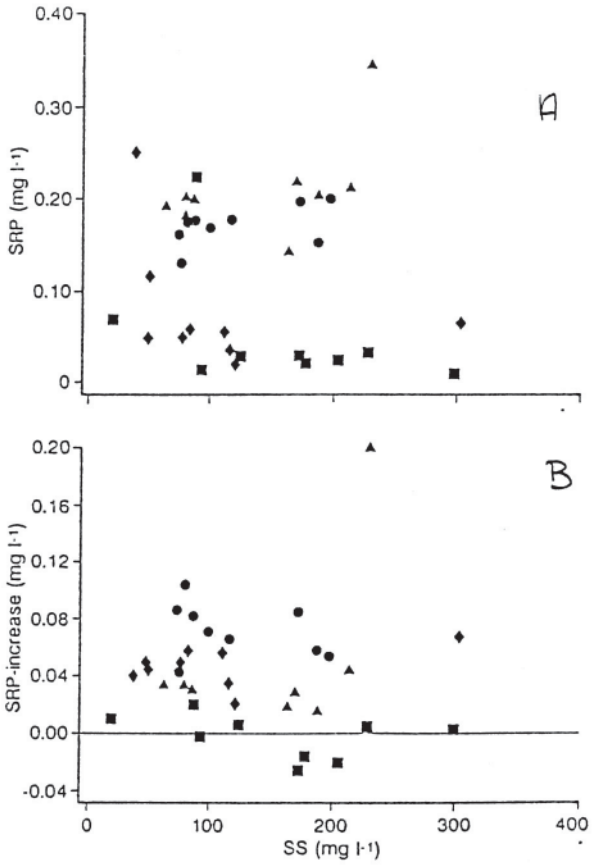



*Fig. 3.* The relationship between the increase in SS and SRP equilibrium concentration on cessation of resuspension (A), the SRP-increase during resuspension (B). First resuspension, May: ◆; second resuspension, May: ■; first resuspension, August: ●; and second resuspension, August: ▲.

## Discussion

Wind-induced resuspension occurs very frequency in Lake Arresø, as discussed by Kristensen *et al.* (1992). Based on the observed changes in SS and Tot-P, and their relationship to WV, it was predicated that wind-induced resuspension occurred about 50% of the time and, on average, caused Tot-P to increase to 0.18 $mg\,l^{-1}$ and SS to 42 $mg\,l^{-1}$.

Depending on the particular properties of the lake water and sediment, resuspension may either increase, decrease or have no effect on the lake water SRP concentration (Andersen, 1974; de

Groot, 1981; Peters & Cattaneo, 1984). Laboratory simulation of resuspension of Lake Arresø sediment revealed that *in situ* resuspension would probably lead to SRP release. The question is, how much would be released and what circumstances determine the release. Four alternative mechanisms and possible explanations are considered below:

*(1) SRP increase during resuspension is caused by a release of high SRP concentrations dissolved in the porewater.*
As porewater SRP concentration is usually higher than the lake water concentration (Boström *et al.*, 1982), resuspension will tend to increase lake water SRP. Porewater SRP concentration was not measured in Lake Arresø, but in order to account for the typical 40–80 $\mu g\, l^{-1}$ of SRP increase observed in the resuspension experiments, the SRP concentration in the uppermost 1–8 mm of the sediment would have to have been $>5$–$10\ mg\, l^{-1}$, which is unrealistic in view of the values measured in other lakes (Enell & Löfgren, 1988). Furthermore, such a high porewater SRP concentration in the uppermost sediment would be expected to create a higher diffusive flux from the undisturbed sediment than was actually measured. Even though resuspension is able to cause major changes in the SRP-profile to a sediment depth of several cm, despite only 1 cm actually being resuspended (Søndergaard, 1990), only a minor part of the SRP-increase can be attributed to SRP release from porewater.

*(2) SRP release is caused by the mineralization of organic matter at the oxic conditions occurring during resuspension*
Resuspension, by improving $O_2$ supply, accelerates the decomposition of organic matter and thereby the release of SRP. However, the very fast SRP equilibrium concentration reached during resuspension (within 2–4 hours), indicates that SRP-release is more likely attributable to adsorption-desorption processes. If the release was caused by biological decomposition of organic matter, then the release of o-P would be expected to increase more steadily during resuspension.

*(3) SRP release is governed by the amount of resuspended sediment.*
If correct this would imply that SRP should have increased with increasing SS. However, no such trend was detected during the experiments (Fig. 3B).

*(4) SRP release from resuspended sediment is governed by SRP concentration in the water column during resuspension.*
SRP-release during resuspension depends on the adsorption-desorption kinetics of P bound to sediment particles i.e. depends on whether the resuspended particles are over- or undersaturated with respect to SRP at the physical and chemical conditions existing in the water column. As the adsorption-desorption of SRP from particles involves many different mechanisms (e.g. Boström *et al.*, 1982; Holtan *et al.*, 1988), the SRP changes are difficult to predict. A relatively constant SRP concentration at different degrees of resuspension in both the experiments with sediment sampled in May and August (Fig. 3A) indicates, however, that SRP-release largely depends on the SRP concentration. The reason for the higher equilibrium concentration in the experiment with sediment sampled in August must be seasonal variation in sediment proporties. $NH_4Cl$-P, in particular, may exhibit large seasonal variation in eutrophic lakes, with low concentration during winter and high concentration during summer (Søndergaard, 1988). In Lake Arresø, $NH_4Cl$-P constituted 12% of Tot-P in the uppermost layer of May-sediment. Although $NH_4Cl$-P was not measured in August-sediment, that part of $NH_4Cl$-P representing loosely adsorbed org-P might be expected to have been higher in August-sediment due to the high sedimentation rate of phytoplankton during summer.

The very low Fe:P ratio of only 2–3 in the upper 5 cm of the sediment (Table 2) may have provoked SRP-release from the resuspended sediment. Jensen *et al.* (in press) thus found an inverse relationship between Fe:P ratio and aerobic SRP-release from 15 Danish lake sediments and concluded that the Fe:P ratio has to be above 15 before Fe is able to prevent P release at aerobic

conditions. They also found a positive relationship between $NH_4Cl$-P and SRP-release. The relative large proportion of Tot-P that is extractable as $NH_4Cl$-P in Lake Arresø, as compared with most other lakes (Boström *et al.*, 1982), may be important in determining the degree of SRP-release during resuspension.

During the second resuspension of sediment sampled in August SRP-release increased significantly. This indicates that SRP-release is not solely determined by the equilibrium conditions because these were unchanged between the first and second simulation of resuspension. It is possible that in the interval between the 1st and 2nd resuspension, a pool of loosely adsorbed P is built up during the mineralization of organic matter which changes adsorption-desorption proporties and causes a higher equilibrium to be reached in the second resuspension. The lack of any effect of the second resuspension on sediment sampled in May can be ascribed to a lower concentration of easy degradable organic material than later in the season.

SRP and Tot-P release rates recorded from the undisturbed sediment prior to and following resuspension were similar and hence were not affected by resuspension per se. This indicates that the non wind-induced P-release is governed by the slow but steady release of org-P during decomposition. That Tot-P release rate was generally higher than SRP-release rate may be interpreted as being due to the release of org-$P_{diss}$. However, continuous mixing of the water column, combined with ebullition of gases formed by anaerobic metabolism and sediment disturbance, may also have caused a 'release' of $P_{part}$. The tendency towards lower SRP-release rate from sediment sampled in August may be a consequence of the higher SRP-concentration in the water column, as this would decrease the diffusional flux from the sediment.

In relation to the *in situ* lake water SRP concentration of 10–20 $\mu g\,l^{-1}$ of SRP in May and 90–140 $\mu g\,l^{-1}$ of SRP in August (Hovedstadsrådet, 1989b), the laboratory studies indicate that *in situ* resuspension would probably lead to an increase in SRP. On the basis of a typical SRP-increase of 50 $\mu g\,l^{-1}$ in the experiments, it can be calculated that a single resuspension event in the lake would lead to the release of approx. 150 mg $m^{-2}$. Assuming a minimum sinking velocity of the chlorococcal green algae of 0.1 m $d^{-1}$ (Reynolds, 1984; Sommer, 1989) this implies that the daily SRP release rate from the sediment most be at least 70 mg $m^{-2}$ $d^{-1}$ to account for the continuously high P concentrations in the lake water. This is about 20–30 times higher than the release from the undisturbed sediment, thereby stressing the importance of resuspension. The release caused by resuspension corresponds to 12% of all the resuspended P.

The reason why SRP remained low in the lake water during the whole summer, in spite of several resuspension events having taken place, is probably due to a rapid algal uptake. The combination of algal uptake and resuspension, thereby may act as mechanisms pumping P out of the sediment.

In addition to SRP-release, the algal availability of P adsorbed on the resuspended sediment may be high enough to support algal growth at normally P-limiting conditions (Bonnie & Stanford, 1988; Viner, 1988). Thus the possible effects of resuspension in Lake Arresø may involve not only the effects of SRP-release but also the effect of a Tot-P release in the order of 500 mg $m^{-2}$ $d^{-1}$. Although only a minor part will be available to algae, it could imply that phosphorus never becomes a limiting nutrient in Lake Arresø. Mean settling velocity of resuspended sediment was estimated to be 0.29 m $h^{-1}$ (Kristensen *et al.*, 1992) which means that even short windy periods causes resuspended sediment to remain in suspension long enough in for the algal uptake to occur.

Resuspension in shallow lakes can therefore be regarded as a mechanism which accelerates the flux of solutes between sediment and lake water, both in the sense of increasing flux of o-P from sediment to water and in increasing the introduction of $O_2$ and other oxidizing elements from water to sediment. Thus shallow and wind-exposed eutrophic lakes may have a generally low Secchi depth, not only because of the high con-

centration of suspended matter, but also because of high nutrient generation and flux from the sediment and a consequently high algal production.

Besides increasing the potential algal biomass, resuspension may also influence the summer succession of phytoplankton by favouring certain groups due to the different nutrient and mixing conditions (Olrik, 1981; Padisák *et al.*, 1988). Because of the frequent occurrence of resuspension in Lake Arresø, and the resultant high nutrient supply, fast growing small species of chlorococcal green algae, which usually dominate at very high P concentrations (Kristensen *et al.*, in press) are favoured. In contrast cyanobacteria, which mainly appear at lower nutrient concentrations will only dominate during shorter periods of calm weather. Resuspension can also have a positive effect on cyanobacteria; however, the shaded environment caused by resuspension may favour cyanobacteria as they are better able to utilizate of low light (Reynolds, 1987).

## Conclusion

Extrapolation from laboratory studies indicates that phosphate is released during the frequently occurring wind-induced resuspension of sediment in Lake Arresø.

The P release caused by sediment resuspension was 20–30 times greater than the release from undisturbed sediment cores. SRP-release during resuspension is mainly determined by the SRP equilibrium conditions, and proceeds until a certain equilibrium concentrations is reached. The equilibrium concentration was approximately 50 $\mu g\, l^{-1}$ for May-sediment and approximately 180 $\mu g\, l^{-1}$ for August-sediment. The higher equilibrium concentration for August-sediment is interpreted as being due to a change in the chemical composition of the sediment i.e. a higher concentration of loosely adsorbed phosphorus. The SRP-release during resuspension was enhanced by the low Fe:P ratio and the high concentration of $NH_4Cl$-P in the sediment.

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# Phytoplankton biomass reduction after planktivorous fish reduction in a shallow, eutrophic lake: a combined effect of reduced internal P-loading and increased zooplankton grazing

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*Key words:* eutrophic, biomanipulation, fish removal, phytoplankton, zooplankton, internal loading

**Abstract**

No recovery was recorded in the shallow and eutrophic Lake Væng, Denmark, after a sewage diversion in 1981, due to an internal phosphorus loading and a dominance of planktivorous fish. In order to improve lake water quality by means of biomanipulation, a total of 2.5 tons of bream (*Abramis brama*) and roach (*Rutilus rutilus*) was removed during 1986 and the spring of 1987. The planktivorous/benthivorous fish biomass was thereby reduced by approximately 50%, from 30 to 15 g WW $m^{-2}$. After the reduction, the biological structure of the lake changed markedly. The zooplankton community changed from a dominance of rotifers before the fish reduction to larger cladocerans afterwards. Zooplankton mean summer biomass increased from 0.4 mg DW $l^{-1}$ in 1986 to 2.7 mg DW $l^{-1}$ in 1987 and to 1.3 mg DW $l^{-1}$ in 1988. Phytoplankton biomass decreased from a mean summer level of 25 $mm^3$ $l^{-1}$ in 1986 to 12 in 1987 and to 7 $mm^3$ $l^{-1}$ in 1988. Qualitatively, the phytoplankton changed from a dominance of cyanobacteria and small diatoms to a dominance of larger diatoms, larger greens and cryptophytes. Secchi depth increased from a mean summer level of 0.6 m in 1986 to 1.0 m in 1987 and 1.3 m in 1988. After the fish removal, the internal phosphorus loading was reduced markedly. This reduction is considered primarily to be caused by the improved redox conditions in the sediment due to reduced sedimentation, and by the increased micro-benthic primary production and phosphorus uptake. Submerged macrophytes, mainly *Potamogeton crispus* and *Elodea canadensis*, increased in abundance due to improved light climate at the lake bottom.

## Introduction

In Denmark, improvements of lake water quality are usually attempted by reducing the external nutrient loading. However, due to an internal phosphorus loading, lake recovery is often considerably delayed (Sas, 1989; Søndergaard, 1989; Jensen & Andersen, in press). Furthermore, a resistance in the biological structure towards reduction in nutrient loading as described by Hrbácek *et al.* (1978) and Jagtman *et al.* (1988) may also delay lake water improvements significantly. As a method of improving lake water quality in such lakes, biomanipulation by reducing the planktivorous/piscivorous fish ratio has been introduced and recently received much attention

(Andersson *et al.*, 1978; Shapiro & Wright, 1984; McQueen *et al.*, 1986; Benndorf, 1987; Gulati & Van Donk, 1989).

The present work is part of a comprehensive study on the effects of reducing the planktivorous fish biomass in shallow eutrophic or hypertrophic lakes in Denmark (Jeppesen *et al.*, 1990a, 1990b. The aim is to analyse the preliminary results concerning the overall changes in the biological and chemical structure after the removal of planktivorous fish from the eutrophic and shallow Lake Væng.

## Study area

Lake Væng is situated in central Jutland, Denmark, in an extensively cultivated area. The lake area is 15 hectares, mean water depth 1.2 m, max. depth 1.9 m, and hydraulic retention time between 2 and 3 weeks (Fig. 1). 90% of the water inflow come from seepage through the lake bottom or from springs along the lake shore.

From 1964 to 1981 Lake Væng received only mechanically treated wastewater. In 1981 the sewage was diverted and the external phosphorus loading was reduced from about 4.1 to 1.5 g $P\ m^{-2}\ y^{-1}$. Phosphorus in the lake water, however, was only reduced from a mean summer (May-Sept. inclusive) concentration of 0.18 mg tot-$P\ l^{-1}$ in 1978 to 0.16 mg tot-$P\ l^{-1}$ in 1986. This was due to an internal phosphorus loading from the sediment of about 1.7 g $P\ m^{-2}\ y^{-1}$. Total phosphorus concentration in the upper 10 cm of the sediment is 3.8 mg P g $DW^{-1}$, most of which is organically bound or bound to iron (Jensen, unpublished results).

No significant increase in Secchi depth was recorded from 1978 to 1986 (Fig. 5). Both in 1978 and 1986 phytoplankton was dominated by cyanobacteria from June/July to September (Fig. 2, and Jeppesen *et al.*, 1989a).

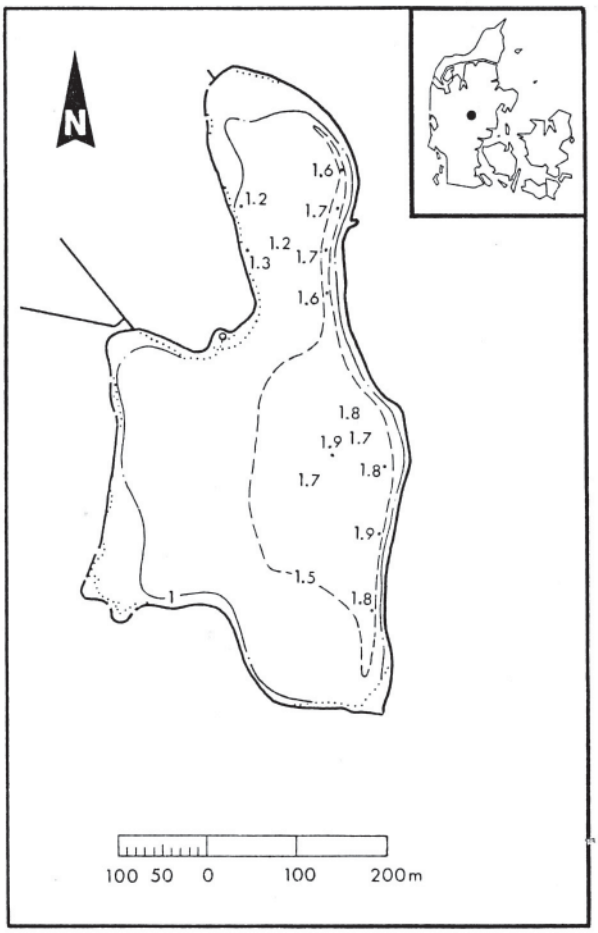



*Fig. 1.* Location and morphology of Lake Væng.

## Materials and methods

### *Investigated parameters*

Lake water parameters were measured with monthly intervals in 1978, 1981 and 1984. From 1986 and onwards samples were taken once or twice a week during summer and monthly during winter. This paper deals mainly with the data from 1986 to 1988.

Phosphate ($PO_4$-P) was analysed by the ascorbic acid molybdene blue method (modified from Murphy & Riley (1972)), total phosphorus (tot-P) was determined after persulphate digestion according to Koroleff (1970), ammonium ($NH_4$-N) by the phenol hypochlorite method modified from Solorzano (1969), nitrate + nitrite

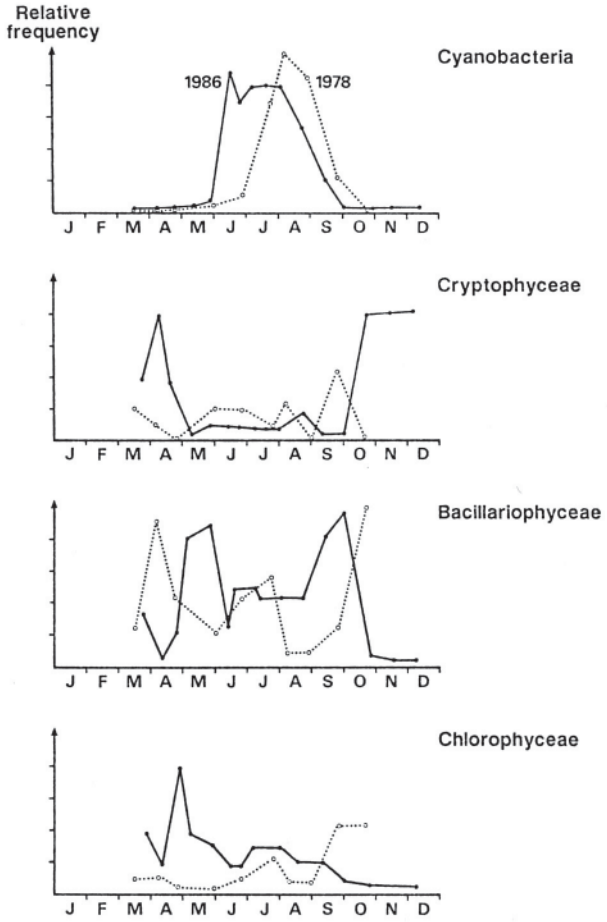



*Fig. 2.* Relative frequency of the main phytoplankton classes in 1978 (broken line) and 1986 (solid line),(1978 is based on semi-quantitative measurements).

according to Dahl (1974), total nitrogen (tot-N) after potassium persulphate digestion according to Solorzano & Sharp (1980), and chlorophyll *a* (chl-*a*) spectrophotometrically after extraction with ethanol according to Holm-Hansen & Riemann (1978). pH was determined potentiometrically.

Zooplankton was sampled with a Ruttner sampler from 0.2 and 1.0 m depths from 1–3 locations. The composite sample from each station was filtered through a 20 $\mu$m net and fixed with Lugol. Zooplankton was subsampled and counted using an inverted microscope. Large individuals were counted under a stereo microscope using the whole sample. Cyclopoid copepods were counted as adults, copepodites and nauplii.

Daphnids were determined to species level and counted. Copepod, cladocera and rotifer biomasses were estimated by use of length/weight relationships according to Dumont *et al.* (1975), Ruttner-Kolisko (1977) and Culver *et al.* (1985). 10–50 individuals of each species/genus were measured on each sampling date. For further details concerning counting procedure, see Jeppesen *et al.* (1990c).

Phytoplankton was sampled with a Ruttner sampler from the same position and depths as zooplankton and fixed in Lugol. Countings were made using an inverted microscope until the maximal error on biomass was less than $\pm 20\%$. Phytoplankton was determined to genus or species level. Phytoplankton biomass was expressed as volume ($mm^3 l^{-1}$) by fitting each genus/species to simple geometric figures.

*Fish stock*

The fish stock, except young-of-the-year (YOY), prior to the manipulation was estimated by a mark-recapture method (Table 1). The YOY were estimated by setting buoyant nets on several days in mid August 1986 (number of nets = 147). Very few bream larger than 10 cm were marked, and none were recaptured. However, in spring 1987 during the spawning period about 100 bream between 25–35 cm and about 400 bream between 45–55 cm in length were caught. These catches are equivalent to 7.8 g WW $m^{-2}$ and were added to the mark-recapture estimate. As a result the estimate of bream has to be considered as a minimum value. The total population of fish was estimated to approximately 500 000. Eleven species were recorded (see Table 1), but numerically only roach (*Rutilus rutilus*) and perch (*Perca fluviatilis*) were important, accounting for 84% and 14% respectively of the total number. In terms of biomass the fish stock was dominated by bream (*Abramis brama*) and roach accounting for 31% and 57% respectively of the total biomass. Both the bream and perch populations were dominated by $0^+$ group fish, while the $1^+$ group out-

*Table 1.* Estimated numbers and biomass of the fish stock in August/September 1986 and number (±95% C.L.) of eels larger than 30 cm in June 1987. + + = species present in small numbers. + = very rare species.

| Species | Age | Number | % | Biomass | |
|---|---|---|---|---|---|
| | | | | (g WW $m^{-2}$) | % |
| Pike, *Esox lucius* | All ages | 500 | 0.1 | 1.7 | 5.5 |
| Crucian carp, *Carassius carassius* | | + | | | |
| Gudgeon, *Gobio gobio* | | + | | | |
| Tench, *Tinca tinca* | | + | | | |
| Bream, *Abramis brama* | $0^+$ | 19000 | 3.8 | 9.3 | 31.2 |
| | $1^+$ and older | 8100* | 1.6 | | |
| Minnow, *Phoxinus phoxinus* | | + | | | |
| Rudd, *Scardinius erythrophthalmus* | | + + | | | |
| Roach, *Rutilus rutilus* | $0^+$ | 128000 | 25.4 | 17.5 | 56.8 |
| | $1^+$ | 251000 | 49.7 | | |
| | $2^+$ and older | 25000 | 4.9 | | |
| Perch, *Perca fluviatilis* | $0^+$ | 67000 | 13.3 | 1.5 | 4.9 |
| | $1^+$ and older | 2000 | 0.4 | | |
| Ruffe, *Gymnocephalus cernua* | All ages | 4100 | 0.8 | 0.5 | 1.6 |
| Total | | 504700 | 100.0 | 30.5 | 100.0 |
| Eel, *Anguilla anguilla* | | 1350 (1030–1920) | | | |

* Includes catch of large fish in May 1987.

numbered the $0^+$ group in the roach population.

Piscivores were almost exclusively represented by pike, accounting for about 5% of the total biomass. Piscivorous perch were only present in very small numbers.

*Fish manipulation*

In the autumn of 1986 and in spring 1987, 2.48 tons (equivalent to 16 g WW $m^{-2}$) of planktivorous fish were removed by fishing with different gears including beach seine, electrofishing and gill, fyke and pound nets. This included 8.3 g WW $m^{-2}$ of roach, 7.8 g WW $m^{-2}$ of bream and 0.1 g WW $m^{-2}$ combined of rudd (*Scardinius erythrophthalmus*) and ruffe (*Gymnocephalus cernua*). Except for bream, composed primarily of large individuals, the fish removed were almost equally distributed among the different size classes.

Since 1986 the fish biomass has been estimated by the CPUE-method (catch per unit of effort) using standard gill nets with different mesh sizes. Accordingly (unpublished results), we estimate

that the removal of 16 g WW $m^{-2}$ of planktivorous fish reduced their biomass by approximately 50% in 1987. In 1988 additionally 6.1 g WW $m^{-2}$ of planktivorous fish were removed.

## Results

### *Zooplankton*

Before the fish removal in late 1986 the filter feeding zooplankton community was dominated by rotifers (Fig. 3), mainly *Keratella* spp. and *Polyartra* spp. After the fish removal, the filter feeding zooplankton community changed towards cladocerans, mainly *Daphnia cucullata* and *Bosmina coregoni*, and cyclopoids in 1987 and 1988.

The average summer biomass of filter feeding zooplankton (May-Sept. inclusive) increased from 0.4 mg DW $l^{-1}$ in 1986 to 2.7 mg DW $l^{-1}$ in 1987 and 1.3 mg DW $l^{-1}$ in 1988 (Jeppesen *et al.*, 1990a). *Daphnia* spp. changed from a summer mean of 24 ind. $l^{-1}$ in 1986 to 172 ind. $l^{-1}$ in 1987 and 90 ind. $l^{-1}$ in 1988 (Fig. 3). *Bosmina* spp. showed a summer mean of 55 ind. $l^{-1}$ in 1986, 775 ind. $l^{-1}$ in 1987 and 395 ind. $l^{-1}$ in 1988. The number of cyclopoid copepods increased from 90 ind. $l^{-1}$ in 1986 to 160 ind. $l^{-1}$ in 1987 and 220 ind. $l^{-1}$ in 1988. Rotifers decreased from a summer mean of 4100 ind. $l^{-1}$ in 1986 to 2100 ind. $l^{-1}$ in 1987 and 730 ind. $l^{-1}$ in 1988. Predacious zooplankton was represented by *Leptodora kindtii* (day-time sampling only) appearing with a summer mean of less than 1 ind. $l^{-1}$ all years.

*Daphnia* spp. and *Bosmina* spp. showed also changes in mean body size. The mean summer length of *Daphnia cucullata* increased from 0.6 mm in 1986 to 0.9 mm in 1987 and 1988, while the mean body length of *Bosmina* spp. increased from 0.30 to 0.45 mm.

### *Phytoplankton*

Cyanobacteria and to some extent diatoms dominated the phytoplankton community in the pre-

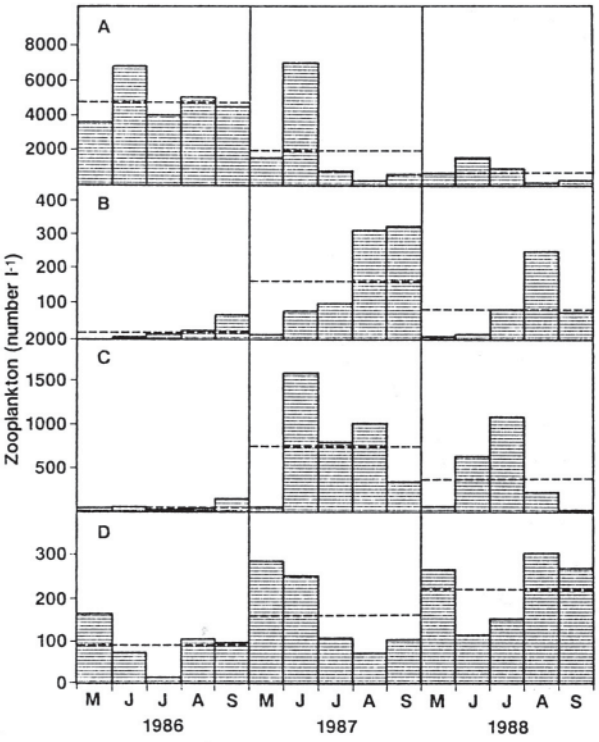



*Fig. 3.* Mean monthly zooplankton numbers (May-Sept. inclusive). A: Rotifers, B: *Daphnia spp.*, C: *Bosmina spp.*, D: Cyclopoid copepods. Summer mean zooplankton number indicated by ---.

biomanipulation period in 1978 and 1986 (Figs. 2 and 4). Among the cyanobacteria the dominating genera were *Anabaena* spp. including some *Gomphosphaeria* spp., while the diatoms were dominated by *Stephanodiscus hantzchii* and to a lesser extent by *Melosira* spp.

In 1987 and 1988 the cyanobacteria were only present in small quantities (Fig. 4). Highest biomass was reached in spring 1987 where *Anabaena spp.* in a short period accounted for 8 $mm^3$ $l^{-1}$. Instead, cryptophytes dominated, mainly *Cryptomonas* spp., and in 1987 also diatoms. As opposed to 1986, the dominating diatom in 1987 was the larger *Asterionella* spp. In 1988, diatoms were present only in low numbers. The large desmid *Closterium limneticum* had a short maximum of 12 $mm^3$ $l^{-1}$ in mid summer 1988.

Mean summer phytoplankton biomass de-

creased from about 25 $mm^3\ l^{-1}$ in 1986 to 12 $mm^3\ l^{-1}$ in 1987 and to 7 $mm^3\ l^{-1}$ in 1988. Correspondingly, mean summer chlorophyll *a* decreased from 76 $\mu g\ l^{-1}$ in 1986 to 39 $\mu g\ l^{-1}$ in 1987 and 29 $\mu g\ l^{-1}$ in 1988. Secchi depth increased from 0.6 m in 1986 to 1.0 m in 1987 and 1.3 m in 1988 (Fig. 5).

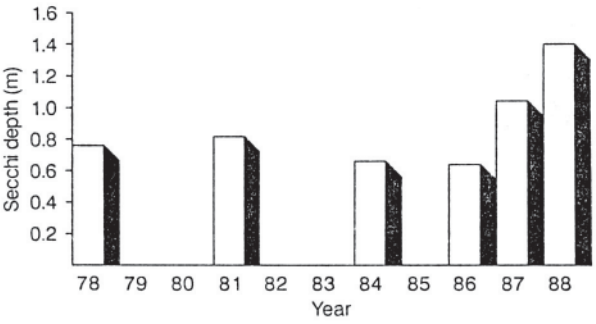



*Fig. 5.* Mean summer Secchi depth (May-Sept. inclusive).

*Submerged macrophytes*

The only submerged macrophytes recorded in Lake Væng were *Potamogeton crispus* and *Elodea canadensis*. The distribution and development of the two species still need to be quantified in details, but the sediment area covered, increased markedly from 1986 to 1987 and from 1987 to 1988, especially on the sandy and wind-exposed east coast (Lauridsen, unpublished results). However, still less than 2% of the bottom area was estimated to be covered by macrophytes in 1988.

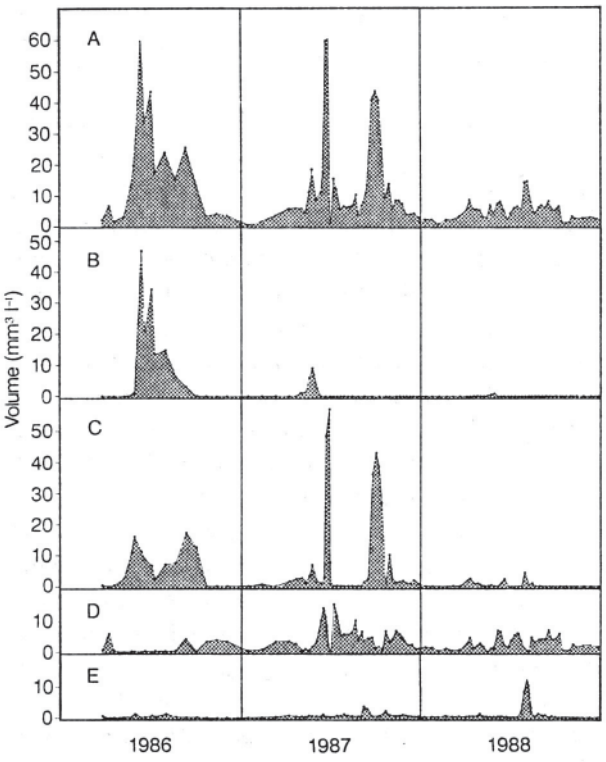



*Fig. 4.* Phytoplankton biomass. A: Total biomass, B: Cyanobacteria, C: Diatoms, D: Cryptophytes, E: Chlorophytes.

*Nutrients*

Before the fish removal, mean summer (May-Sept. inclusive) total phosphorus concentration was 178 $\mu g$ tot-P $l^{-1}$ in 1978 and 157 $\mu g$ tot-P $l^{-1}$ in 1986 (Fig. 6). Maximum concentrations were reached during summer due to the release of phosphorus from the sediment (270 $\mu g$ tot-P $l^{-1}$ August 1978 and 240 $\mu g$ tot-P $l^{-1}$ July 1986). In 1987 mean summer phosphorus concentrations were 125 $\mu g$ tot-P $l^{-1}$ and 20 $\mu g$ $PO_4$-P $l^{-1}$ respectively, decreasing to 87 $\mu g$ tot-P $l^{-1}$ and 8 $\mu g$ $PO_4$-P $l^{-1}$ in 1988. Particularly in 1987, a marked summer sediment release was observed causing a total phosphorus concentration of 250 $\mu g$ tot-P $l^{-1}$ in July, while total phosphorus only reached a maximum of 160 $\mu g$ tot-P $l^{-1}$ in 1988.

From 1978 to 1988 average summer nitrogen concentrations decreased from 1.94 mg tot-N $l^{-1}$ in 1978 to 0.97 mg tot-N $l^{-1}$ in 1986, 1.15 mg tot-N $l^{-1}$ in 1987 and 0.67 mg tot-N $l^{-1}$ in 1988. Nitrate and nitrite concentrations were less than 0.01 mg N $l^{-1}$ from May to July 1987 and during long periods of summer 1988. With the exception of a maximum of 200–300 $\mu g$ $NH_4$-N $l^{-1}$ in July/August 1987 ammonium was between 10 and 40 $\mu g$ $NH_4$-N $l^{-1}$ during most of the summer in both 1987 and 1988. Tot-N : tot-P ratio was 10.9 (1978), 6.2 (1986), 9.2 (1987), and 7.7 (1988).

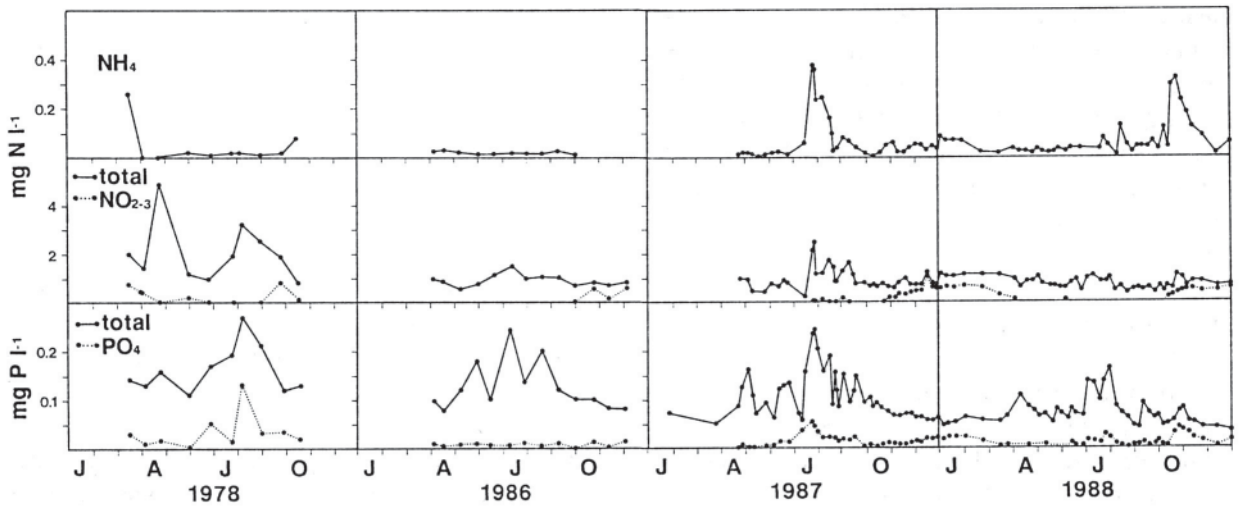



*Fig. 6.* Lake water concentration of ammonium (upper), total nitrogen and nitrate + nitrite (middle); and total phosphorus and ortho-phosphate (lower).

## Discussion

The reduction of the planktivorous and benthivorous fish stock resulted in several quantitative and qualitative alterations of the biological structure of Lake Væng which must primarily be interpreted and assigned to changes caused by the reduced abundance of zooplanktivorous fish.

Hence, the composition of filter feeding zooplankton, in 1986 dominated by rotifers, changed to the larger crustacean zooplankton genera Daphnia and Bosmina in 1987 and 1988. Zooplankton biomass thereby increased 3- to 6-fold. This effect of zooplanktivores is in general agreement with the 'size-efficiency hypothesis' of Brooks & Dodson (1965), and also shown by Andersson *et al.* (1978), and Post & McQueen (1987). Mean body size of *Daphnia cucullata* and other cladocerans also increased in accordance with e.g. Stenson *et al* (1978), Henrikson *et al.* (1980) and Tatrai *et al.* (1985). The changes towards larger zooplankton species and larger zooplankton individuals thus indicate that the present zooplankton structure in Lake Væng is a result of decreased predation by zooplanktivorous fish. The decrease in number of cladocerans from 1987 to 1988 is considered to be caused primarily by food limitation, reflected in unchanged ratio of larger filter feeding zooplankton to phytoplankton (Jeppesen *et al.*, 1990a) rather than increased predation by zooplanktivorous fish which would have been expected to have caused a decrease the length of *Daphnia* spp.

From empirical data, McQueen *et al.* (1986) predicted that bottom-up (producer controlled) forces are strongest at the bottom of the food web, whereas top-down (consumer controlled) forces are strongest at the top of the food web and have less or little impact on the lower trophic levels. The impact of the fish removal in Lake Væng on the zooplankton level seems clear as discussed above. However, the question arises whether the decrease in phytoplankton biomass in Lake Væng is top-down or bottom-up controlled, that is, if the reduction is due to increased zooplankton grazing or to decreased nutrient loading.

It may appear that the decrease in phytoplankton biomass could be a result of the decreasing phosphorus concentration, which declined from a mean summer of 157 $\mu$g tot-P $l^{-1}$ in 1986 to 87 $\mu$g tot-P $l^{-1}$ in 1988. However, this must solely be ascribed to a lower net phosphorus release from the sediment, since no changes have been recorded in the external loading. Furthermore, the decrease in internal phosphorus loading

must be attributed to changes caused by the fish manipulation. Hardly any changes in phosphorus concentrations were recorded from 1978 to 1986, whereas phosphorus concentrations dropped considerably from 1986 to 1987 and 1988 immediately after the fish removal. It is therefore unlikely that this sudden decrease of in-lake phosphorus could be explained by a reduction of internal phosphorus loading due to exhaustion of the exchangeable phosphorus pool.

Nutrient decline after biomanipulation has been reported previously as reviewed by Wright & Shapiro (1984). In Lake Væng the reduction of total phosphorus concentrations caused by the fish removal may be due to several factors: 1) Reduced biomass and sedimentation of phytoplankton and thereby a decreased respiration and oxygen demand at the sediment reducing the internal phosphorus loading (e.g. Boström *et al.*, 1982). 2) Improved light climate at the lake bottom increased micro-benthic primary production and micro-benthic phosphorus uptake. Thus, in 50% of the summer of 1986 Secchi depth reached only 6% of the lake bottom, while 15% and 65% were reached in 1987 and 1988, respectively (Fig. 7). In terms of sufficient light to support a net primary production (calculated as 2.2 Secchi depth), this corresponds to 56% in 1986 and 100% in 1987 and 1988 of the lake bottom in 50% of the summer. This means that a larger proportion of the phosphorus released from the sediment may never reach the lake water, and net release of phosphorus is reduced. The actual benthic primary production has not been measured in Lake Væng, but the importance of benthic algae on nutrient uptake from the sediment has been shown by e.g. Jansson (1980) and Hansson (1989). 3) Reduced abundance of large bream, able to increase sediment phosphorus release by stirring-up sediment particles, may also be important considering the reduced internal phosphorus loading of the lake (e.g. Andersson *et al.*, 1988). This effect is probably important, although it is difficult to verify in Lake Væng because of the problems concerning the biomass estimation of bream. 4) Reduced roach-induced transport of phosphorus from sediment to water. Excretion of phosphorus by bottom-feeding roach may be important (Braband *et al.*, in press), but can not be quantified in Lake Væng due to the lack of data on roach food consumption. 5) Reduced primary production and pH may affect phosphorus composition and phosphorus release from sediments (Andersen, 1975; Søndergaard, 1988). However, mean summer pH in Lake Væng was only reduced from 8.0 in 1986 to 7.8 in 1987 and 1988, which is probably not sufficient to have any significant impact on phosphorus release rates. 6) Reduced phosphorus concentration due to an increased downward translocation of nutrients shifting from a small-bodied to a large-bodied zooplankton community (Wright & Shapiro, 1984), although this mechanism is not considered important in the shallow and well-mixed Lake Væng.

The reduced net release of phosphorus from the sediment after the biomanipulation is probably attributable to not just one but several of the factors mentioned above among which especially, the effect of reduced sedimentation and increased light climate at the bottom is considered important. In addition, a considerable response may be

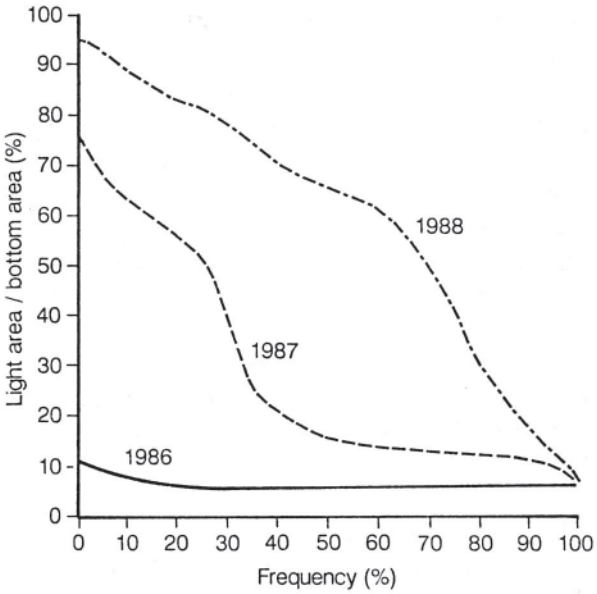



*Fig. 7.* Frequency distribution (summer) of lake area reached by Secchi depth (light area/bottom area). Calculated according to the water depth (Fig. 1) and Secchi depth frequency distribution (Jeppesen *et al.*, 1990a).

expected by only minor changes in the biological structure when close to a nutrient threshold level (Sas, 1989; Jeppesen *et al.*, 1990b). For example, phosphorus in the sediment is bound mainly to iron and thus very sensitive to changes in redox potential. Minor changes associated with sediment oxygen demand may therefore be able to improve the phosphorus binding capacity markedly.

Apart from the reduced internal loading the reduced phytoplankton biomass must also be related to a reduced amount of available phosphorus. Hence, the chlorophyll *a* to total phosphorus ratio decreased from about 0.5 before the fish removal to 0.3 afterwards (Table 2). This decrease is primarily interpreted as an effect of the increased amount of phosphorus bound in zooplankton thereby immobilizing phosphorus previously available to algae. Accordingly, the zooplankton biomass in periods of summer 1988 accounted for about half the total phosphorus concentration in the lake water (estimated from Lehman & Naumoski (1985) based on mean summer numbers and size of *Daphnia spp.*).

Inorganic nitrogen concentrations were low during most of the summer indicating that nitrogen in periods may have been a limiting growth factor to phytoplankton. The increased concentrations in 1987 and 1988 of ammonium which is preferently used by phytoplankton, however, suggest that phytoplankton was not nitrogen limited after the fish removal. Furthermore, the decrease in total nitrogen concentrations after the fish removal was smaller than the decrease in phosphorus which may be related to the much higher percentage of nitrogen originating from external loading.

*Table 2.* Mean summer (May-Sept. inclusive) phosphorus (Tot-P) and chlorophyll-*a* (Chl-*a*) concentration and chlorophyll-*a* to phosphorus ratio.

| Year | Chl-*a* $\mu g\,l^{-1}$ | Tot-P $\mu g\,l^{-1}$ | Chl-*a*/tot-P |
|---|---|---|---|
| 1978 | 94 | 178 | 0.53 |
| 1986 | 76 | 157 | 0.48 |
| 1987 | 39 | 125 | 0.31 |
| 1988 | 29 | 87 | 0.33 |

Tot-N : tot-P ratio did not change significantly after the fish removal and cannot explain the reduced dominance of cyanobacteria. Instead, the increased grazing by zooplankton probably directly prevented the development of large cyanobacterial colonies. The fish removal also affected other phytoplankton groups. Thus, larger phytoplankton species of *Asterionella* spp. and *Closterium* spp. became more abundant indicating a higher grazing pressure of zooplankton on small edible algae, thus sparing inedible or less edible species. The occurrence and dominance of cryptophytes after the fish removal is also interpreted as effects of the increased grazing pressure from zooplankton as *Cryptomonas* spp. is commonly reported to dominate in lakes with a high grazing pressure from zooplankton (Andersson *et al.*, 1978; Fott *et al.*, 1980; Reinertsen & Olsen, 1984). The structural changes which took place in Lake Væng thus seem related both to direct changes in the food web and to indirect changes in nutrient concentrations caused by the fish removal.

As emphasized by Grimm (1989) submerged vegetation is an important factor of lake water quality by manipulating the planktivorous/piscivorous fish ratio. In 1988 the abundance of submerged macrophytes in Lake Væng increased markedly (unpublished results) due to the improved light climate at the lake bottom (Fig. 7), although the sediment area covered by macrophytes in 1988 still did not exceed 2% of the total sediment area. It is questionable if this yet is enough to have any effect on the general lake water quality. On the other hand, Secchi depth reached the lake bottom most of 1988 (Fig. 7) due to the shallowness (mostly between 1.0 m and 1.5 m) of the lake. Empirical data have shown that the Secchi depth corresponds to the maximum depth of submerged macrophyte colonization in Danish lakes (Jeppesen *et al.*, 1989b). Lake Væng, therefore, has reached a light threshold level which will enable macrophytes to recolonize most of the lake bottom. Accordingly, recent results from 1989 (unpublished) show a dramatic

increase in the area covered by submerged macrophytes.

Considering nutrient threshold levels Benndorf (1987) and Jeppesen *et al.* (1990b) stressed the importance of an upper limit of external phosphorus loading, which should not be exceeded if the food web control is to operate successfully. For the Gräfenhain and Bautzen reservoirs the threshold levels of external loading were estimated to between 0.5 and 1.0 g tot-P $m^{-2} y^{-1}$ (Benndorf, 1987). In Lake Væng, pre-biomanipulation external phosphorus loading was 1.5 g P $m^{-2} y^{-1}$, not including the internal phosphorus loading. This level, however, seems to have been sufficiently low to accelerate the restoration process (see also Jeppesen *et al.*, 1990a, b).

Furthermore, the biomanipulation project in Lake Væng at least on a short-term scale seems to have been successful in spite that only about 50% of the planktivorous/benthivorous fish biomass were removed. In the Dutch lake Bleiswijkse Zoom a 50% reduction of planktivorous fish, and in the Danish Frederiksborg Castle Lake an 85% reduction did not suffice to improve water quality significantly (Meijer *et al.*, 1989; Jeppesen *et al.*, 1990a; Riemann *et al.*, 1990). The success in Lake Væng is proposed to be a combination of its shallowness and the low external phosphorus loading, combined with the fact that Lake Væng was close to a nutrient threshold level which could change the primary production from pelagic towards benthic dominated.

## Conclusion

A reduction of approximately 50% of the planktivorous/benthivorous fish stock in Lake Væng had a marked impact on both zooplankton and phytoplankton communities. The observed changes indicate a strong top-down effect of the fish removal towards larger-bodied zooplankton species and increased zooplankton biomass. The phytoplankton biomass decreased and changed towards larger diatoms, larger chlorophytes and cryptophytes instead of cyanobacteria. In addition, the in-lake phosphorus concentrations decreased markedly indicating a certain top-down control also on the nutrient level. The in-lake phosphorus was reduced due to a reduced net phosphorus release from the sediment supposingly caused primarily by reduced sedimentation and increased micro-benthic primary production and uptake of phosphorus.

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# An empirical model describing the seasonal dynamics of phosphorus in 16 shallow eutrophic lakes after external loading reduction

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*Abstract*

Based on monthly mass balances on 7–8 yr of data from 16 shallow (mean depth: 1–10 m), eutrophic, unstratified, or only temporarily stratified Danish lakes, we developed a simple empirical model relating the seasonal variation in lake total phosphorus (TP) concentrations to external loading, accumulated phosphorus in the sediment, hydraulic retention time, and water temperature. The aim was to describe the early recovery phase following an external loading reduction, i.e., when internal phosphorus loading is high, and to include seasonal dynamics. We calibrated a common set of model parameters for all 16 lakes and lake-specific estimates of the exchangeable phosphorus pool in the sediment ($P_s$). Estimated annual mean TP deviated on average 12% from observed values in the 16 lakes. Moreover, the estimated seasonal dynamics and trend following the external loading reduction closely mimicked the observed pattern. The model was successfully tested on nine of the lakes for which data were available for an additional 7-yr period. The results suggest that TP in the sediment does not provide an adequate description of the exchangeable P pool. In Lake Arreskov, which has shifted from a turbid to a clear-water state following fish kill and biomanipulation, the model significantly overestimated TP, indicating that the model is inadequate for describing seasonal dynamics during the shift from a turbid to a clear-water state. Although simple, the empirical model predicts reasonably well the seasonal dynamics of TP following a P-loading reduction in a variety of shallow turbid lakes.

Cultural eutrophication has considerably impaired lake ecosystems worldwide (Hutchinson 1973; Sas 1989). In northern temperate lakes, total phosphorus is regarded as the key factor of eutrophication (Schindler 1975). To improve water quality, many countries have during the last two decades reduced the external nutrient loading of lakes by improving wastewater treatment, including P removal; by increasing catchment retention capacity; and by reducing the phosphorus content of fertilizers and detergents (Phillips et al. 1999; Van der Molen and Boers 1999). However, following the external loading reduction, many lakes suffer from high internal loading, which delays recovery (Marsden 1989; Sas 1989; Jeppesen et al. 2005*b*).

To help managers define acceptable external phosphorus loading levels and predict the effects of various measures to reduce the loading, numerous simple empirical and dynamic models have been developed. The most simple of the models relate lake water total phosphorus (TP) to external loading and require that the lakes be in a steady state (Dillon and Rigler 1974; OECD 1982). These models cannot, however, describe the transient phase following a loading reduction when internal loading is high. They often considerably underestimate lake water TP, since internal loading may in some cases prevail for more than two decades after external loading reduction (Sas 1989; Nürnberg 1998; Søndergaard et al. 2003). To account for internal loading, the sediment pool and sediment release rates have been included in a number of empirical and dynamic models (Nürnberg and LaZerte 2004). These models most frequently belong to the two-compartment type that includes a water and a sediment phase plus an interchange between the two (sedimentation and release). Two-compartment models typically operate with time steps of 1 yr and cannot, therefore, describe the seasonal variation in in-lake TP concentrations. Seasonal dynamics are, however, included in several complex dynamic models, but these models often require comprehensive knowledge of numerous variables, as well as calibration and selection of a large number of unknown parameters (Jørgensen and Mitsch 1983). To our knowledge, no simple models describe changes in seasonal TP in the period following nutrient loading reduction.

From 1989 to 2003, mass balances have been developed on 16 Danish shallow lakes, the majority of which are in a transient state following a TP-loading reduction (Søndergaard et al. 1999). We used the data from 1989 to 1996 to develop a phosphorus model that, based on information on external loading and the exchangeable phosphorus pool in the sediment, allows prediction of the seasonal dynamics of in-lake total phosphorus concentrations of lakes in equilibrium and during the transient state following changes in the external phosphorus loading. The model was subsequently tested on the data from 1997 to 2003. The model was also tested on a biomanipulated lake (Lake Arreskov), which has shifted from turbid to a clear-water state.

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## Materials and methods

*Sampling and analysis*—The 16 lakes included in the study were all encompassed by The Danish Nationwide Lake Monitoring Programme, and the applied sampling procedures and nutrient analyses followed its standardized guidelines (Kronvang et al. 1993). For nutrient analyses, the main inlets of each lake were sampled 18–26 times annually, depending on seasonal variation in discharge, while the minor inlets were sampled less frequently, depending on their relative contribution to total hydraulic and nutrient loading. Lake waters were collected fortnightly during summer and monthly during winter, i.e., 19 times annually. TP was measured as orthophosphate using the method of Murphy and Riley (1972) after persulphate digestion (Koroleff 1970), and chlorophyll *a* was determined after ethanol extraction (Jespersen and Christoffersen 1987).

Total discharge in the main inlets and outlets ($Q_m$) of the lakes was measured monthly using an OTT propeller. The water levels ($H$) in the inlet and outlet streams were automatically recorded during the entire study period. Daily discharge ($Q_d$) was calculated by the use of the relationship obtained for $H$ and $Q_m$ in the inlet and outlet, respectively. In minor inlets, discharge ($q$) was measured with an OTT propeller and daily discharge values were calculated from $q/Q_d$ relationships.

Monthly water balances were calculated for each lake using the following equation: $Q_{\text{inm}} + Q_{\text{inu}} + \text{Prec} = V_{\text{dif}} + Q_{\text{outm}} + Q_{\text{outu}} + \text{Evap}$, where $Q_{\text{inm}}$ and $Q_{\text{outm}}$ are the total discharges measured in inlets and outlets, respectively. Prec and Evap are monthly evaporation and precipitation obtained from meteorological stations situated in the vicinity of the lakes. $V_{\text{dif}}$ is the monthly variation in lake volume. $Q_{\text{inu}}$ and $Q_{\text{outu}}$ are the unmeasured input from the catchment without stream inlet and the output from the lake, respectively. The net value of $Q_{\text{outu}}$ and $Q_{\text{inu}}$ was determined monthly by adjusting the water balance, if $V_{\text{dif}} + Q_{\text{outm}} > Q_{\text{inm}} + \text{Prec}$, then $Q_{\text{inu}}$ equals to the difference; otherwise, $Q_{\text{outu}}$ is equal to the difference.

TP loading was then estimated for each inlet as the product of the daily water discharge and phosphorus concentration (obtained by linear interpolation). TP concentrations of the unmeasured discharges to and from the lake ($Q_{\text{inu}}$, $Q_{\text{outu}}$) were assumed to equal the $Q$-weighted concentrations in the measured inlets and outlets. Atmospheric deposition on the lake surface was estimated using an average rate of 20.0 kg P km$^{-2}$ yr$^{-1}$ (Hovmand et al. 1993).

Lake sediment was sampled during winter months and analyzed at least once during the investigation period. Dry weight was determined by drying at 105°C for 24 h and loss on ignition was subsequently determined by drying to constant weight at 550°C. Total phosphorus in the sediment (sed-TP) was analyzed spectrophotometrically as molybdate reactive phosphorus after extraction of ash-free sediment with 1 mol L$^{-1}$ HCl. Phosphorus in depths from 0–5, 5–10, and 10–20 cm was fractionated according to the sequential extraction technique of Hieltjes and Lijklema (1980) into $NH_4Cl$-P, NaOH-P, HCl-P, and residual phosphorus (Res-P). Res-P was calculated as the difference between sed-TP and the sum of $NH_4Cl$-P, NaOH-P, and HCl-P. For more details of sampling and analysis, *see* Søndergaard et al. (1996). The exogeneous variables (inlet TP, temperature, and hydraulic loading) were interpolated linearly to daily values or with shorter time intervals to match the time scale needed for solving the differential equations with sufficient accuracy. The differential equations were solved by means of either the Euler-scheme or the fourth order Runge-Kutta scheme. The unknown model parameters were estimated by means of a least squares method, where the least square contributions of the lakes are weighted by the number of observations (minus one) per lake and added on a logarithmic scale, i.e., if $\sigma_k^2$ and $n_k$ denote the mean squared errors and the number of observations minus one, respectively, for the $k$th lake, then the total criterion function to be minimized is given by $\Sigma_{k=1}^{16} n_k \log \sigma_k^2$. Hence, a common set of model parameters was estimated for the 16 lakes. The criterion function was minimized by means of the downhill simplex method (Nelder and Mead 1965; Press et al. 1989).

*The model*—The model has two state variables: total phosphorus in the lake water (in-lake TP) and exchangeable TP in the sediment. The driving variables in the model are the monthly inlet concentration of TP, the corresponding monthly water discharge, and the lake water temperature.

The dynamics of in-lake TP are given by the difference between input and output, the sedimentation of TP is deducted, and the release of TP from the sediment is added.

$$\frac{dP_l}{dt} = \frac{Q}{V} \times (f_d \times P_i - P_l) - \text{SED} + \text{REL} \qquad (1)$$

where $P_l$ is in-lake TP (g m$^{-2}$), $P_i$ inlet TP (g m$^{-2}$), SED sedimentation of phosphorus (g P m$^{-2}$ d$^{-1}$), REL sediment release of phosphorus to lake water (g P m$^{-2}$ d$^{-1}$), $f_d$ the fraction of $P_i$ entering the lake water pool (the rest enters the sediment pool). $f_d$ is defined as $1/(1 + \sqrt{V/Q}/365)$, thus declining with increasing hydraulic retention time.

Accordingly, the change in TP in the sediment is given by the following equation:

$$\frac{dP_s}{dt} = \frac{Q}{V} \times (1 - f_d) \times P_i + \text{SED} - \text{REL} \qquad (2)$$

The sedimentation of TP is calculated as a constant ($bS$) multiplied by in-lake TP. The temperature dependence of this process is modeled by a standard Van Hoff's equation.

$$\text{SED} = bS \times (1 + tS)^{T-20} \times \frac{P_l}{Z} \qquad (3)$$

where $tS$ is the temperature correction for $bS$ and $T$ is lake water temperature (°C).

The release is a first order reaction:

$$\text{REL} = bF \times (1 + tF)^{T-20} \times P_s \qquad (4)$$

where $bF$ is a constant, $tF$ the temperature correction for $bF$, and $P_s$ exchangeable phosphorus in the sediment.

The model was implemented in Delphi 3 (Borland International Inc. 1997). The SAS package (SAS Institute 1990) was used for additional graphical and statistical processing of input data and model output. We generally used the Euler

Table 1. Selected physicochemical variables for the 16 lakes (annual mean values for the years 1989–1996).

| Lake | Lake area ($km^2$) | Mean depth (m) | Max depth (m) | Water retention time (d) | Inlet P (annual) (mg P $L^{-1}$) | Lake P (annual) (mg P $L^{-1}$) | Lake P (summer) (mg P $L^{-1}$) | Chloro-phyll *a* (summer) ($\mu$g $L^{-1}$) | Secchi depth (summer) (m) |
|---|---|---|---|---|---|---|---|---|---|
| Borup Sø | 0.10 | 1.1 | 2.0 | 24 | 0.128 | 0.149 | 0.217 | 115 | 0.6 |
| Byrup Langsø | 0.38 | 4.6 | 9.0 | 82 | 0.116 | 0.090 | 0.086 | 38 | 2.0 |
| Dons Nørresø | 0.36 | 1.0 | 1.5 | 18 | 0.094 | 0.169 | 0.262 | 350 | 0.4 |
| Fuglesø* | 0.05 | 2.0 | 2.8 | 56 | 0.151 | 0.223 | 0.332 | 135 | 0.7 |
| GundSømagle Sø | 0.32 | 1.2 | 1.9 | 30 | 0.963 | 0.849 | 0.991 | 269 | 0.4 |
| Hejrede Sø | 0.51 | 0.9 | 3.5 | 53 | 0.170 | 0.139 | 0.148 | 66 | 0.5 |
| Hinge Sø | 0.91 | 1.2 | 2.6 | 18 | 0.114 | 0.126 | 0.172 | 131 | 0.5 |
| Jels Oversø† | 0.09 | 1.2 | 1.9 | 7 | 0.136 | 0.274 | 0.387 | 160 | 0.6 |
| Kilen | 3.34 | 2.9 | 6.5 | 266 | 0.150 | 0.168 | 0.239 | 154 | 0.5 |
| Langesø | 0.17 | 3.1 | 4.5 | 200 | 0.207 | 0.275 | 0.309 | 96 | 1.0 |
| Lemvig Sø | 0.16 | 2.0 | 3.7 | 30 | 0.212 | 0.286 | 0.463 | 110 | 0.5 |
| St. Søgård Sø† | 0.60 | 2.7 | 6.6 | 82 | 0.230 | 0.442 | 0.561 | 67 | 0.8 |
| Søgård Sø | 0.27 | 1.6 | 2.7 | 24 | 0.142 | 0.229 | 0.366 | 193 | 0.4 |
| Tystrup Sø | 6.62 | 9.9 | 21.7 | 180 | 0.295 | 0.257 | 0.215 | 58 | 1.6 |
| Vesterborg Sø | 0.21 | 1.4 | 2.9 | 26 | 0.146 | 0.217 | 0.310 | 111 | 0.5 |
| Øm Sø | 0.42 | 4.0 | 10.5 | 18 | 0.124 | 0.094 | 0.100 | 53 | 1.4 |
| Min | 0.05 | 0.9 | 1.5 | 7 | 0.094 | 0.090 | 0.086 | 38 | 0.4 |
| Median | 0.34 | 1.8 | 3.2 | 30 | 0.148 | 0.220 | 0.286 | 113 | 0.6 |
| Mean | 0.91 | 2.5 | 5.3 | 70 | 0.211 | 0.249 | 0.322 | 132 | 0.8 |
| Max | 6.62 | 9.9 | 21.7 | 266 | 0.963 | 0.849 | 0.991 | 350 | 2.0 |

* 1989–1990, 1992–1996.
† 1990–1996.

integration routine. However, final results were always recalculated using the Runge-Kutta integration routine to ensure adequate precision of the calculations.

Table 2. Parameters used for testing the model. The parameter values are obtained from the calibration on the data from 1989 to 1996.

| Parameter | | Calibrated value measured |
|---|---|---|
| Sedimentation rate, *bS* (m $d^{-1}$) | | 0.0470 |
| Temperature dependence of *bS, tS* | | 0 |
| Sediment release rate, *bF* ($d^{-1}$) | | 0.000595 |
| Temperature dependence of *bF, tF* | | 0.0800 |
| Phosphorus in sediment, $P_s$ ($t=0$) g P $m^{-2}$ | Borup Sø | 20.9 |
| | Byrup Langsø | 13.8 |
| | Dons Nørresø | 42.4 |
| | Fuglesø | 47.9 |
| | Gundsømagle Sø | 90.3 |
| | Hejrede Sø | 12.9 |
| | Hinge Sø | 26.2 |
| | Jels Oversø | 68.7 |
| | Kilen | 32.0 |
| | Langesø | 48.0 |
| | Lemvig Sø | 63.7 |
| | St. Søgård Sø | 95.1 |
| | Søgård Sø | 48.5 |
| | Tystrup Sø | 40.0 |
| | Vesterborg Sø | 33.4 |
| | Øm Sø | 0.00 |

## Results

*Test data set*—The 16 turbid lakes used for testing the model were shallow with a mean depth ranging between 0.9 and 9.9 m and without permanent summer stratification (Table 1). Lake surface area ranged between 0.05 and 6.62 $km^2$. All lakes had a short water retention time (7 to 266 d). The annual mean inlet and in-lake TP concentrations were high (0.094 to 0.963 mg P $L^{-1}$ and 0.090 to 0.849 mg P $L^{-1}$, respectively), chlorophyll *a* thus being high (38–350 $\mu$g $L^{-1}$) and Secchi depth low (0.4–2.0 m). During the past 10–20 yr, the external phosphorus loading to most of the lakes has been reduced, and they accordingly suffer from internal loading (Søndergaard et al. 1999). Consequently, the median of annual mean TP was higher in the lake water than in the inlet, and in-lake TP exceeded inlet TP in 11 of the 16 lakes (Table 1).

Since the exchangeable sediment pool is difficult to estimate using measured data (Søndergaard et al. 2003), we first calibrated initial $P_s$ on the entire data series for each lake. $P_s$ is given in Table 2 and observed and estimated in-lake TP values are shown in Fig. 1. Generally, we observed good correspondence between observed and estimated TP. CV ranged between 26% and 75% (mean = 41%) and root mean square error (RMSE) of predictions between 0.03 and 0.38 (median = 0.07). With the exception of Lake Borup, the inlet concentrations differed significantly from the in-lake concentration, particularly during summer, suggesting that the seasonal dynamic of internal loading is traced well by the model. In the 16 lakes, the estimated annual mean lake water TP during the 7–8 yr of study only deviated 0–44%

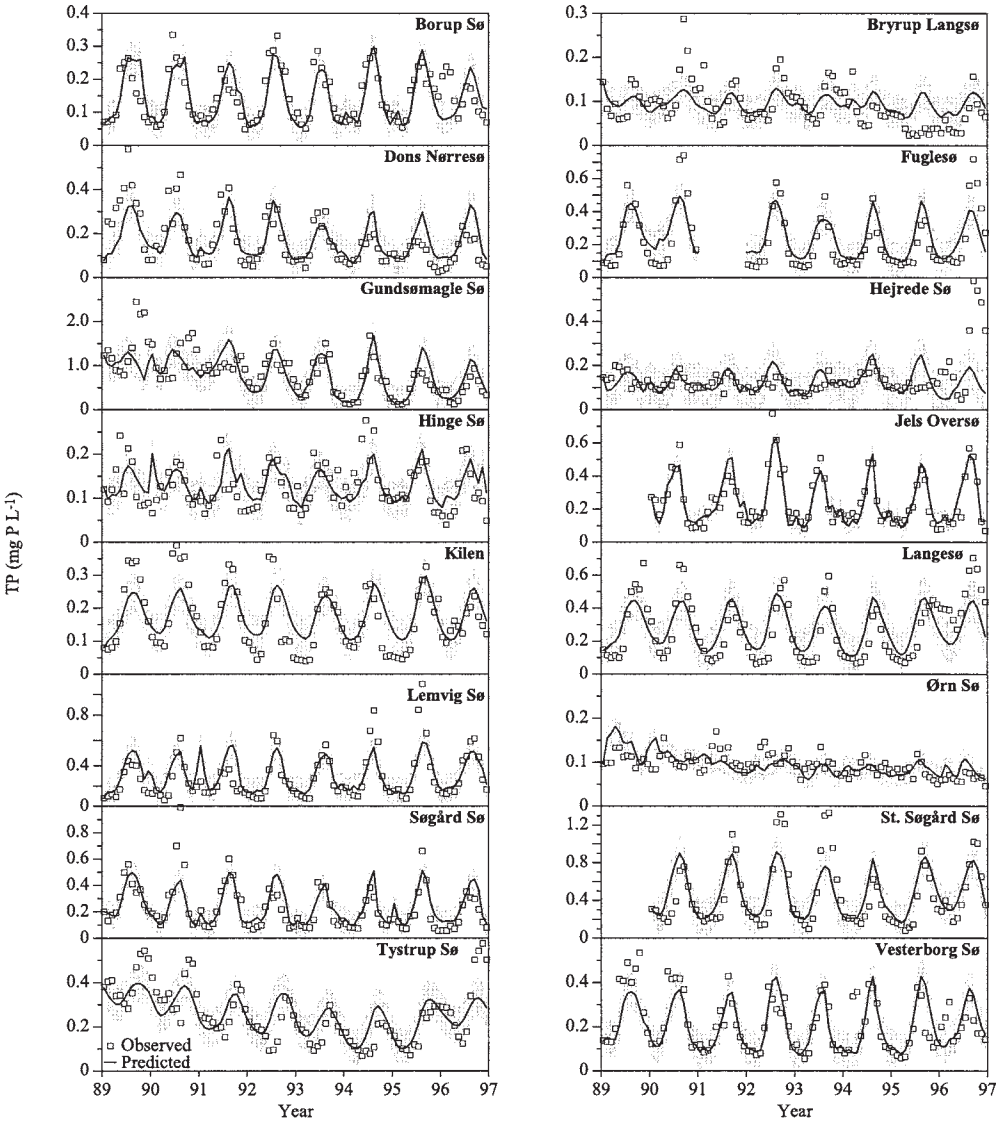



Fig. 1. Observed and predicted total phosphorus (TP) for the 16 shallow lakes during 1989–1996 using the model parameters in Table 2. The gray areas represent predicted values ± one standard deviation ($\sigma_k$ in the criterion function).

(mean = 12%) from observed values, while predictions based on the Vollenweider steady state model deviated 2–237% (mean = 56%) (Table 3, Fig. 2).

The estimated $P_s$ in the 16 lakes did not relate significantly to any of the phosphorus fractions in the sediment (Spearman correlation, $p > 0.05$) but was significantly related to the measured TP pool in the uppermost 20 cm of the sediment (Fig. 3; Spearman correlation, $p < 0.05$). The scatter of the relationship was, however, high (Fig. 3). Particularly one lake (Ørn Sø), which has very high iron concentrations in the sediment (140 mg Fe $g^{-1}$ dry weight, unpubl. data) deviated from the general relationship by having very high TP concentrations in the sediment. By using the measured pools instead of the calibrated ones, estimated annual mean TP deviated 4–176% from observed values (mean = 34%) for the 16 lakes (Table 3).

When calibrating the sediment phosphorus pool, we used data from all 7–8 yr studied. Since time series of that length are not frequently found, we have also tested how the correspondence between measured and calculated in-lake TP depends on the number of years used for the calibration (Fig. 4). We found only small changes in RMSE when reducing the number of years, median RMSE increased successively from 0.09 to 0.10 and 0.15, when data from the first 7, 4, and 1 yr, respectively, were used for calibration. Likewise, in most lakes only minor reductions in RMSE were found during the first year, when data from this year only were used for calibration (Fig. 4).

*Biomanipulated lake*—Lake Arreskov (3.17 $km^2$, mean depth 1.9 m) shifted from a turbid to a clear-water state following fish kill in 1991 and a subsequent moderate fish manipulation (Jeppesen et al. 1998; Fig. 5). The shift resulted in a major TP decline, unmimicked by the model.

Table 3. Annual mean total phosphorus concentrations observed and estimated using the calibrated exchangeable pool of phosphorus in the sediment and the measured pool in the upper 20 cm of the sediment.*

| Lake | Observed TP (mg P $L^{-1}$) | Estimated TP (using calibrated $P_s$) (mg P $L^{-1}$) | Deviation % | Estimated TP (using measured $P_s$) (mg P $L^{-1}$) | Deviation % | Estimated TP (using Vollenweider) (mg P $L^{-1}$) | Deviation % |
|---|---|---|---|---|---|---|---|
| Borup Sø | 0.149 | 0.149 | 0 | 0.176 | 18 | 0.102 | −32 |
| Byrup Langsø | 0.090 | 0.090 | 0 | 0.240 | 166 | 0.079 | −12 |
| Dons Nørresø | 0.168 | 0.168 | 0 | 0.131 | −22 | 0.077 | −54 |
| Fuglesø† | 0.222 | 0.241 | 8 | 0.196 | −12 | 0.108 | −51 |
| Gundsømagle Sø | 0.839 | 0.792 | −6 | 0.736 | −12 | 0.728 | −13 |
| Hejrede Sø | 0.139 | 0.078 | −44 | 0.251 | 80 | 0.125 | −10 |
| Hinge Sø | 0.127 | 0.102 | −20 | 0.210 | 66 | 0.093 | −27 |
| Jels Oversø‡ | 0.249 | 0.226 | −9 | 0.285 | 14 | 0.118 | −53 |
| Kilen | 0.167 | 0.123 | −27 | 0.140 | −16 | 0.081 | −51 |
| Langesø | 0.272 | 0.234 | −14 | 0.316 | 16 | 0.123 | −55 |
| Lemvig Sø | 0.285 | 0.275 | −4 | 0.396 | 39 | 0.165 | −42 |
| St. Søgård Sø‡ | 0.447 | 0.420 | −6 | 0.429 | −4 | 0.158 | −65 |
| Søgård Sø | 0.228 | 0.192 | −16 | 0.175 | −24 | 0.113 | −50 |
| Tystrup Sø | 0.257 | 0.219 | −15 | — | — | 0.175 | −32 |
| Vesterborg Sø | 0.216 | 0.144 | −33 | 0.274 | 27 | 0.115 | −47 |
| Øm Sø | 0.094 | 0.083 | −11 | 0.254 | 176 | 0.102 | 9 |
| Min | 0.090 | 0.078 | −44 | 0.131 | −24 | 0.077 | −65 |
| Median | 0.219 | 0.180 | −10 | 0.251 | 16 | 0.114 | −44 |
| Mean | 0.247 | 0.221 | −12 | 0.280 | 34 | 0.154 | −37 |
| Max | 0.839 | 0.792 | 8 | 0.736 | 176 | 0.728 | 9 |

* Also shown is the estimated phosphorus concentration based on the model by Vollenweider (1976), assuming equilibrium with external loading and deviations between the estimated and observed (as percentages of observations) total phosphorus concentrations by the three different calculation methods.
† 1989–1990, 1992–1996.
‡ 1990–1996.

Both when using measured $P_s$ in the sediment and $P_s$ estimated by calibration on the entire study period, highly positive residuals before fish kill and highly negatively values afterward were obtained (Fig. 5). The residuals tended to be even more negative after 1991 when $P_s$ was calibrated on data from 1989 to 1990.

*Sensitivity analysis*—Sensitivity analyses were performed by calculating the relative increase in RMSE in response to halving and doubling the value of each of the parameters *bS, bF,* and *tF* in Table 2 and by changing the value of *tS* to 0.08. In each scenario only one parameter value was changed. This was done keeping the initial phosphorus concentrations in the sediment at the estimated values in Table 2 (Fig. 6, upper plot) and by reestimating the phosphorus pool in the sediment (Fig. 6, center plot). For the latter case, box plots of the absolute increases in the estimated initial phosphorus concentration in the sediment are presented in the lower plot in Fig. 6. Our model responds qualitatively as expected to the altered parameters values, and the model appears to be more sensitive to the values of the sedimentation and release rates than to the temperature parameters. However, the most sensitive parameters are clearly the initial $P_s$ values. If these are reestimated, changes in the model parameters imply only moderate changes in RMSE, whereas the reestimated initial $P_s$ values may change quite dramatically. This emphasizes that our model is primarily a model for in-lake TP and that it may model in-lake TP data using sedimentation and release rates at different levels by adjusting the level of the sediment pool accordingly. Hence, lake-specific estimates of the model parameters may, artificially, differ considerably, which is why it may be important to estimate common values of *bS, bF,* and *tF* for several lakes.

*Model test on data from 1997 to 2003*—Equivalent data are available for the period 1997–2003 for 9 of the 16 lakes allowing a further test of the model (Fig. 7). The model was applied to these data using observed values of the exogeneous variables in the test period. The relative increase in RMSE for the test period relative to the estimation period 1989–1996 ranged from −51% to 17% with a median of −0.03%, i.e., the RMSE values were generally lower in the test period than in the estimation period.

## Discussion

The developed model, though simple using only few parameters, predicted reasonably well the seasonal dynamics of the turbid lakes despite the highly variable hydraulic retention times and external TP loadings. The generally lower RMSE in the test period compared with the estimation period should not be interpreted as evidence of a better model fit in the test period because the explanation is probably that the level of in-lake TP is generally lower in the test period, hence a measure of absolute deviation is likely to be smaller. Generally, the model predicted the observed in-lake TP concentration fairly well, however, with a tendency to overestimation. A conservative model predicting too high TP level

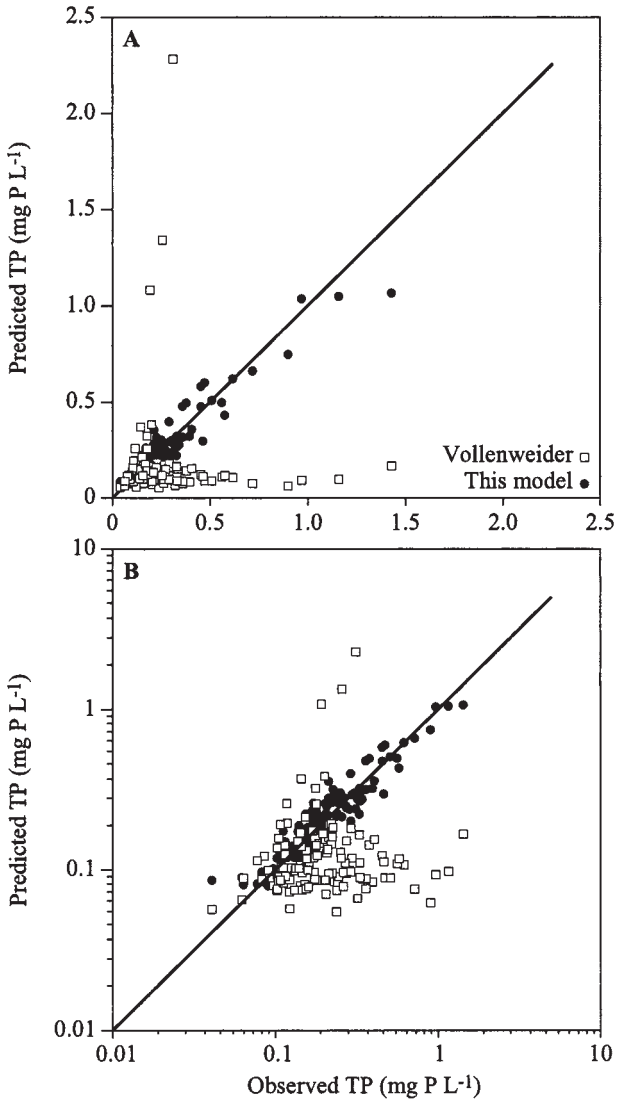



Fig. 2. Inlet, observed, and estimated annual mean (TP) in the lake water showing our model and Vollenweider. The line shows the 1 : 1 ratio. (A) Linear scale, (B) $\log_{10}$ scale.

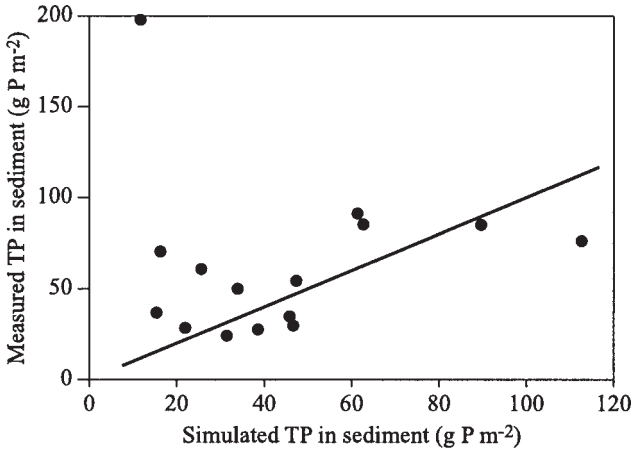



Fig. 3. Observed total phosphorus (TP, g P $m^{-2}$, 0–20 cm) in the sediment of the 16 lakes and average monthly simulated concentrations of exchangeable P in the sediment. The line shows the 1 : 1 ratio.

and a too long recovery period may to some extent be explained by the fact that the model parameters were estimated on data in a recovery period with, probably, a larger internal loading in the initial phase of recovery (release rate, *bF*). Furthermore some of the lakes might have shifted in the direction of a more clear-water state (though still relatively turbid) during the study period (Jeppesen et al. 2005*a*), which will induce a much higher retention of phosphorus

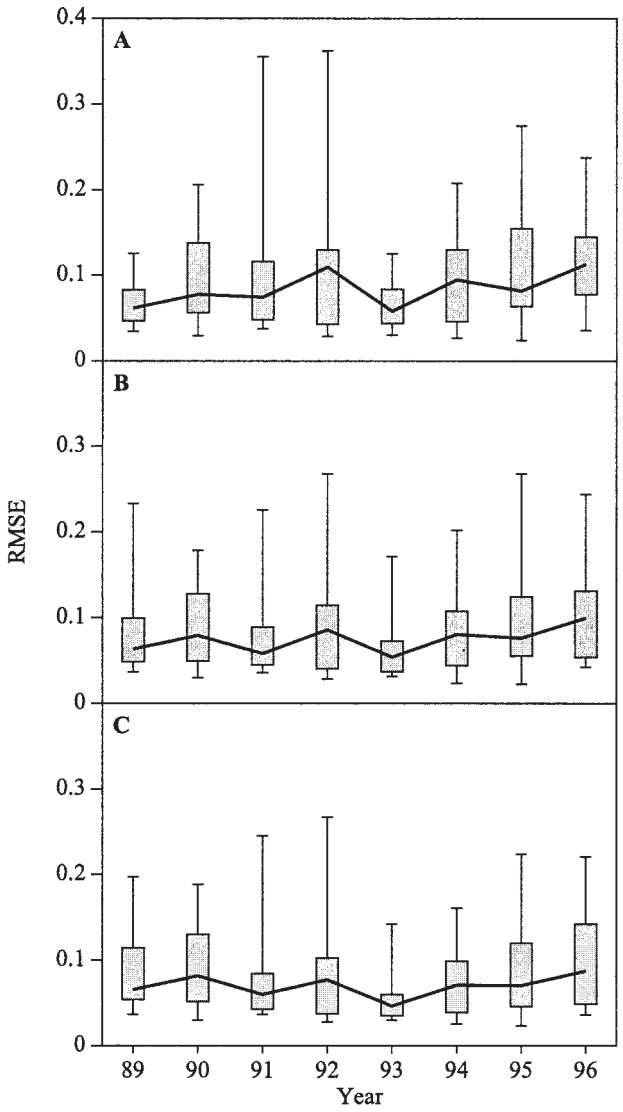



Fig. 4. Box plots showing root mean square error (RMSE) on the prediction of total phosphorus (TP) in the 16 turbid lakes studied during 8 yr, when calibrating the exchangeable P pool in the sediment on (A) the first (upper panel), (B) four (middle panel), and (C) seven (lower panel) years of data, respectively. Full line indicates median values. Also shown are 10%, 25%, 75%, and 90% percentiles.

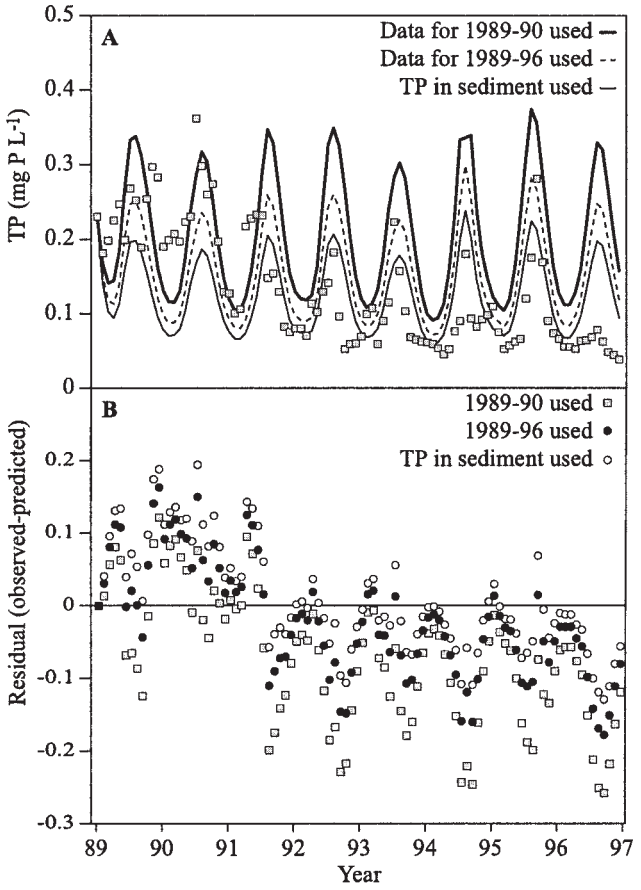



Fig. 5. Lake Arreskov before and after a major fish kill in 1991 and subsequent fish manipulation. (A) Observed and predicted total phosphorus (TP) concentrations based on calibration of $P_s$ on data from 1989 to 1990, data from 1989 to 1996 and measured TP in sediment. (B) Residuals for the same relationships. RMSE was 0.095 when $P_s$ is calibrated on data from 1989 to 1990, RMSE was 0.059 when $P_s$ is calibrated on data from 1989 to 1996, and RMSE was 0.068 when $P_s$ is the measured TP in the sediment.

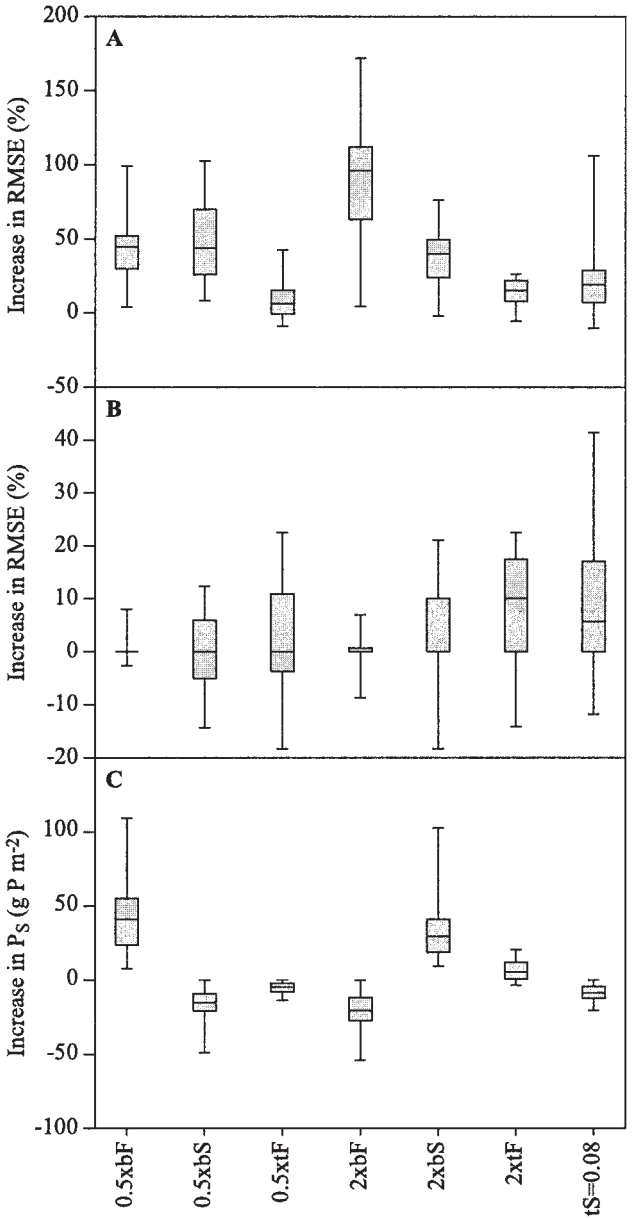



Fig. 6. Box plots of the relative increases in RMSE in response to changing one of the model parameters while keeping the estimated initial $P_s$ values at the estimated values in Table 2 (upper plot) and (B) while reestimating the initial $P_s$ values (center plot). The lower plot shows box plots of the absolute increases in the estimated initial $P_s$ values.

(Søndergaard et al. 1999), as clearly indicated by the results from biomanipulated Lake Arreskov that has shifted to a clear-water state (Fig. 5). In conclusion, some refinement of the model is needed to make it amenable for long-term predictions.

The calibrated temperature coefficient for phosphorus release from the sediment corresponds to a $Q10$ of 2.2 and is, thus, close to the value of a physiological temperature response. That temperature plays a key role for the seasonal variation in the phosphorus release of shallow lakes has been evidenced by several studies (Boström et al. 1982; Jensen and Andersen 1992). However, the reason is not only increased phosphorus release due to increased mineralization of organic matter, but also the consequence of reduced redox level in the top surface sediment affecting the capacity of iron to retain phosphorus (Mortimer 1941; Stauffer 1981).

The coefficient of sedimentation was calibrated to 0.047 m $d^{-1}$ or approximately 5% of the in-lake TP pool $d^{-1}$ in a 1-m deep lake. This rate is low compared with those given by Reynolds (1984), but an explanation may be that a considerable fraction of TP in the lakes consists of orthophosphate during summer (Søndergaard et al. 2005). The sedimentation part of the model could be made more causal if only particulate TP was included. This would, however, require a substantially more complex model including processes for the exchange between dissolved TP and particulate TP and, therefore, as a minimum, additional models (and data) for algal phosphorus uptake and release by grazing. Enhanced complexity would make the model less valuable for managers.

It is remarkable that a simple model is capable of covering

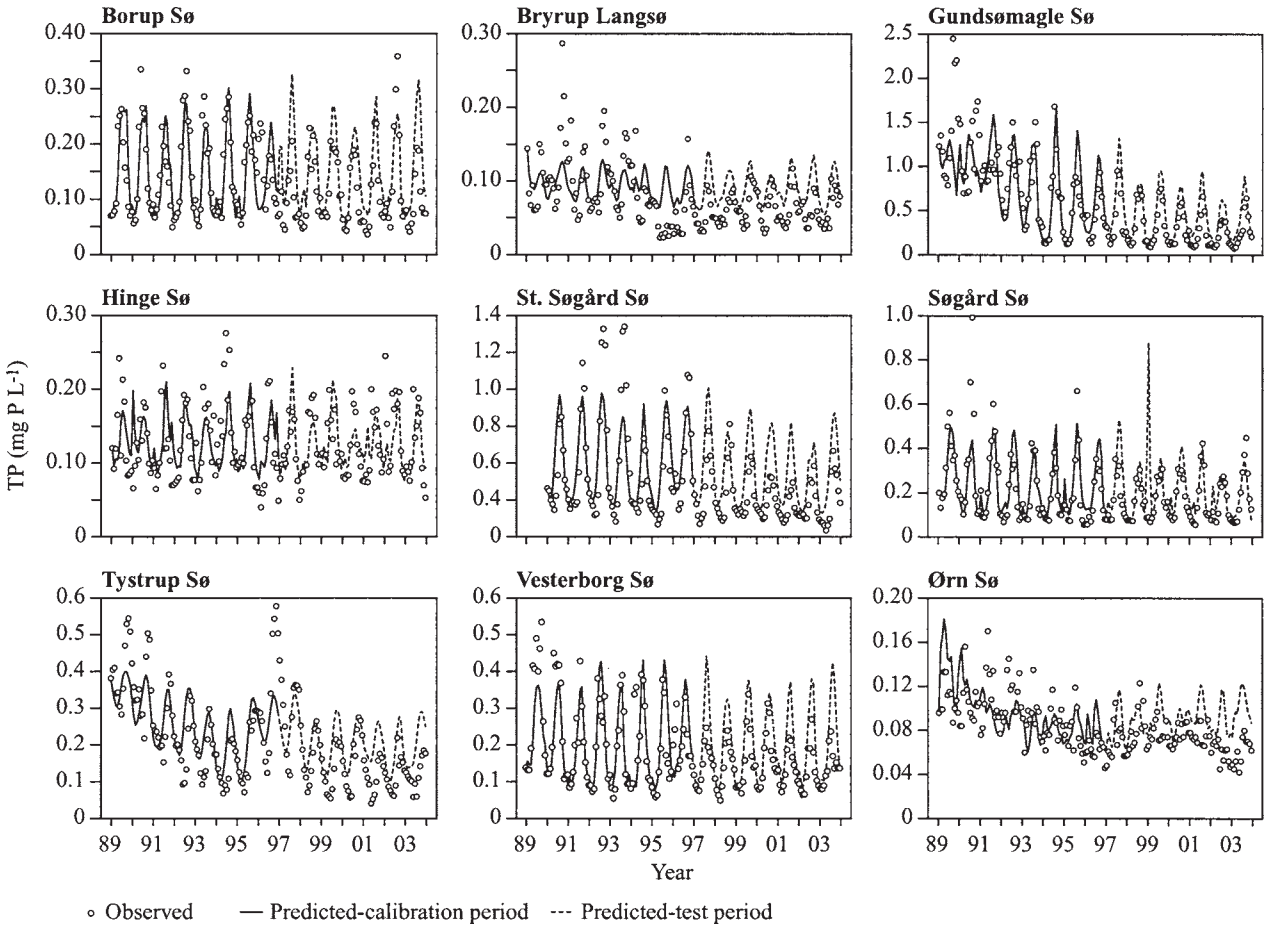



Fig. 7. The model applied to data from 1997 to 2003 for 9 of the 16 lakes using observed values of the exogeneous variables for all years. Predicted values in the calibration period 1989–1996 are connected by a solid line and by a dashed line in the test period 1997–2003.

so well the seasonal dynamics of TP in the 16 turbid lakes used for testing the model, considering that temperature is far from the only factor influencing phosphorus release from the sediment (Boström et al. 1982). The explanation is probably that temperature integrates most of the seasonal mechanisms responsible for the phosphorus release in eutrophic relatively iron-rich lakes. In such lakes, phosphorus release is stimulated in summer when nitrate reaches critically low levels (Andersen 1982) and when pH increases (Boström et al. 1982; Welch and Cooke 1995). Low nitrate and high pH typically occur in summer (in some lakes pH may be high in spring too) when the temperature is high, and this may explain the high predictive power of temperature in our model. We did not deliberately include nitrate and pH in the model because complex mechanisms are included in the seasonal dynamics of these two variables that are difficult to forecast. P release may also be enhanced during spring and summer when the decomposition rate increases, mobilizing both organically bound phosphorus and inorganic phosphorus sorbed to redox-sensitive compounds (mainly iron) because of the diminished oxidized layer of sediment (Søndergaard et al. 2003).

By calibrating $P_s$, closer correspondence between observed and estimated lake TP was obtained from the 16 test lakes than by using the observed TP pool in the upper 20 cm of the sediment. This is not surprising, since the exchangeable sediment phosphorus pool is difficult to determine. First, it is difficult to define the maximum depth from which phosphorus is released. Studies of the changes in the sediment phosphorus pool of highly eutrophic Lake Søbygård in Denmark have shown that phosphorus is released from depths down to 20 cm (Søndergaard et al. 1999), and we therefore chose this depth for the present investigation. The "active" depth is, however, likely to vary from lake to lake, depending on factors such as sediment type and shear stress, and sometimes only 10 cm of the upper sediment is considered to be actively involved in the sediment–water interactions (Boström et al. 1982). Second, the exchangeable pool depends on how phosphorus is bound in the sediment (Boström et al. 1988). To identify the exchangeable P pool, several fractionation methods have been developed (Hieltjes and Lijklema 1980; Boström et al. 1982; Psenner and Puscko 1988). It is believed that the exchangeable P pool mainly consists of the loosely and iron-bound phosphorus, which can be extracted by ammonium chloride and sodium-dithionite, respectively (Psenner and Puscko 1988). However, the value of using fractionation for determining the exchangeable P pool has been debated extensively

(Stauffer 1981; Boström et al. 1988; Jensen et al. 1992), and so far it has not been possible to establish any distinct relationship. Owing to the difficulties involved in determining the exchangeable P pool, we recommend it to be calibrated. For most lakes, RMSE of the prediction was not sensitive to the number of years used for calibrating the exchangeable P pool. Even when the pool was calibrated on data from a single year only, the model generally had a relatively high predictive power. The very high TP levels measured in the sediment of one lake with high iron concentrations emphasize the importance of iron in binding phosphorus in the sediment (Søndergaard et al. 1996).

While the models using the estimated or measured TP pool in the sediment for some lakes underestimated and for others overestimated annual mean in-lake TP, the predictions by the model developed by Vollenweider (1976) generally and often substantially underestimated lake water TP (Table 3). This was to be expected since the Vollenweider model is based on steady state conditions; most Danish lakes, including the study lakes, are, however, in a transient state following the reduced external loading, resulting increased internal loading (Jeppesen et al. 1991; Søndergaard et al. 1999). As an illustrative example, net retention in eutrophic Lake Søbygård was still negative in 1998, sixteen years after loading reduction (Søndergaard et al. 1999), despite the lake's short hydraulic retention time (approx. 1 month). Thus, our results suggest that the duration of the period with excess internal loading may be long even in lakes with a short hydraulic retention time (<1 month) (Jeppesen et al. 1991; Søndergaard et al. 1999).

The results from the biomanipulated Lake Arreskov indicate that the model was unable to track the changes occurring in the seasonal P dynamics of shallow lakes shifting from a turbid to a clear-water state. The model markedly overestimated in-lake TP in the clear-water state. The response of Lake Arreskov is typical for fish-manipulated lakes, and a number of investigations have shown phosphorus and nitrogen retention to increase considerably after a shift to the clear-water state (Søndergaard et al. 2003; Boers et al. 1991; Jeppesen et al. 1998). This is probably the explanation of the model's inadequacy. Higher P retention may reflect a light-mediated increase in the growth of microbenthic algae enhancing sediment oxidation and thus the phosphorus-binding capacity of iron (Hansson 1989; Van Luijn et al. 1995). However, other factors also may be involved. Hence, lower phytoplankton abundance means lower sedimentation of phytoplankton and accordingly lower oxygen consumption in the sediment, potentially enhancing the redox potential. In addition, an increase in the abundance of benthic invertebrates due to a decline in fish predation pressure (Andersson et al. 1978; Giles et al. 1989) may contribute to higher sediment oxidation, although the role of invertebrates for P release is ambiguous (Andersson et al. 1988). Furthermore, increased abundance of submerged macrophytes due to improved light conditions may also be of importance, though high densities of macrophytes in eutrophic lakes may stimulate sediment P release (Perrow et al. 1994; Moss et al. 1996). For Lake Arreskov, however, uptake by macrophytes cannot be the main reason for the decline in in-lake TP and enhanced TP retention in the sediment (Fig. 7), since the recolonization of submerged plants started 2 yr after the abrupt decline in summer TP (Jeppesen et al. 1998). More complex models need to be established to cover the effects of such changes in trophic structure on the P dynamics of shallow lakes.

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# Does resuspension prevent a shift to a clear state in shallow lakes during reoligotrophication?

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*Abstract*

Water managers often debate whether resuspension of sediment with high organic matter and water content accumulated during eutrophication delays improvement of water clarity after reduction of external nutrient loading. Using data from 15 shallow (mean depth <5 m) eutrophic lakes surveyed during 8–12 yr, we show that the reduction in phytoplankton biomass after external loading reductions of phosphorus or changes in the abundance of planktibenthivorous fish was accompanied by a proportional or nearly proportional reduction in detritus and inorganic suspended solids. The reduction occurred irrespective of lake size (0.1–40 km$^2$), extent of phytoplankton biomass reduction (up to 10-fold), and despite dominance of sediments with high water and organic content. Therefore, we conclude that recovery of shallow lakes after nutrient loading or fish stock reduction is apparently not significantly delayed by resuspension of organic or inorganic matter accumulated in the sediment during eutrophication.

World-wide, many lakes suffer from eutrophication due to high external loading from sewage, industries, and runoff from cultivated soils. Large efforts have been made during the last two decades to combat eutrophication by reducing excess loading (Sas 1989; Van der Moelen and Boers 1994), and this has led to improvement of the ecological state and water quality of some lakes (Sas 1989; Jeppesen et al. 2002). However, many lakes show great resistance to improvement due to high internal loading of phosphorus (Sas 1989), homeostasis in the fish community (Gulati et al. 1990; Perrow et al. 1997), and waterfowl grazing (Søndergaard et al. 1997). Furthermore, it has been argued that resuspension of the sediment with high organic matter and water content accumulated during eutrophication also delays or even prevents a shift to the clearwater state in large, shallow, wind-exposed lakes (Bachmann et al. 1999; Meijer et al. 1999). The arguments suggest that continuous resuspension of detritus reduces the light climate sufficiently to prevent growth of submerged macrophytes (Bachmann et al. 2001*a*) or that the sediment in the reoligotrophication phase is too loose

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Table 1. Physicochemical characteristics of the 15 study lakes. The chemical data represent summer averages (1 May–1 Oct 1989–2000).

| | Mean | Median | Minimum | Maximum |
|---|---|---|---|---|
| Lake area ($km^2$) | 3.34 | 0.42 | 0.10 | 40 |
| Mean depth (m) | 2.2 | 1.9 | 1.0 | 4.6 |
| Hydraulic retention time (yr) | 0.7 | 0.2 | 0.04 | 2.7 |
| Total phosphorus (mg $L^{-1}$) | 0.20 | 0.15 | 0.06 | 0.85 |
| Total nitrogen (mg $L^{-1}$) | 2.1 | 1.9 | 0.9 | 3.7 |
| Chlorophyll *a* ($\mu$g $L^{-1}$) | 107 | 78 | 11 | 326 |
| Suspended solids (SS) (mg $L^{-1}$) | 25 | 22 | 5 | 64 |
| Inorganic suspended solids: SS (%) | 26 | 25 | 1 | 46 |
| Nonalgal organic suspended solids: SS (%) | 54 | 53 | 32 | 88 |
| Secchi depth (m) | 1.1 | 0.9 | 0.4 | 2.0 |

and therefore unsuitable for establishment of rooted plant communities (Meijer et al. 1999). Yet plant establishment has occurred in several large shallow reoligotrophic lakes, like Lake Veluwe and Lake Wolderwejd in The Netherlands (Hosper 1997) or following fish manipulation as seen in the U.S.A. (Hanson and Butler 1990). Lowe et al. (2001) also observed a reduction in suspended solids roughly proportional to the reduction in Chl *a* in large Lake Apopka in Florida, where plant establishment remains poor (Lowe et al. 2001). The ongoing debate on the future perspectives of trophic state and management for Lake Apopka (Bachmann et al. 1999; Lowe et al. 2001; Bachman et al. 2001*a,b*; Schelske and Kenney 2001) illustrates well the lack of consensus on the role of resuspension for lake recovery.

We contribute to the debate by presenting data on inorganic and organic fractions of suspended matter from 15 shallow Danish lakes after major reductions in external total phosphorus (TP) loading (Jeppesen et al. 2002; Søndergaard et al. 2002). The lakes vary in size from 0.1 to 40 $km^2$ and in mean depth from 1.0 to 4.6 m and were generally eutrophic (Table 1).

## Materials and methods

Water samples (depth-integrated samples from the photic zone) for analyses of chemical variables and phytoplankton biovolume were taken at a midlake station biweekly during summer (1 May–1 October) and monthly during the remainder of the year. Phytoplankton was counted on Lugol-fixed samples using an inverted microscope. Biovolume was calculated by fitting the different species and genera to geometric forms. A factor of 0.29 was used to convert phytoplankton biovolume ($mm^3$ $L^{-1}$) to biomass (mg organic dry weight $L^{-1}$) (Reynolds 1984).

Suspended solids (SS) were determined as matter retained on GF/C filters after drying at 105°C for 24 h and the organic content as loss-on-ignition (LI) (550°C, 2 h) of SS. LI may potentially include some $CaCO_3$, thereby overestimating the organic content. Yet parallel measurements of LI and particulate COD (chemical oxygen demands) on 1,811 samples, however, show good correspondence between the two measures (Jeppesen et al. 1999). Calculations were made to determine nonalgal organic suspended solids (naorgSS) by subtracting phytoplankton biomass from LI, inorganic suspended solids (inorgSS) by subtracting LI from SS, and nonalgal suspended solids (naSS) by summing up naorgSS and inorgSS. Chlorophyll *a* (Chl *a*) was measured after ethanol extraction of matter retained on a GF/C filter.

Total discharge of tributaries and outlets ($Q_{out}$) was measured monthly with an OTT-propeller. Water level (H) in the inlet streams was automatically and continuously recorded and daily discharge calculated by use of the relationship obtained between H and $Q_m$. Daily TP loading was estimated for each inlet as the product of the daily water discharge and phosphorus concentration obtained by linear interpolation. Loading from the lake catchment not covered by streams was calculated as $Q_{out} - Q_{in}$ assuming TP to equal the *Q*-weighted mean concentrations in the measured inlets. Atmospheric deposition on the lake surface was estimated using an average rate for Denmark of 0.2 kg P $ha^{-1}$ $yr^{-1}$.

Sediment cores were taken with a Kajak sampler (5.2 cm in diameter) at 4–7 midlake stations in each of four lakes, then sliced and analyzed for wet weight, dry weight, and loss-on-ignition (550°C, 1 h) and total phosphorus (TP) (as molybdate reactive phosphorus after extraction of ash-free sediment with 1 mol HCl $L^{-1}$). Only data on the upper 5 cm of the sediment was used in the present analyses.

To test trends in the time series at selected physicochemical variables, we used the seasonal Kendall trend test (Hirsch and Slack 1984). This test is a robust nonparametric statistical method commonly used in environmental science for testing seasonal trends in time series. We used data from 7 yr, from the months of April to October, inclusive. The other calendar months were excluded because of too many missing values. For a given year and month, we averaged all observations before analysis.

## Results and discussion

*Regression analyses*—The 15 lakes are shallow and nutrient rich (summer mean: 0.064–0.850 mg P $L^{-1}$) with high phytoplankton biomass (Chl *a*), high concentrations of SS (Table 1), and, accordingly, low Secchi depth (summer mean: 0.4–2 m). The contribution of both naorgSS and that of inorgSS to SS were overall high, which is typical of shallow Danish lakes (Jeppesen et al. 1999), and reflects the shallowness of the lakes and the frequent occurrence of resuspension. NaorgSS averaged 54% and inorgSS 26% dur-

Table 2. Pearson correlation coefficients for some selected physicochemical variables in 15 study lakes over 8–12 years. Number of samples ranged between 1,775 and 3,144. All pairwise comparisons were significant ($P<0.0001$), though the weakest ones must be interpreted with care because of the large number of samples.

| | InorgSS | NaorgSS | Chl *a* | TP | Secchi | Area | Mean depth |
|---|---|---|---|---|---|---|---|
| Suspended solids (SS) | 0.84 | 0.71 | 0.83 | 0.74 | −0.90 | 0.26 | −0.25 |
| Inorganic suspended solids (inorgSS) | | 0.50 | 0.65 | 0.63 | −0.76 | 0.12 | −0.42 |
| Nonalgae suspended organic solids (naorgSS) | | | 0.58 | 0.61 | −0.64 | 0.47 | −0.20 |
| Chlorophyll *a* (Chl *a*) | | | | 0.71 | −0.84 | 0.18 | −0.22 |
| Total phosphorus (TP) | | | | | −0.77 | 0.11 | −0.33 |
| Secchi depth (Secchi) | | | | | | −0.18 | 0.32 |
| Lake area (area) | | | | | | | 0.28 |

ing summer (Table 1). Pearson correlation analyses on log-transformed data showed highly significant positive correlations between SS, inorgSS, naorgSS, Chl *a*, and in-lake TP (Table 2). In addition, all forms of suspended matter (SS, inorgSS, naorgSS, and Chl *a*) were weakly positively correlated to lake area and negatively so to lake mean depth (Table 2). In a multiple regression, Chl *a,* in-lake TP, lake area, and mean depth contributed significantly to the variation in SS, inorgSS, and naorgSS. We tested for collinearity by calculating a condition index (Rowlings 1988). A condition index around 10 indicates that collinearity affects the regression, a collinearity of 30 and 100 being moderate to strong and values over 100 indicating serious collinearity problems. For the three models with Chl *a,* TP, area, and depth as explanatory variables, we obtained condition indices of around 20, indicating moderate problems. However, by exclusion of TP from the models (Table 3), the index was $<10$.

As Secchi depth is highly significantly related to SS, inorgSS, and naorgSS (Table 2), both detritus and inorganic suspended solids could potentially delay or prevent the clearing up of lakes during the recovery phase following nutrient loading reductions, provided that these variables are not affected themselves by the reduction in phytoplankton biomass. To illustrate how SS, naorgSS, and inorgSS are affected by changes in phytoplankton biomass, we focus on the four lakes with the largest changes in Chl *a* during the survey period, due to either reduced external TP loading prior to (data not shown) or during the course of the investigation (Figs. 1–4). For one lake, Lake Arreskov, in addition to TP reduction, the biomass of planktivorous fish was reduced substantially by fish kills in 1991 and stocking of piscivorous pike in the following years (Fig. 4). For all four lakes, SS followed closely the changes in Chl *a* and also largely the algal biomass (Figs. 1–4, panels B and C). Accordingly, only comparatively minor changes occurred in the proportion of naorgSS and inorgSS to SS even though contributions to SS were high throughout the year (Figs. 1–4, panel D).

*Time series analyses*—Lake Dons (0.36 km$^2$, mean depth: 1.0 m), showed a significant decline (detrended for seasonal variations) in in-lake TP ($P = 0.018$), Chl *a* ($P = 0.017$), while Secchi depth increased ($P = 0.010$). Summer mean Chl *a* declined from 444 to 232 $\mu$g L$^{-1}$ from 1989–1997 (Fig. 1), SS from 76 to 32 mg L$^{-1}$, and LI from 42 to 22 mg L$^{-1}$. We found no significant changes ($P > 0.05$) in the fractions of inorgSS, phytoplankton biomass, nonalgae suspended matter to SS or naorgSS, all detrended for seasonal variations.

In Lake Arresø (40 km$^2$, mean depth: 2.9 m), Chl *a* and SS showed increasing trends until 1993, followed by a major decline coinciding with a reduction of in-lake TP (Fig. 2). The share of nonalgae suspended matter was constantly high except for a reduction at the end of the study period, while the proportion of inorgSS tended to increase. Considering the entire period only, in-lake TP showed a significant decline during the period. However, if only the period with a declining trend (1994–2000) was included, a significant decline was found for in-lake TP ($P = 0.010$), Chl *a* ($P = 0.018$), SS ($P = 0.009$), while no significant decline ($P > 0.05$) was found for the contribution of inorgSS to SS. Only the contribution of naorgss to SS decreased slightly ($P = 0.029$).

In Lake Vesterborg (0.21 km$^2$, mean depth: 1.4 m), a significant decline was observed for in-lake TP ($P = 0.003$), Chl *a* ($P = 0.007$), SS ($P = 0.001$) and phytoplankton biomass ($P = 0.006$), while Secchi depth increased ($P = 0.001$).

Table 3. Multiple linear regression of logarithmically transformed (natural log) total suspended solids (SS), inorganic suspended solids (inorgSS), and nonalgae organic suspended solids (naorgSS) (all in mg DW L$^{-1}$) in the 15 study lakes studied during 8–12 years versus a number of independent variables, chlorophyll *a* (Chl *a*), lake area (area), and mean depth (depth). Other units as in Table 1. In all cases, the relationship was statistically significant ($P < 0.0001$). Total phosphorus was excluded due to collinearity.

| | Intercept | Log (Chl *a*) | Log (area) | Log (depth) | $r^2$ | *n* |
|---|---|---|---|---|---|---|
| $\log_e$ (SS) | 0.77±0.05 | 0.57±0.01 | 0.13±0.008 | −0.32±0.03 | 0.75 | 1,759 |
| $\log_e$ (inorgSS) | 0.03±0.09 | 0.49±0.02 | 0.10±0.02 | −0.58±0.08 | 0.47 | 1,759 |
| $\log_e$ (naorgSS) | 0.57±0.7 | 0.47±0.01 | 0.23±0.01 | −0.48±0.04 | 0.61 | 1,814 |

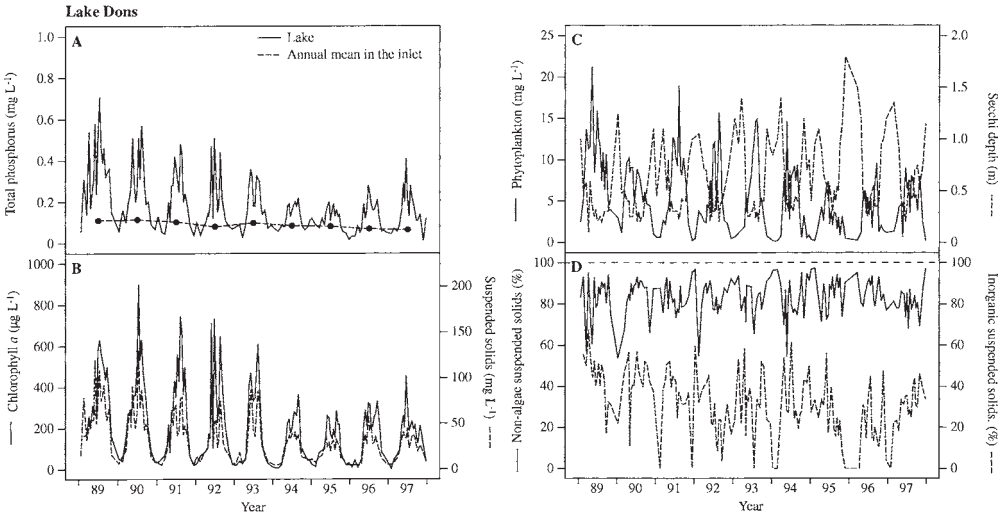



Fig. 1. Lake Dons: (A) discharge-weighted annual mean inlet concentration and lake concentration of total phosphorus, (B) chlorophyll *a* and total suspended solids, (C) biomass of phytoplankton and Secchi depth, and (D) the proportion of nonalgae and inorganic suspended solids of total suspended solids.

Summer mean Chl *a* ranged from 175 to 80 $\mu$g L$^{-1}$ and SS from 43 to 23 mg L$^{-1}$ (Fig 3). InorgSS was only monitored for the last 3 yr and the analysis is therefore restricted to the naSS, which showed no significant changes during the period ($P > 0.05$).

In Lake Arreskov (3.17 km$^2$, mean depth: 1.9 m), a marked reduction in Chl *a* and SS occurred following fish kills in 1991 and a subsequent addition of piscivorous fish, followed by somewhat higher values between 1999–2000 (Fig. 4). If we consider only the period with declining Chl *a* (1989–1997), then the 10-fold reduction in summer mean Chl *a* (from 130 to 12 $\mu$g L$^{-1}$) was accompanied by nearly similar proportional reductions in SS (from 61 to 6 mg L$^{-1}$) and in LI from 40 to 3 mg L$^{-1}$. Accordingly, no significant changes occurred in the contribution of naorgSS or inorgSS to SS during the period. For the entire study period, a significant decline was observed in in-lake TP ($P = 0.037$), Chl *a* ($P < 0.05$), SS ($P < 0.01$), and phytoplankton biomass ($P < 0.025$), while Secchi depth increased ($P = 0.017$).

Thus, the changes in the proportion of different fractions of SS through time were small or insignificant compared with the major changes recorded in SS and Chl *a* concentrations. The decline in naorgSS and inorgSS concentrations cannot be attributed to a plant-mediated reduction in the shear stress at the lake bottom, as otherwise seen in lakes with abundant plant coverage (James and Barko 1994; Ham-

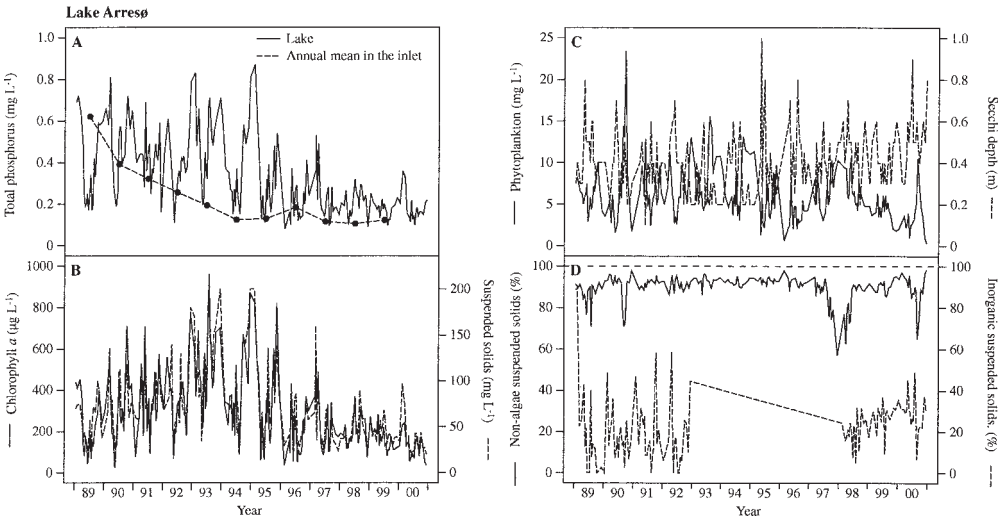



Fig. 2. Lake Arresø: (A) discharge-weighted annual mean inlet concentration and lake concentration of total phosphorus, (B) lake concentration of chlorophyll *a* and total suspended solids, (C) biomass of phytoplankton and Secchi depth, and (D) the proportion of nonalgae and inorganic suspended solids of total suspended solids.

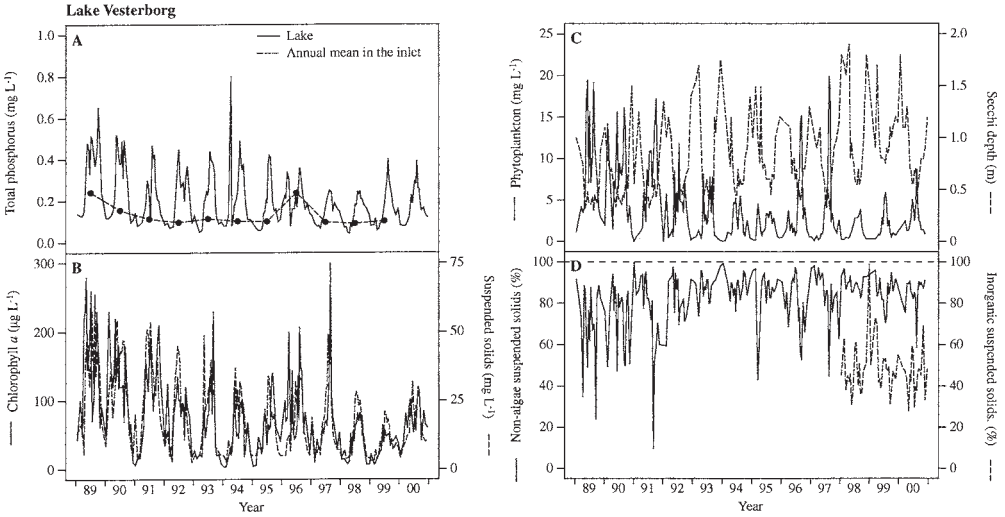



Fig. 3. Lake Vesterborg: (A) discharge-weighted annual mean inlet concentration and lake concentration of total phosphorus, (B) lake concentration of chlorophyll *a* and total suspended solids, (C) biomass of phytoplankton and Secchi depth, and (D) the proportion of nonalgae and inorganic suspended solids of total suspended solids.

ilton and Mitchell 1996), as no macrophyte colonization occurred in Lake Dons, Lake Arresø, or Lake Vesterborg. Only in Lake Arreskov may plants potentially have contributed to reduce shear stress as macrophyte coverage here increased from zero to a maximum of 61% in these lakes in 1997 and thereafter ranged between 1 and 30% (Hansen 2001). The observed major reductions in naorgSS and inorgSS cannot be attributed to low sensitivity of the sediment to resuspension either, as all four lakes have sediment with high content of water and organic matter in the top 5 cm of the sediment sampled at midlake stations. The water content averaged 89, 90, 94, and 95% in Lake Dons, Lake Arresø, Lake Vesterborg, and Lake Arreskov, respectively, and the contribution of organic matter to dry weight averaged 22, 23, 36, and 34%, respectively.

Besides the four lakes analyzed above, 7 of the remaining 11 lakes showed a significant ($P < 0.05$) decline in Chl *a* or SS during the study period (detrended for seasonal effects). We did not find any significant change in the contribution of inorgSS, naorgSS, or phytoplankton biomass to SS with time ($P > 0.05$) for any of these lakes. As could be expected, we did not find any significant ($P > 0.05$) changes with time either in the different fractions for the four remaining lakes without significant changes in Chl *a* or SS.

The relatively close correspondence between the SS and Chl *a* reduction in Lake Arresø is particularly noteworthy,

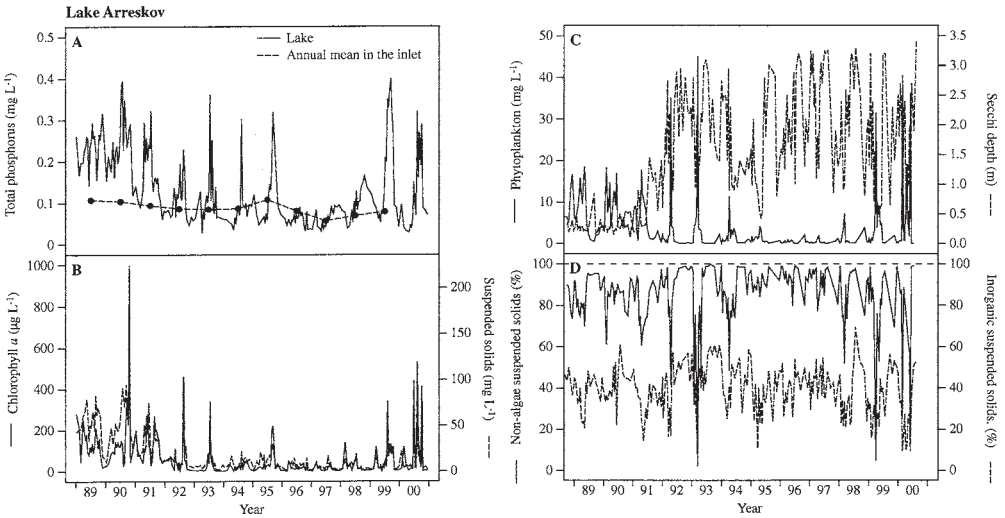



Fig. 4. Lake Arreskov: (A) discharge-weighted annual mean inlet concentration and lake concentration of total phosphorus, (B) lake concentration of chlorophyll *a* and total suspended solids, (C) biomass of phytoplankton and Secchi depth, and (D) the proportion of nonalgae and inorganic suspended solids of total suspended solids. Fish kills occurred in the summer of 1991.

as this lake is large (40 km$^2$) and, moreover, situated in a wind-exposed area in a flat landscape close to the sea. A detailed study of resuspension conducted in 1991 (Kristensen et al. 1992) showed that resuspension occurred on average every second day and that the increase in SS due to resuspension alone could reduce Secchi depths to below 1 m. Nevertheless, even in this lake, rapid reduction of naorgSS and inorgSS occurred concurrently with the observed reduction in Chl *a* (Fig. 4).

Our results therefore suggest that, for Danish lakes, resuspension of sediment even with high organic matter and water content does not prevent a shift to the clearwater state during reoligotrophication following reduction in external nutrient loading. This contradicts the view of Bachmann et al. (1999) but supports suggestions forwarded by Lowe et al. (2001). There may be several reasons for the simultaneous decrease in Chl *a* and naSS. First, a reduction in Chl *a* is accompanied by a decline in the abundance of benthivorous fish (Jeppesen et al. 2002). Consequently, disturbance by sediment-foraging fish is also likely reduced. Bream (*Abramis brama*), which is abundant in Danish lakes, and carp (*Cyprinus carpio*) (although absent from the investigated Danish lakes), may have a substantial effect on the concentration of SS in shallow lakes (Hosper 1997). Second, decreased predation by benthivorous fish may enhance the abundance of benthic invertebrates (Andersson et al. 1978) and thereby indirectly also the consolidation of the sediment mediated by tube-building chironomids that also oxidize the sediment by ventilation. Third, enhanced light penetration of the water stimulates benthic algae production, which, in turn, reduces the risk of resuspension (Paterson 2001). Fourth, the consolidation period between resuspension events is prolonged by the less frequent disturbance of the sediment by fish and by the higher biomass of benthic algae and invertebrates, which enhance the shear stress threshold for resuspension, as has been shown after prolonged consolidation periods with stream sediments (Partheniades 1965). Finally, low phytoplankton production reduces the accumulation of ''new'' detritus in the water. It also reduces sedimentation and thus diminishes the amount of detritus that may potentially be resuspended by fish or wave action.

In conclusion, resuspension of loosely organically rich sediment is apparently not a major factor for delaying the recovery of shallow Danish lakes. We emphasize, though, that our analysis does not include lakes with a high content of silt or humic substances nor does it include very large lakes (>40 km$^2$). Yet the recent results from large (124 km$^2$) and heavily wind-exposed Lake Apopka in Florida (Lowe et al. 2001) suggest that our findings also apply to somewhat larger lakes. Resuspension may, however, indirectly influence the recovery process, as resuspended sediment can release nutrients for phytoplankton growth or sometimes trap nutrients depending on phosphorus adsorption relative to the equilibrium state (Kamp-Nielsen 1974; Søndergaard et al. 1992), just as internal P loading from the undisturbed sediment pool (accumulated during eutrophication) may delay recovery (Sas 1989; Søndergaard et al. 2002).

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