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Dissemination Level

PU	Public	X
PP	Restricted to other programme participants (including the Commission Services)	
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CO	Confidential, only for members of the consortium (including the Commission Services)	

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Non-technical summary

In this deliverable we have evaluated how selected BQE's (Biological Quality Elements) and metrics react to nutrient loading reduction. We concentrate on fish, zooplankton (not a BQE today, but hopefully in the future as it is an important indicator) and phytoplankton. Too few data were available on macrophytes and macroinvertebrate to draw any firm conclusions. The conclusions are as follows:

Fish

- Fish catch-per-unit-effort (CPUE) by numbers and biomass and the contribution of omnivores have been recommended as fish metrics for Water Framework Directive (WFD) assessments. These metrics all seem sensitive to nutrient loading reduction.
- Climate change may in part counteract the response to nutrient loading reduction. Changes in fish assemblage composition, size and age structure during the last decades are profound. A cross system analysis of long time series of fish assemblages in European lakes indicates a decline in cold-stenothermal species, in particular in shallow lakes, an increase in eurythermal species even in deep, stratified lakes, and a decrease in the average size of the dominant species roach (*Rutilus rutilus* L.) and perch (*Perca fluviatilis* L.). This development has occurred despite an overall reduction in nutrient loading that should have favoured fish typically inhabiting cold-water low nutrient lakes and larger-sized individuals.
- Warming will result in a fish-mediated increase in eutrophication that will partly counteract the effect of nutrient loading reduction. This implies that it will be more difficult to obtain the good ecological status required by the WFD in lakes facing climate warming.

Zooplankton

- Zooplankton and cladoceran remains in the sediment are important and cost-efficient indicators of the pressures, structure and function of freshwater lake ecosystems and their ecological status.
- Zooplankton size structure, proportion of large zooplankton, cladoceran size and the zooplankton:phytoplankton biomass ratio are suitable indicators of top-down processes in lakes.
- Zooplankton biomass, the proportion of rotifers by numbers and the proportion of calanoid copepods by biomass are important indicators of bottom-up processes.
- Combination of top-down and bottom-up indicator metrics might yield a solid assessment of trophic conditions in the pelagic of lakes.
- It is strongly recommended that the EU includes zooplankton and cladoceran remains in the surface sediment as central BQEs in the WFD assessments in the future.

Phytoplankton

- Phytoplankton biomass (and chlorophyll *a*) responds to nutrient loading reduction triggering a shift in the relative contribution of eutrophication indicators, such as cyanobacteria and green algae, towards indicators of lower trophic state.

- Genera evenness also responds to nutrient loading reduction, although the direction of changes varies among lakes and year to year variability is high, also in lakes with no change in loading.
- Phytoplankton also reacts to warming, which in part counteracts the effect of nutrient loading reduction.

Objectives

The goal of this deliverable is to summarize results from studies on how BQE's respond to reduced pressures, in particular to reduced nutrient loading. A further aim has been to investigate whether zooplankton is a useful BQE to include in studies of lake recovery.

Fish respond fast to nutrient loading reduction and climate changes

Fish respond to enhanced nutrient loading. Both the biomass and number of fish increase, while the individual size decreases (Jeppesen et al., 2000; Brucet et al., submitted). Moreover, a shift occurs towards less dominance of salmonids and percids and higher dominance of cyprinids, such as roach, carp (*Cyprius carpio* L.) and bream (*Abramis brama* L.), enhancing the predation on large-bodied zooplankton. Fish CPUE by numbers and biomass and the contribution of omnivores have been recommended as fish metrics in WFD assessments. These metrics all seem sensitive to nutrient loading reduction (Jeppesen et al., 2005a,b). A study of 36 European and North American lakes differing widely in depth and size (Jeppesen et al., 2005a, Fig. 1; Table 1) showed a decrease in the catch of fish by either commercial fishermen, anglers or in fish surveys in 82% of the lakes. These decreases were often substantial. In half of the lakes on which quantitative data are available, the reduction exceeded 25%, and reductions >50% were observed in 18% of the cases. In addition, the percentage of potential piscivores, such as pike (*Esox lucius* L.), pikeperch (*Sander lucioperca* L.) and perch increased (and the omnivorous contribution thus decreased) in 80% of the case studies, but the decline was significant in deep lakes only. The density of littoral fish species like tench (*Tinca tinca* L.), Crucian carp (*Carassius carassius* L.) and rudd (*Scardinius erythrophthalmus* L.) increased in importance in some cases, suggesting enhanced plant coverage.

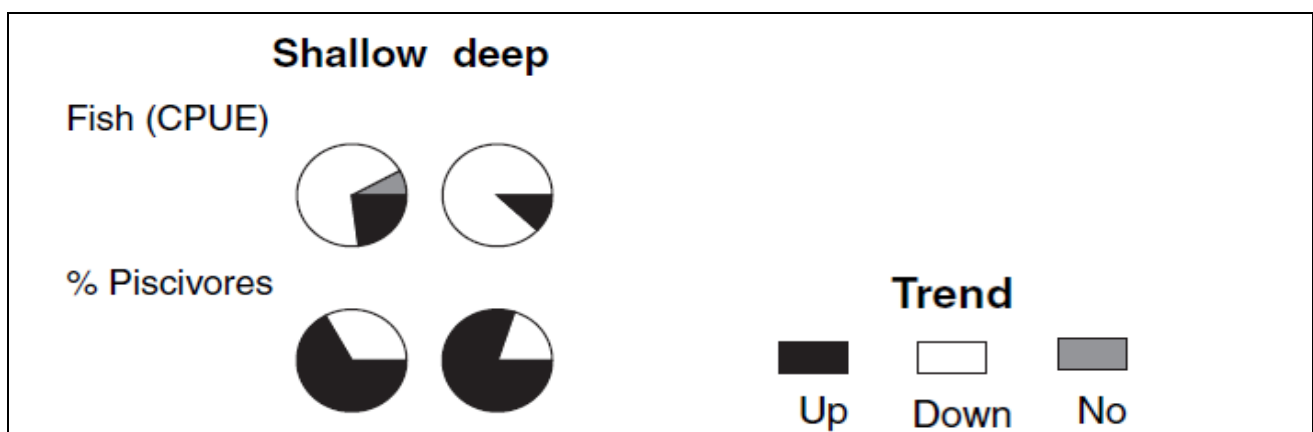


Fig 1. Summary of fish response to nutrient loading of 25 study lakes divided into three categories according to the direction of responses to reduced nutrient loading: increase = up, decline = down, no change = no (from Jeppesen et al., 2005a).

Table 1. Changes in the fish community during recovery of 25 lakes from excessive nutrient loading

Lake no.	Lake name	Fish catch method	Period covered (year)	CPUE	Community change		% Piscivores by weight (including small perch)	
					From	To	From	To
24	Eemmeer and Gooimeer	Trawling	20	-35%	Bream-(roach-smelt)	Bream-pikeperch-roach	5	23
3	Gundsømagle	Gill nets	11	-29%	Roach-crucian carp-tench	Roach-perch-pike-crucian carp	5	29
5	Veleuwemeer*	Trawling	25	-75%	Bream-pikeperch	Roach-perch-bream-pikeperch		-
6	Søgård	Gill nets	10	+43%	Roach-bream-perch	Bream-roach perch	16	23
7	Albufera	Commercial/gill nets	10		Mullets	Mullets	1	
8	Vesterborg	Gill nets	10	-33%	Bream-roach-perch	Roach-bream-perch-pike	9	25
9	Arresø	Gill nets	10	-31%	Bream-roach	Bream-roach-pikeperch	1	32
11	Bagsvaerd	Gill nets	11	-25%	Bream-roach-pikeperch	Bream-roach-perch	18	11
13	Müggelsee	Commercial	10		Bream-roach			
14	Damhussoen	Gill nets	11	+3%	Perch-tench-roach	Perch-tench-roach	46	39
15	Brynup	Gill nets	10	0%	Roach-perch	Roach-perch	26	27
16	Ørnsø	Gill nets	5	-8%	Bream-roach-perch	Roach-bream-perch	19	18
17	Balaton	Gill nets	16	-50%*	Bream-roach			+
20	Leven	Sport fishing		(+)				
22	Peipsi	Trawling	15	-67%	Pikeperch-pike-roach-perch	Pikeperch-ruffe-roach	69	70
24	Tystrup	Gill nets	9	-12%	Roach-bream-perch	Perch-roach-bream-bleak	21	50
25	Tissø	Gill nets	11	+22%	Roach-perch-bream	Roach-perch-bream	15	34
26	Furesø	Gill nets	11	-36%	Roach-perch-bream-smelt	Roach-perch-bream	12	53
27	Ekoln			(-)				
28	Constance	Commercial	25	-	Perch-(whitefish)	(Perch)-whitefish, decrease in perch catches and a shift of perch from pelagic to the littoral zone		
31	Geneva	Commercial	25	-40%	Perch-whitefish-(roach)	Whitefish-perch-(pike)		
32	Ravn	Gill nets	6	-20%	Perch-roach-bream	Perch-roach	48	58
34	Maggiore	Commercial		Major reduction	Whitefish-Bleak	Whitefish-shad, bleak Major decline in whitefish and bleak		
35	Vättern	Commercial	5	-60%	Whitefish-salmonids	Increase in salmonids, major decrease in whitefish		
Lake type		Trend		P-value of chi-square test		P-value of chi-square test		
Shallow		No, +, -		0.02		NS		
		+, -		0.08		NS		
Deep		No, +, -		0.02		<0.0001		
		+, -		0.02		<0.0001		

*Removal of planktivorous fish in addition to reduction in nutrient loading.

Signs in parentheses are trends that must be interpreted with great caution. Lakes 1-22 are shallow, all others are deep. For chi-square tests at the bottom see legend of Appendix 1.

CPUE, catch per unit effort (or changes in catches when the method is 'commercial'); +, increasing during recovery; -, decreasing; empty cells, no data available.

However, climate change may counteract the response to nutrient loading reduction. An analysis of the effect on fish assemblages by climate change and climate variability has been conducted based on long-term (10 to 100 years) data series from 24 European lakes (Jeppesen et al., under revision). Profound changes in fish assemblage composition, size and age structure were found during the last decades as well as a shift towards higher dominance of eurythermal species. The cold-stenothermic Arctic charr (*Salvelinus alpinus* L.) has been particularly affected and its abundance has decreased in the majority of the lakes where its presence has previously been recorded. The harvest of cool-stenothermal trout (*Salmo trutta* L.) has decreased substantially in two southern lakes (Lake Geneva and Lake Maggiore). Vendace (*Coregonus albula* L.), whitefish (*Coregonus lavaretus* L.) and smelt (*Osmerus eperlanus* L.) have shown a different response depending on lake depth and latitude, with a drastic reduction occurring in Estonian Lake Peipsi. Perch was apparently stimulated in the north, with stronger year classes in warm years, but its abundance has declined in southern Lake Maggiore. Where introduced, roach now seems to take advantage of the higher temperature after years of low populations. Eurythermal species such as bream, pike-perch and shad (*Alosa agone* L.) are on the increase. The climate effects have overall been larger in shallow lakes.

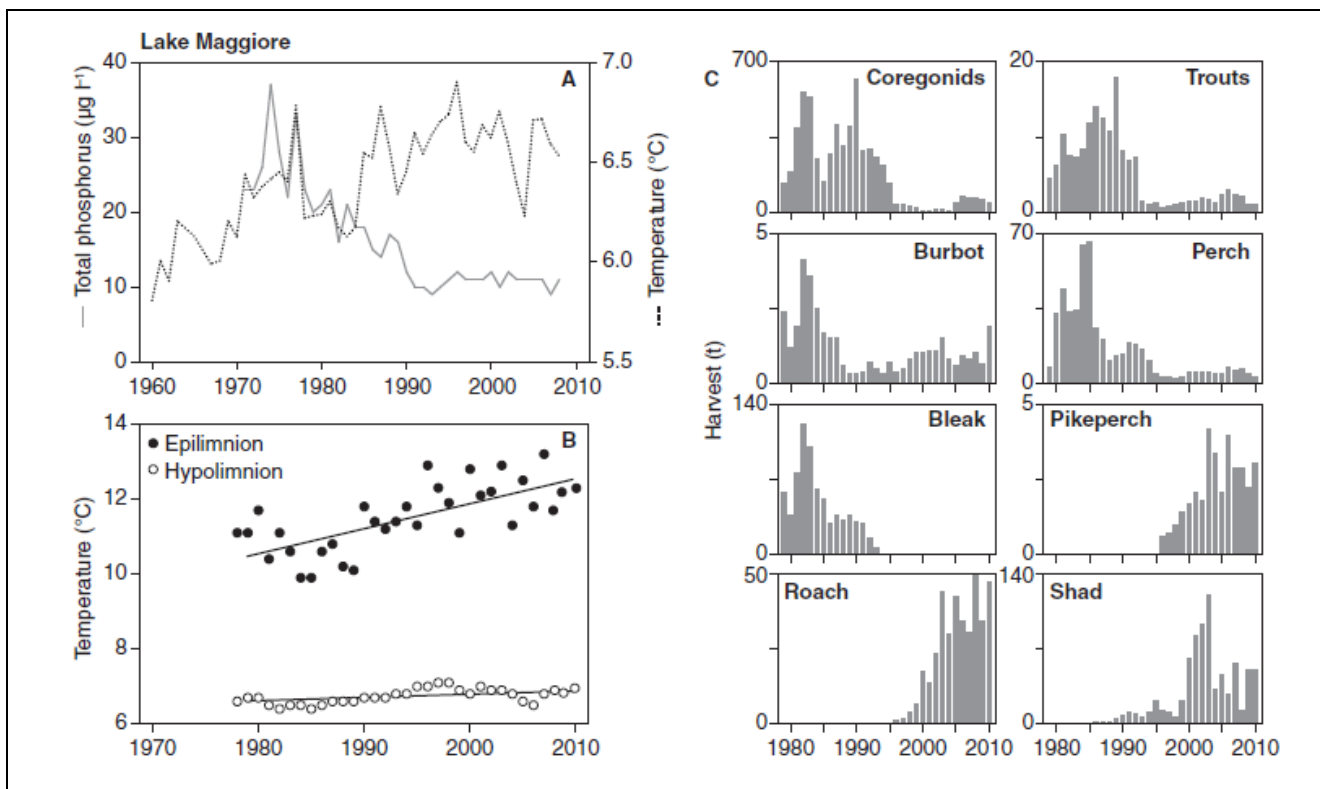


Fig. 2. Changes in Lake Maggiore over 30-50 years. A: Mean water temperature and total phosphorus at the spring overturn. B: Annual mean water temperature of the epilimnion (0-25 m) and hypolimnion (25-360 m). C: Annual commercial harvest of the most important fish species as registered in the fishery statistics by CISPP (from Jeppesen et al., submitted).

In Lake Maggiore, following the decrease in system productivity starting in the 1980s (Fig. 2), the total fishing yield decreased from ca. 50 kg ha⁻¹ to the current 7 kg ha⁻¹. However, fish species contribution to the commercial catches has changed markedly, reflecting the changes in lake fish assemblages, probably triggered by multiple stressors such as increased water temperature, changes in trophy and introduction of invasive non-native species (Volta & Jepsen, 2008; Volta, unpublished data). Coregonid and trout populations have decreased to significantly lower levels than in the past (Fig. 2). Bleak has almost disappeared. In contrast, eurythermal species such as the native shad and roach have begun to appear abundantly in the catches. Also, pike-perch has reached a comparable

catch level to that of perch (Fig. 2), which has declined substantially. Besides these primary commercial species, other tolerant and eurythermal non-native species, such as ruffe and wels catfish (*Silurus glanis* L.), now appear more constantly in the catches (Volta, pers. obs.).

The fish assemblage is not only affected directly by the water warming and changes in the thermal stability of the lakes. Numerous recent studies and reviews indicate that warming will exacerbate existing eutrophication problems, and this will in a self-amplifying manner further stimulate a shift to dominance of eurythermal species, which typically tolerate low oxygen levels and high ammonia concentrations. Also, warming leads to prevalence of small fish. Moreover, reduced ice cover in winter will enhance fish survival (see below), with potential cascading effects within the food web, also reinforcing eutrophication. Therefore, we can expect an allied attack by eutrophication and warming in lakes in the future and the shifts in abundance, size and composition will be reinforced and stimulated by this process (Moss et al., 2011).

The development towards higher abundance of eurythermal fish species and the smaller individuals observed in the fish assemblage and size structure has occurred despite an overall reduction in nutrient loading that should have favoured fish typically living in cold-water low nutrient lakes and larger-sized individuals (Jeppesen et al., under review). This also means that warming will result in a fish-mediated increase in eutrophication, partly counteracting the effect of nutrient loading reduction. This implies that it will be more difficult to obtain the good ecological status required by the WFD in lakes facing climate warming, and the way to (partly) counteract the effect of warming is to reduce the nutrient input to lakes even further than planned under the present-day climate.

Zooplankton as a BQE and indicator of lake recovery

It came as a surprise to many lake ecologists that zooplankton were not included as a BQE in the WFD lake assessment despite the fact that they are considered to be an important and integrated component of the pelagic food web. Using contemporary and sediment samples from particularly Danish, Estonian and British lakes and time series following changes in pressures (eutrophication and top-down control), we have demonstrated that contemporary zooplankton as well cladoceran remains in the sediment have a strong indicator value, and when selecting the right metrics zooplankton are cost-efficient indicators of the trophic state and ecological quality of lakes (Jeppesen et al., 2011).

Zooplankton size structure, the proportion of large zooplankton, cladoceran size and the zooplankton:phytoplankton biomass ratio are suitable indicators of "top-down" processes in lakes, whereas important indicators of "bottom-up" processes could be zooplankton biomass, the proportion of rotifers by numbers and the proportion of calanoid copepods by biomass. The combination of "top-down" and "bottom-up" indicator metrics might yield a solid assessment of trophic conditions in the pelagic of lakes. Time series for lakes in recovery from eutrophication as well as lakes restored by biomanipulation provide further evidence of the strength of zooplankton as strong indicators of changes in pressures. Palaeoecological data further suggest that sedimentary cladoceran assemblages are sensitive to ecological change and are a relatively simple metric summarising a combination of the benthic/pelagic balance of taxa, and size of remains as a measure of fish predation pressure could be a useful predictor of ecological quality.

Do zooplankton respond to changes in the short term?

If a BQE and the associated metrics are good indicators of the ecological state of lakes they should respond to changes in pressures. Here, we provide examples that zooplankton metrics respond to nutrient loading reduction, sudden shifts in ecological state related to fish kills or lake restoration attempts. Furthermore, we provide examples of a gradual change in ecological conditions and a

changing cladoceran community in response to fish stocking and of long-term effects of eutrophication on the ecological structure and function of a shallow lake.

Response by zooplankton to nutrient loading reduction

In a comprehensive study of lakes in recovery from eutrophication, Jeppesen et al. (2005a,b) analysed changes in the seasonal dynamics of zooplankton in eight shallow Danish lakes studied during 13 years (Fig. 3). They found no changes in zooplankton biomass, except for an increase in November and December. However, the biomass of small cladocerans declined during summer and autumn, and the proportion of *Daphnia* to cladoceran biomass increased. The average body weight of *Daphnia* and that of all cladocerans increased. The proportion of calanoids among copepods decreased in summer and the average body weight of cyclopoids and calanoids decreased during summer and in autumn/early winter.

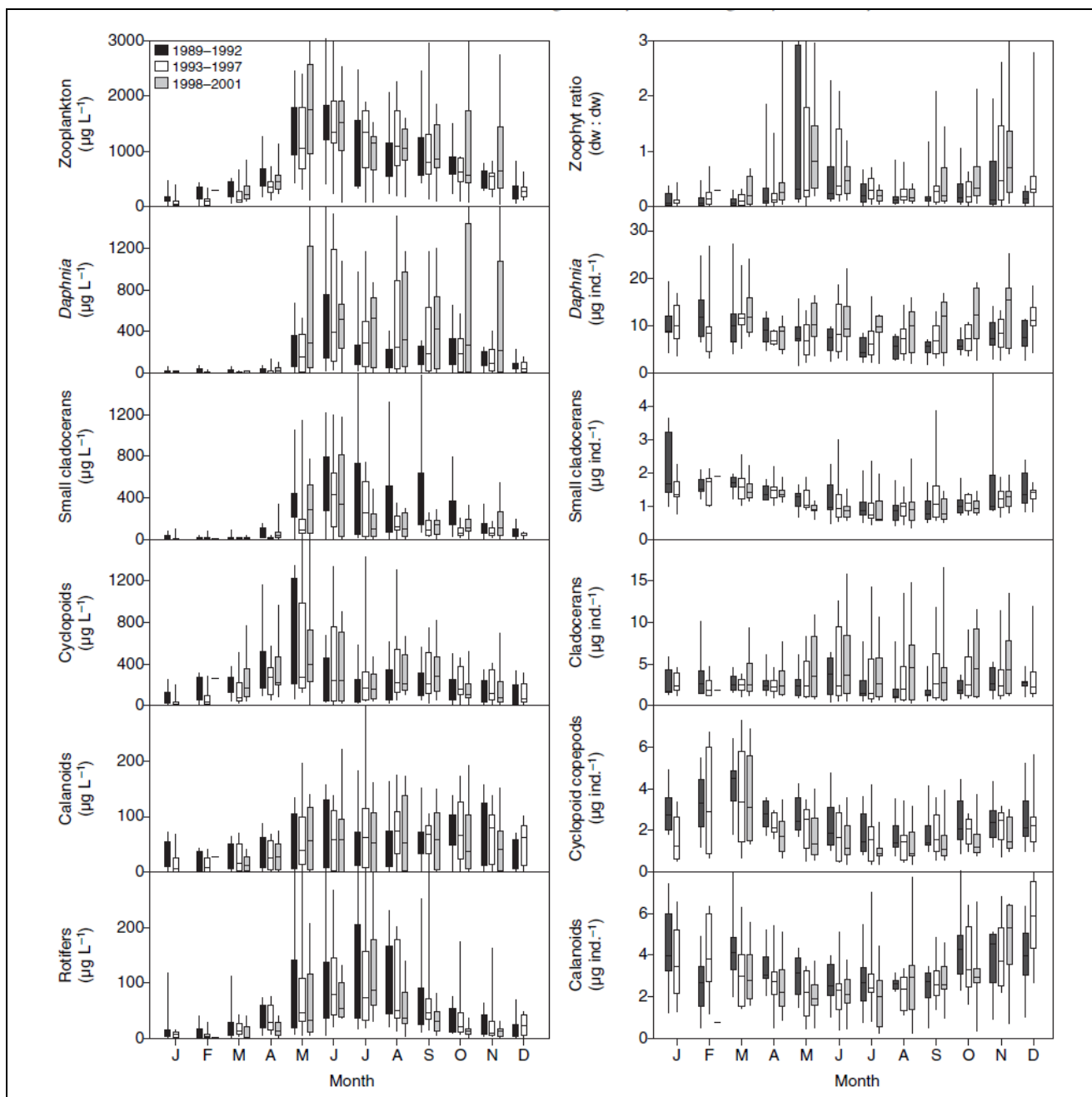


Fig. 3. Box plot (minimum, 25%; median, 75%, maximum) illustrating seasonal variation in the biomass of various zooplankton taxa, the zooplankton to phytoplankton biomass ratio, and the mean body weight of

various zooplankton taxa in eight shallow lakes during three periods. Maximum values are occasionally outside the range of the panels (from Jeppesen et al., 2005b).

Moreover, the zoo:phyto ratio during summer increased, indicating reduced top-down control on zooplankton and enhanced grazing on phytoplankton. Except for the lacking response of the zooplankton biomass, these changes all follow the pattern expected when the nutrient level is reduced from the multi-lake analysis of lakes in different trophic states (Jeppesen et al., 2000), an indication that zooplankton are a sensitive BQE (Fig. 3).

Response by zooplankton to changes in fish abundance

Natural fish kills may substantially influence phytoplankton abundance, composition and biomass and, with it, water clarity. For example, alterations in the duration of ice cover may influence summer plankton due to changes in the proportion of fish that survives the winter/proportion of fish winter survival. In shallow eutrophic lakes covered by ice for up to 5 months a year, fish abundance is typically low due to frequent intense fish kills. Comparative studies of Danish coastal lakes and continental Canadian lakes with similar summer temperatures, but major temperature differences during winter, have shown 4-fold lower chlorophyll *a*: total phosphorus (chl *a*:TP) ratios and higher zoo:phyto ratios in the winter-cold Canadian lakes, perhaps due to lower winter survival of zooplanktivorous fish under ice in the latter (Jackson et al., 2007). Monitoring data from Danish lakes show indications of reduced fish predation in 1996 following the only cold winter with prolonged ice cover (c. 60-90 days) in the monitoring period, 1989-2006. The size structure of the main cladoceran species in the Danish lakes was displaced towards larger size classes in the summer following the cold winter, resulting in a greater grazing capacity on phytoplankton. At community level, zooplankton constituted a larger proportion of larger-bodied taxa during the following summer. Accordingly, phytoplankton biomass (as chl *a*) was lower and grazing (identified as the chl *a*:TP ratio) was higher (Balayla et al., 2010).

Likewise, Ruuhijärvi et al. (2010) showed marked increases in the size of *Daphnia* following recorded fish kills under ice in the winter 2002-2003. They also recorded major concurrent changes in phytoplankton abundance and water clarity, and even more noteworthy also in the composition of phytoplankton, with much less harmful cyanobacteria in the two years following the fish kills. Although not included, the zoo:phyto ratio was also higher in the years after fish kills, demonstrating the value, as for the Danish lakes, of using cladoceran size and zoo:phyto ratios as indicators of alterations in trophic dynamics related to changes in fish stocks.

Response by zooplankton to restoration by biomanipulation

When nutrient loading is reduced, lakes often exhibit a slow response (Jeppesen et al., 2005a,b). In order to reinforce recovery various methods have been used, among which removal of planktivorous fish (termed biomanipulation, Shapiro, 1975; Benndorf, 1995) has been particularly popular. Biomanipulation studies of Danish lakes (Søndergaard et al., 2007, 2008) show the profound effect that fish have on trophic state and WFD selected ecological state indicators, and the importance of including zooplankton as an ecological indicator. Initially, removal of plankti-benthivorous fish resulted in a substantial reduction in chl *a* and suspended matter and improved water clarity. As in the examples of fish kills, major changes also occurred in the phytoplankton composition, and especially the biomass of the harmful cyanobacteria was markedly reduced. While the zooplankton biomass was not affected, both size of *Daphnia* (+) and other (small) cladocerans (-) and not least the zooplankton: chl *a* ratio (+) were affected in many of the lakes (Fig. 4). Following the return of fish after some years to most of the lakes, this tendency was reversed and cyanobacteria returned. Again, cladoceran size and the zoo:phyto ratio appeared to be good indicators of the changes.

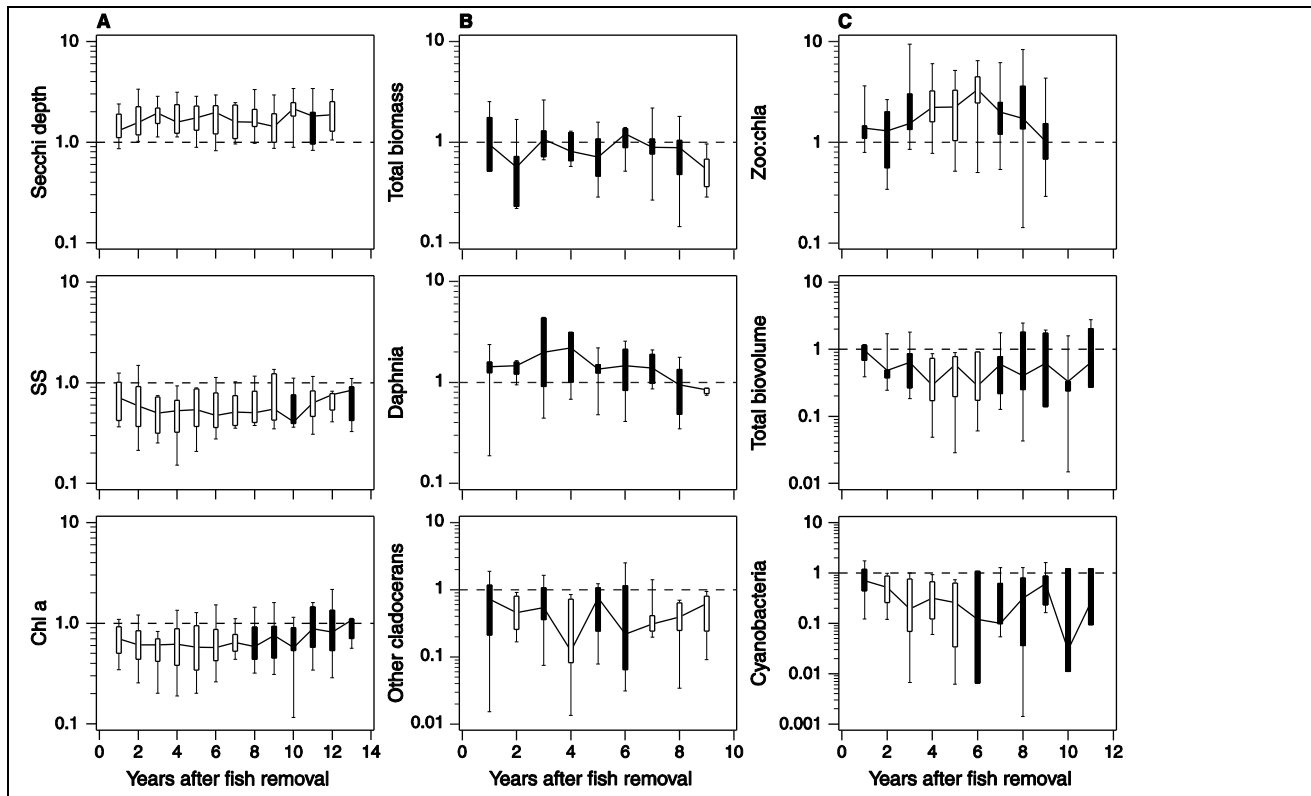


Fig. 4. Response of a number of Danish lakes to biomanipulation during the first 10-14 years after fish removal. The figures change relative to the period immediately prior to the manipulation (1). Light boxes: significant differences from the pre-removal situation. The boxes show 10 and 90% fractiles (outer lines), 25 and 75% fractiles (boxes) (from Søndergaard et al., 2008).

Response by zooplankton to fish stocking

The opposite scenario to biomanipulation is fish introduction. A palaeoecological study at Kenfig Pool, a shallow (max depth = 2.6 m) moderately large (24 ha) lake in an extensive sand dune system in South Wales, provides evidence of the utility of cladocerans in tracking change in ecosystem structure and function associated with fish stocking. The investigation was conducted in order to support the fish management strategy at the site (Sayer et al., 2012). The cladoceran record and the application of the MRT model (Davidson et al., 2010) highlighted two main periods of change, the first around 1850 (45 cm depth) was abrupt and the second, initiated in the 1950s (around 25 cm), was a more gradual change from that point continuing to the present (Fig. 5). Plant macrofossils remains, which have been shown to accurately reflect changes in the dominant components of submerged vegetation (Davidson et al., 2005), were also analysed.

The change around 1850 consisted of a sharp increase in the abundance of charophyte oospores and a pronounced increase in the relative abundance of pelagic cladoceran taxa (in particular *Daphnia* spp.). The cause of these changes is unclear, but a likely candidate is a step change in water depth, caused by human management around 1850, and a large increase in the area of the pelagic habitat. The second point of change starting around 1950 was a shift from dominance of *Daphnia* to *Bosmina*, concurrent with the gradual decline in the relative abundance of *Chydorus sphaericus*. The MRT model infers this post 1950s change as an increase in the abundance of zooplanktivorous fish, which agrees well with documentary evidence of fish stocking events (Sayer et al., 2012). Prior to the 1950s the lake had a fish community consisting of eel (*Anguilla anguilla* L.) and sticklebacks (*Gasterosteus aculeatus* L.). In 1957 3,000 mixed coarse fish were stocked into the lake, and stocking with a variety of species (trout, perch, rudd and even carp) has continued at regular intervals since that time. The MRT model suggests that there has been a gradual increase in predation pressure, as reflected by the

cladoceran community over the past 50 years. The final two changes in MRT group membership from group C at 10 cm to group D at 5 cm and then B at the surface suggest that chl a has increased in the past 10 years, as the average value for group C was $10 \mu\text{g l}^{-1}$ and the mean value for both group D and B is $20 \mu\text{g l}^{-1}$. These recent changes from MRT group B to C suggest a shift, in agreement with the plant macrofossils, from dominance of several charophyte species to the site where fine leaved *Potamogeton* taxa are the most abundant group, which accurately reflects the modern submerged flora of the site (Goldsmith et al., unpublished data).

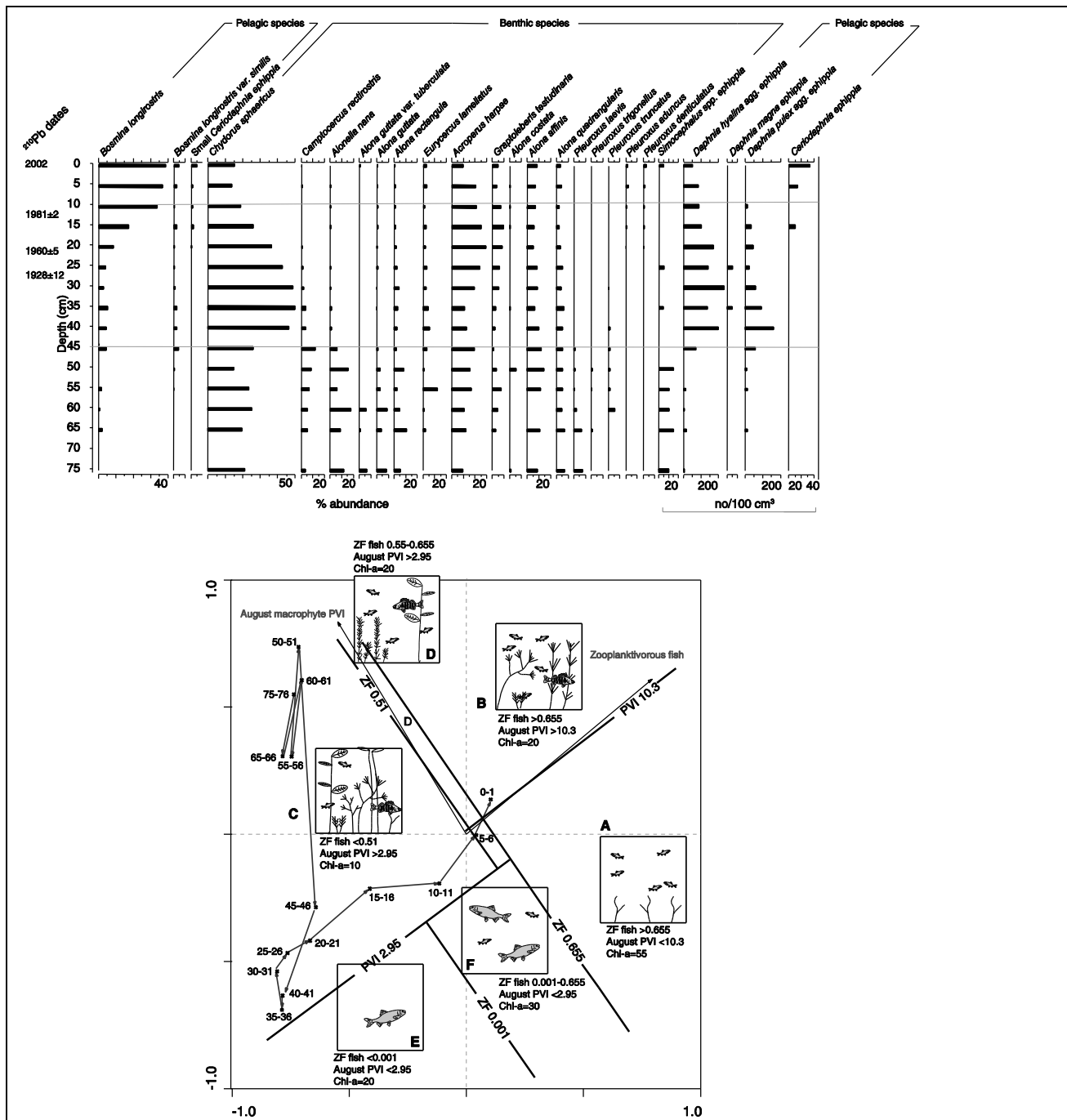


Fig. 5. Above: Cladoceran stratigraphy from Core KENF2 (Kenfig Pool), chitinous remains are expressed as % relative abundance and ephippial remains are expressed as number per 100 cm^3 of sediment. Zones were determined by constrained cluster analysis of both chitinous and ephippial remains. Below: Multivariate regression tree groups superimposed on RDA of surface sediment assemblages from 39 sites with the cladoceran assemblage from core KENF2 passively placed to track changes in macrophyte and fish density (from Davidson, 2006 and Jeppesen et al., 2011).

Thus, in summary, and similar to an example from Felbrigg Lake (Davidson et al., 2010), the changes in the sub-fossil cladoceran assemblage through time at Kenfig Pool reflect variation in the physical and biological elements, such as water depth and the species composition/abundance of macrophytes (Davidson et al., 2010), and also changes in the top down factor predation. Establishing relationships between cladoceran assemblages and their environmental controls from contemporary data has opened up the possibility of disentangling these two drivers of change. Several other studies have also sought to elucidate the effect of fish stocking on lake ecosystems using cladoceran remains in the sediment as proxies (e.g. Verschuren & Marnell, 1997; Buchaca et al., 2011).

The examples illustrate that zooplankton are important indicators of the structure and function of freshwater lake ecosystems and their ecological status. They also clearly show that inclusion of zooplankton may help explaining the variation in chl a at a given nutrient level, but also the variation in risk of cyanobacteria blooming, i.e. some of the metric used in the phytoplankton BQE.

It is acknowledged that zooplankton monitoring can currently be included in schemes of operational monitoring (for waterbodies identified as being at risk of failing to meet their environmental objectives, and for those into which priority list substances are discharged) and in investigative monitoring (if the reason for deviations is unknown, to ascertain the causes of a waterbody or waterbodies failing to achieve the environmental objectives, or to ascertain the magnitude and impacts of accidental pollution). However, to minimise the use of resources, there is a high risk that the policy makers and managers follow the minimum requirement policy and therefore will not include zooplankton in their monitoring programmes. We, therefore, strongly appeal to the relevant EU authorities to consider (and include) zooplankton as an obligatory BQE. We also see the omission of zooplankton as a loss of opportunity for transitional waters and large rivers. The focus mainly on ecosystem structure and less on function in the WFD must be reconsidered, and we have shown that zooplankton are a key element here for understanding lake ecosystem function – and perhaps also for large rivers and transitional waters. Further exercises are needed to develop metrics at the regional level throughout Europe as for other BQE's. So far, the most promising metrics based on contemporary samples are zooplankton biomass, the proportion of rotifers by numbers, the proportion of large zooplankton, zooplankton size, cladoceran size, and the proportion of calanoid copepods; and for surface sediment: size and the proportion of large forms of resting eggs and the proportion of pelagic cladoceran remains.

Response by zooplankton to temperature change

Temperature induced changes in fish structure and size may also affect the zooplankton through predation (Jeppesen et al., 2010a,b). A multiple regression analysis made for August data from Danish lakes demonstrated a decrease in the average size of cladocerans and copepods with increasing temperature (Fig. 6).

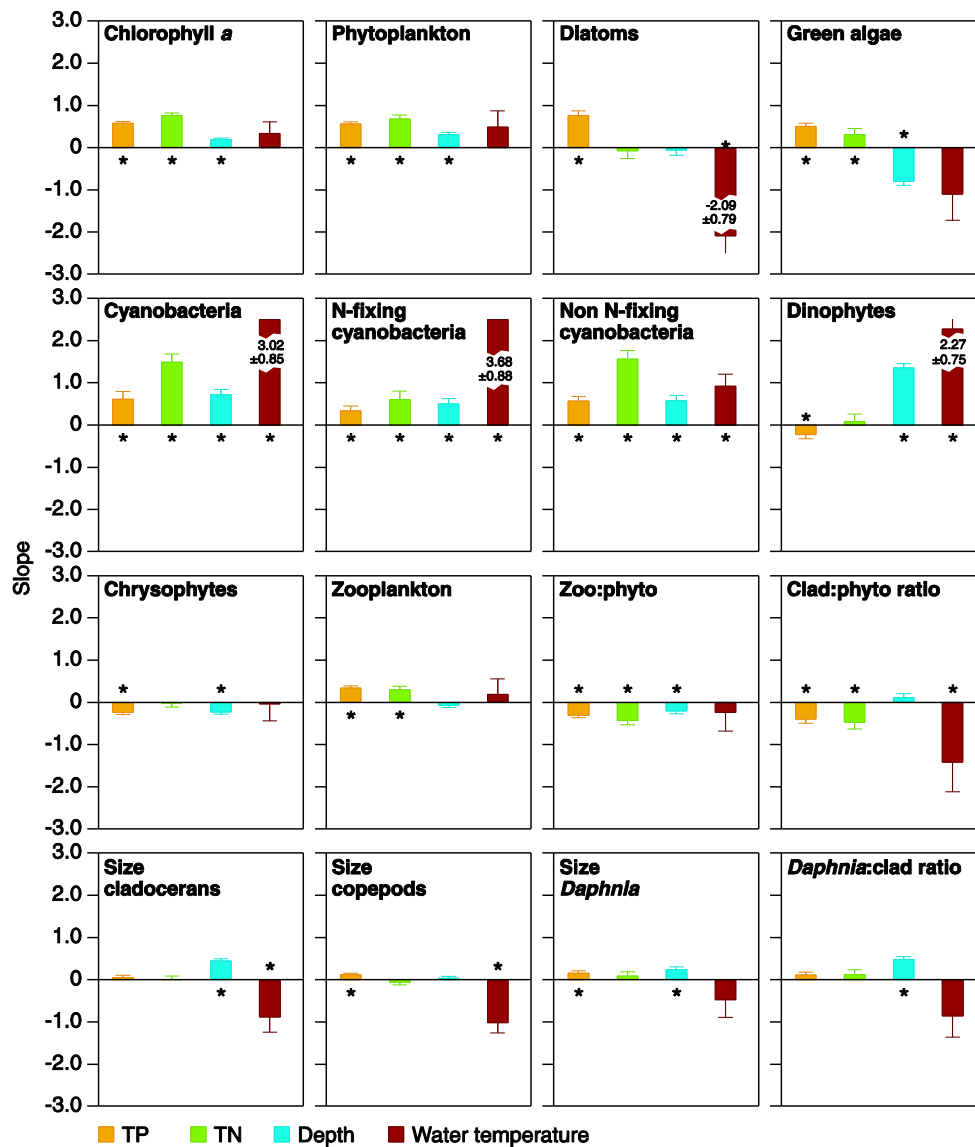


Fig. 6. Multiple regressions between different plankton variables (\log_e -transformed) and total phosphorus, total nitrogen and water temperature of the surface layer and lake mean depth – all \log_e -transformed. When the value is positive, there is a positive effect of a given variable (when including also the other variables) and the opposite, if negative. All data are from August. Significant values are marked with an * (modified from Jeppesen et al., 2009).

This usually suggests enhanced predation by fish. A tendency to a decrease in the zoo:phyto biomass ratio and the proportion of *Daphnia* among the cladocerans provided further evidence of higher fish predation. With a lower proportion of large-sized *Daphnia* and a lower average size of zooplankton, grazing on large-bodied phytoplankton is likely to decline, which will further enhance the risk of dominance by filamentous cyanobacteria. This shift in zooplankton size is likely due to changes in the composition of fish stocks with higher dominance of zooplanktivorous and omnivorous fish, implying increased predation on zooplankton.

Phytoplankton response to nutrient loading reduction

Phytoplankton responds to nutrient loading reduction. A study of long time series in 35 lakes in Europe showed major changes in the majority of the lakes (Fig. 7 and Jeppesen et al., 2005a)

Total phytoplankton, cyanobacteria and green algae decreased in a substantial part of the lakes, while other algae classes increased (Fig. 7).

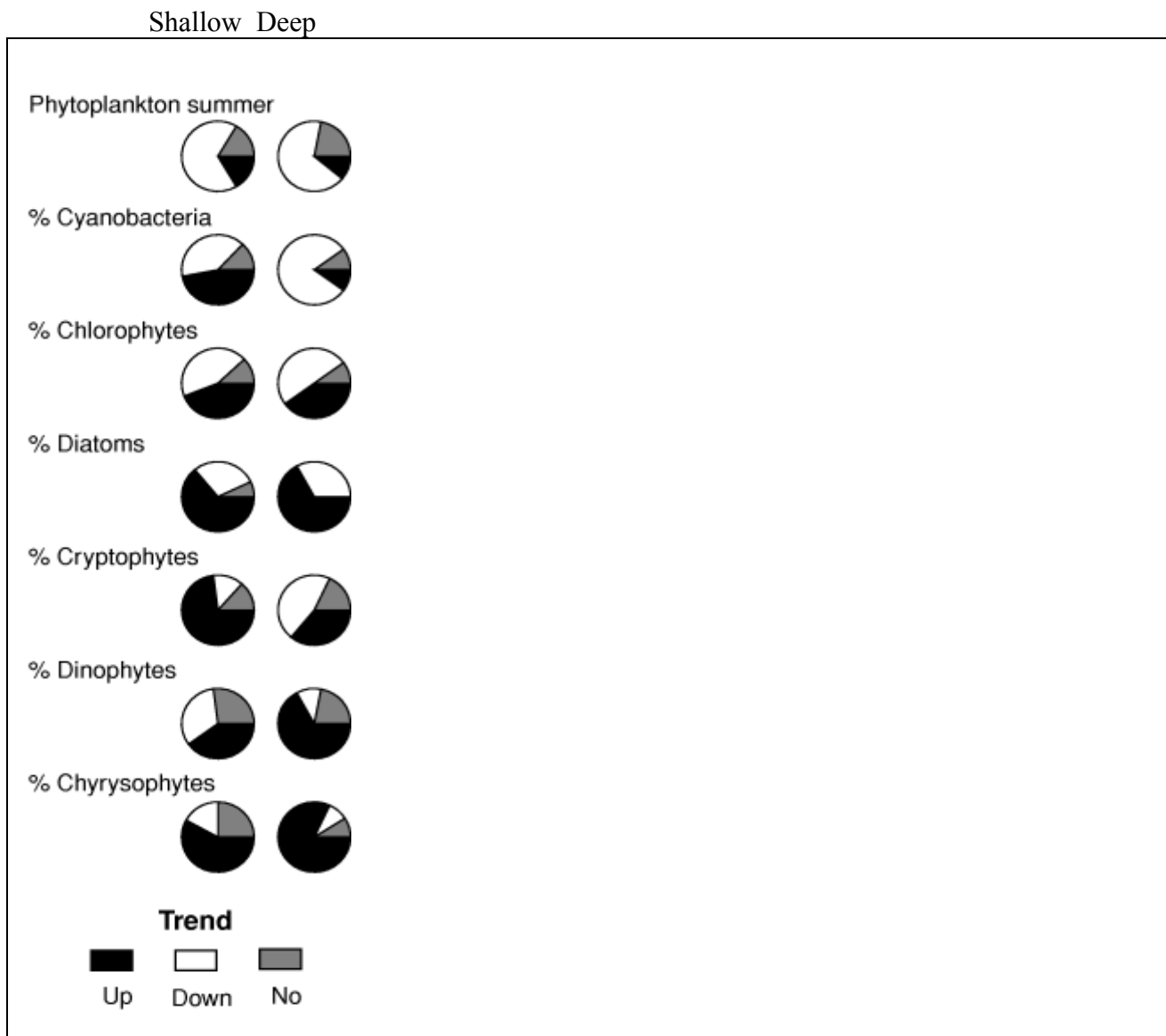


Fig. 7. Summary of phytoplankton response to nutrient loading of all study lakes divided into three categories according to the direction of responses to reduced nutrient loading: increase = up, decline = down, no change = no. (From Jeppesen et al., 2005a).

A detailed study including also seasonal dynamics has been conducted in 8 shallow Danish lakes covering the period 1989-2001 (Jeppesen et al., 2005a,b) and a more comprehensive study including an additional 10 years is underway (Özkan et al., in prep.); a few examples are from these study are presented below:

The 12-year study of Danish lakes (Jeppesen et al., 2005)

Phytoplankton from 8 shallow Danish lakes was analysed. The total phytoplankton volume decreased particularly in spring and early summer and tended to be lower in November and December as well, whereas no significant changes were observed in late summer (Fig. 8). This pattern agrees well with the recorded changes in the dynamics of TP and orthophosphate concentrations (Søndergaard et al., 2005), indicating that nutrient constraints played a key role in

determining the change in the seasonal response. The TP reduction was particularly strong in winter, followed by spring and autumn, while it was less pronounced in summer, largely reflecting changes in the seasonal dynamics of internal P loading (Søndergaard et al., 2002).

In six of the eight Danish lakes analysed here, a decline in diatom dominance was observed during spring (Fig. 7 and table in Jeppesen et al., 2005a,b). Phillips et al. (2005) also showed strong effects on diatoms in spring in shallow Barton Broad, U.K. In Danish lakes this decline occurred without affecting the silicate concentration (seen as a major dip in spring), which is likely to reflect an enhanced benthic production of diatoms mediated by the improved light conditions indicated by the higher Secchi depth. Higher benthic production reduces phosphorus release (because of higher uptake by benthic algae and thus a higher redox potential; Hansson, 1989), which in a self-amplifying manner may further reduce phytoplankton growth and promote benthic algal production. Later in the season when the temperature increases, release of P from the sediment is inevitable and a return to mainly pelagic production occurs (Liboriussen & Jeppesen, 2003). This probably boosts nutrient release further, and thus summer phytoplankton biomass would, as seen in the present study, only be weakly affected during the first phase of oligotrophication until P in surficial sediments declines. Enhanced grazing may also have played a role as the zoo:phyto ratio increased significantly during March to June and again in November and December, accompanied most likely by an increased grazing pressure on the phytoplankton. The Danish results along with those of others (Phillips et al., 2005) show that the strongest response to a loading reduction in shallow lakes in the early recovery phase is to be expected in spring, while the response of summer plankton is most prone to delay. Unfortunately, most sampling programmes are conducted mainly during summer, which renders it difficult to identify improvements during the early recovery phase.

Besides diatoms, cyanobacteria declined significantly in spring in the Danish lakes, while heterocystous cyanobacteria increased significantly during July and August and tended to increase also in September and October. These changes occurred despite an increasing total nitrogen (TN):TP ratio in the inlet (TP reduction larger than TN decline (Table 2 in Søndergaard et al., 2005), indicating that it is P rather than nitrogen (N) (and the capability of nitrogen fixation) that determines the shift, as has earlier been suggested by Jensen et al. (1994) for shallow Danish lakes. The succession from dominance of non-heterocystous to heterocystous cyanobacteria in summer at reduced TP concentrations has been demonstrated in several other studies of lakes undergoing recovery. Phillips et al. (2005) observed a decline first in non-heterocystous cyanobacteria and next, more slowly, in heterocystous genera after 15-20 years. The results also suggest a decline in heterocystous genera from the second to the third period (Fig. 8). Köhler et al. (2000) observed a similar response, although in that case inlet TN concentrations decreased more than did inlet TP concentrations.

In the Danish lakes chrysophytes increased during summer and autumn in the lakes with the lowest initial nutrient level, which is to be expected given that phagotrophic chrysophytes are typically important inhabitants of lakes with low TP concentrations (Kalf & Watson, 1986). Also the increase in cryptophytes in late summer is expected from the phytoplankton-TP pattern described by Watson et al. (1997) and Reynolds (1987).

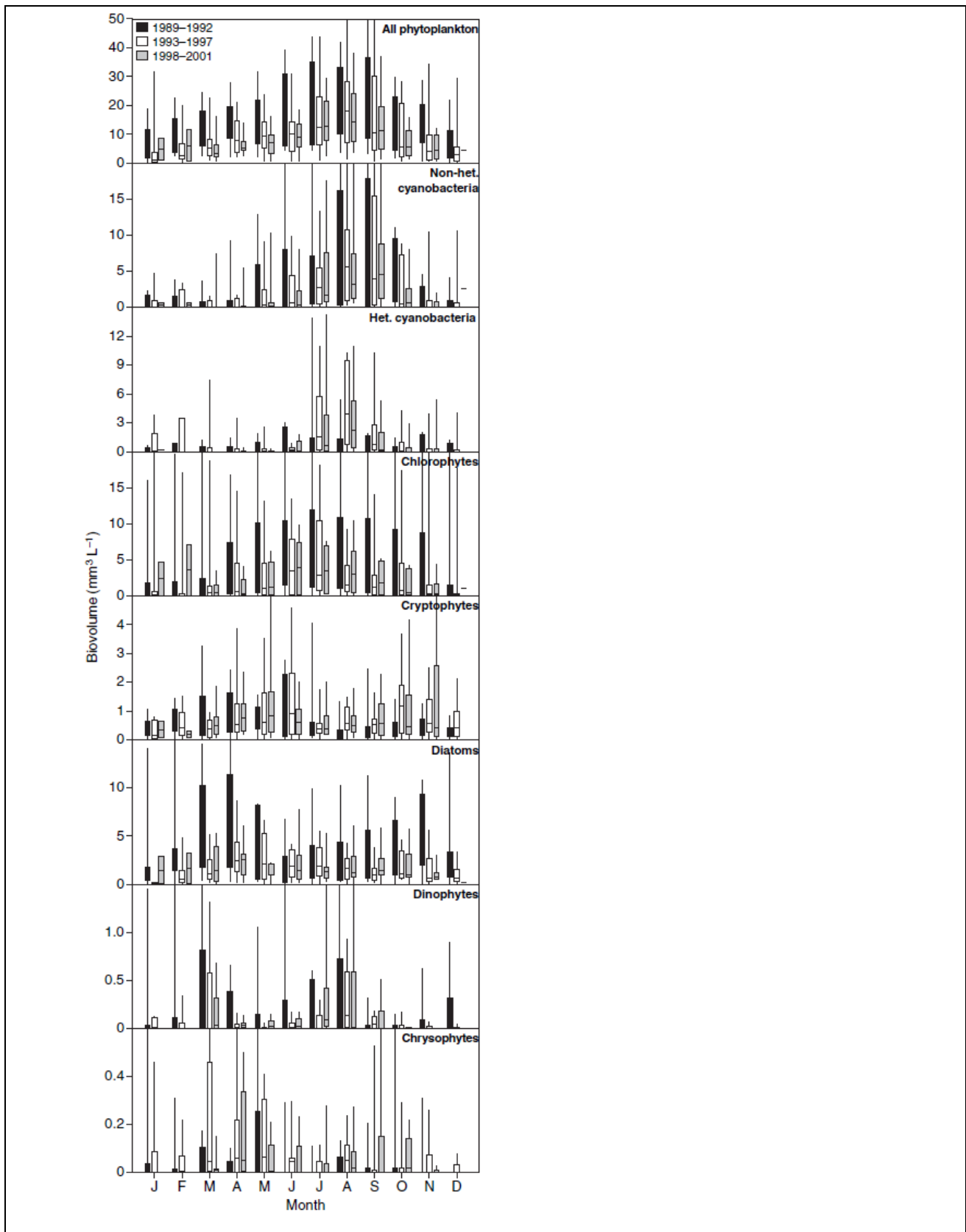
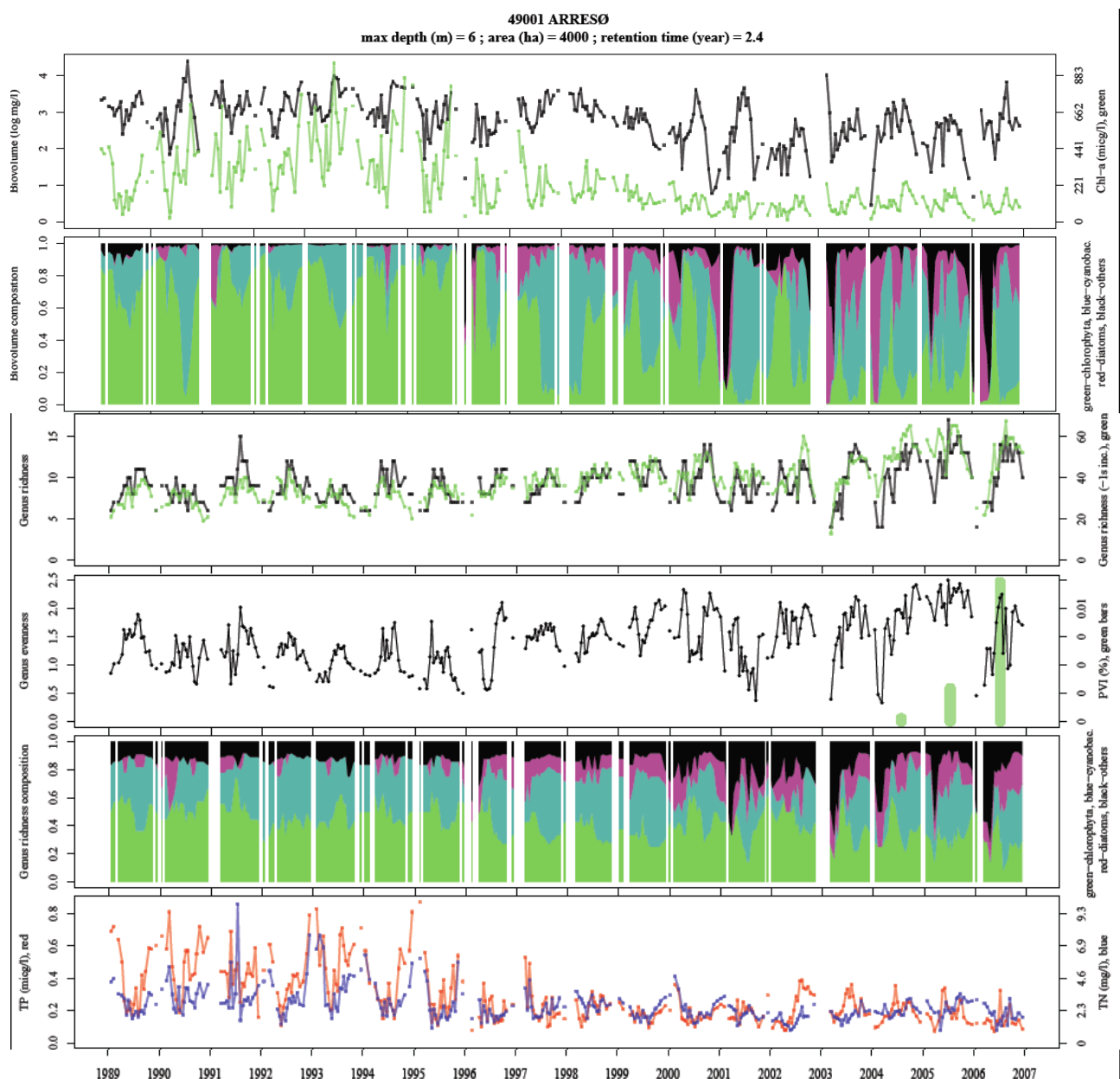


Fig. 8. Box plot (minimum, 25%; median, 75%, maximum) of seasonal variation in the biovolumes of various phytoplankton taxa in eight shallow and four deep lakes during three periods. Note the change of scale between phytoplankton taxa of shallow and deep lakes. Maximum values are occasionally outside the range of the panels.

The 20-year study of Danish lakes (Özkan et al., unpublished results)

Here we show two examples from a lake subjected to a major reduction in nutrient loading (Lake Arresø) (Fig. 9) and a lake included as a reference (no significant change in loading) (Lake Holm (Fig. 9). In Arresø loading reduction has resulted in a major reduction in chl a and a reduction – though more moderate – in phytoplankton biovolume. A major shift has also occurred in the phytoplankton composition from dominance of green algae all year around to higher importance of cyanobacteria and diatoms (a typical shift for shallow hypertrophic Danish lakes, Jensen et al., 1994). Moreover as demonstrated above, seasonality has changed markedly, with less phytoplankton in spring and autumn. Genus richness and genus evenness have increased, also here exhibiting stronger seasonality than before, i.e. a clear response of some of the key metrics recommended by WISER. By contrast, no clear trends for most variables are observed for the reference lake, Lake Maglesø.



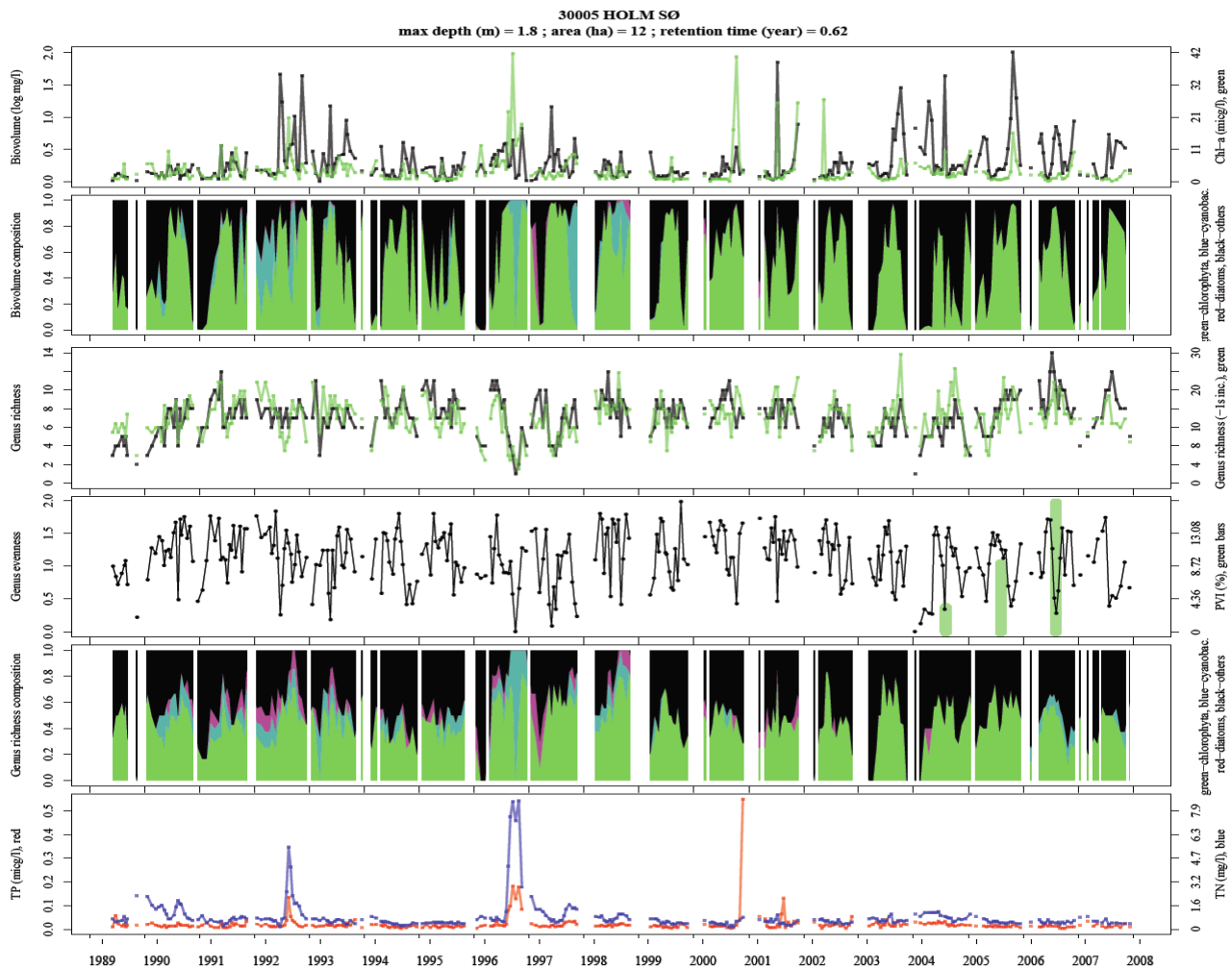
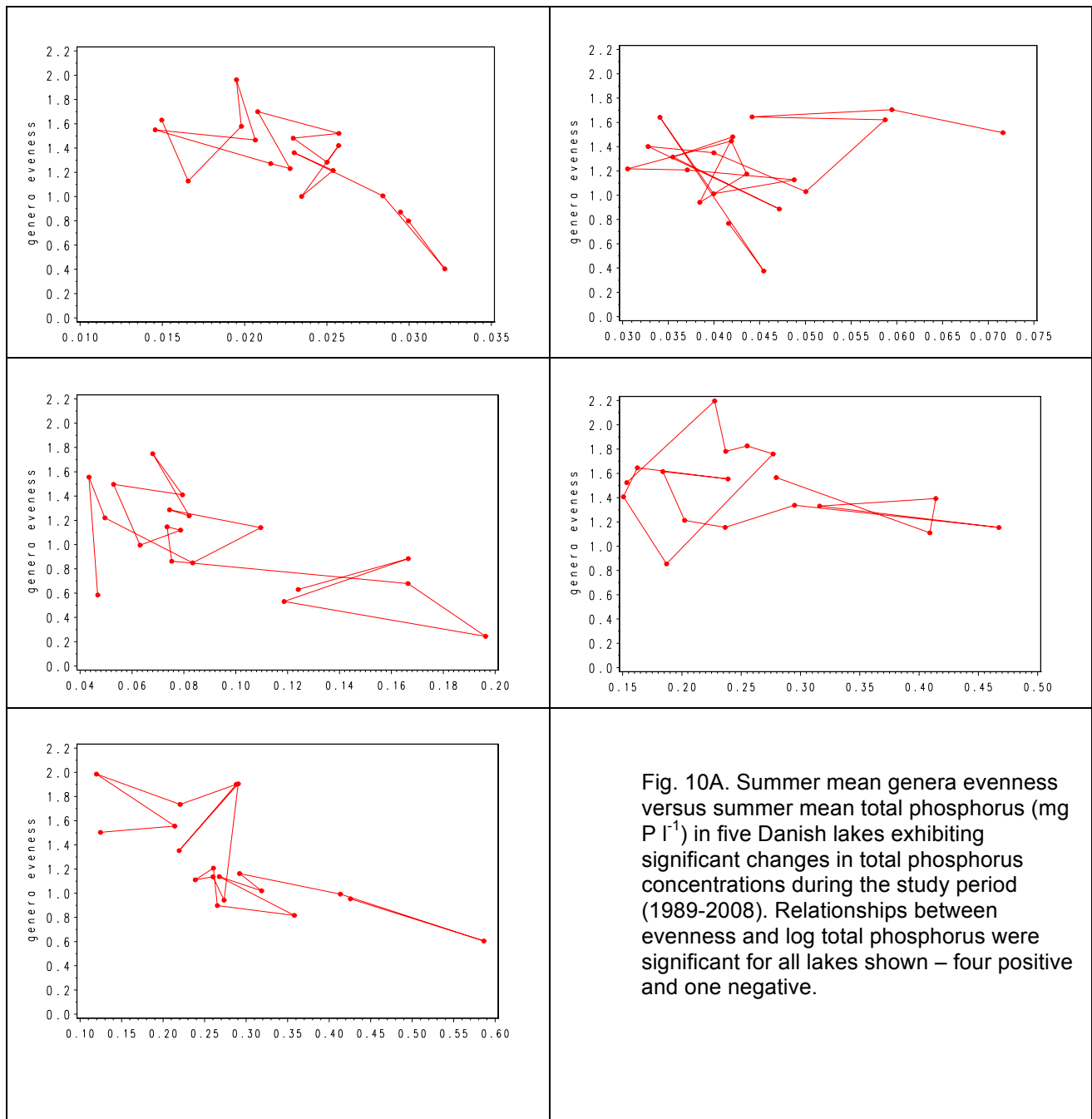
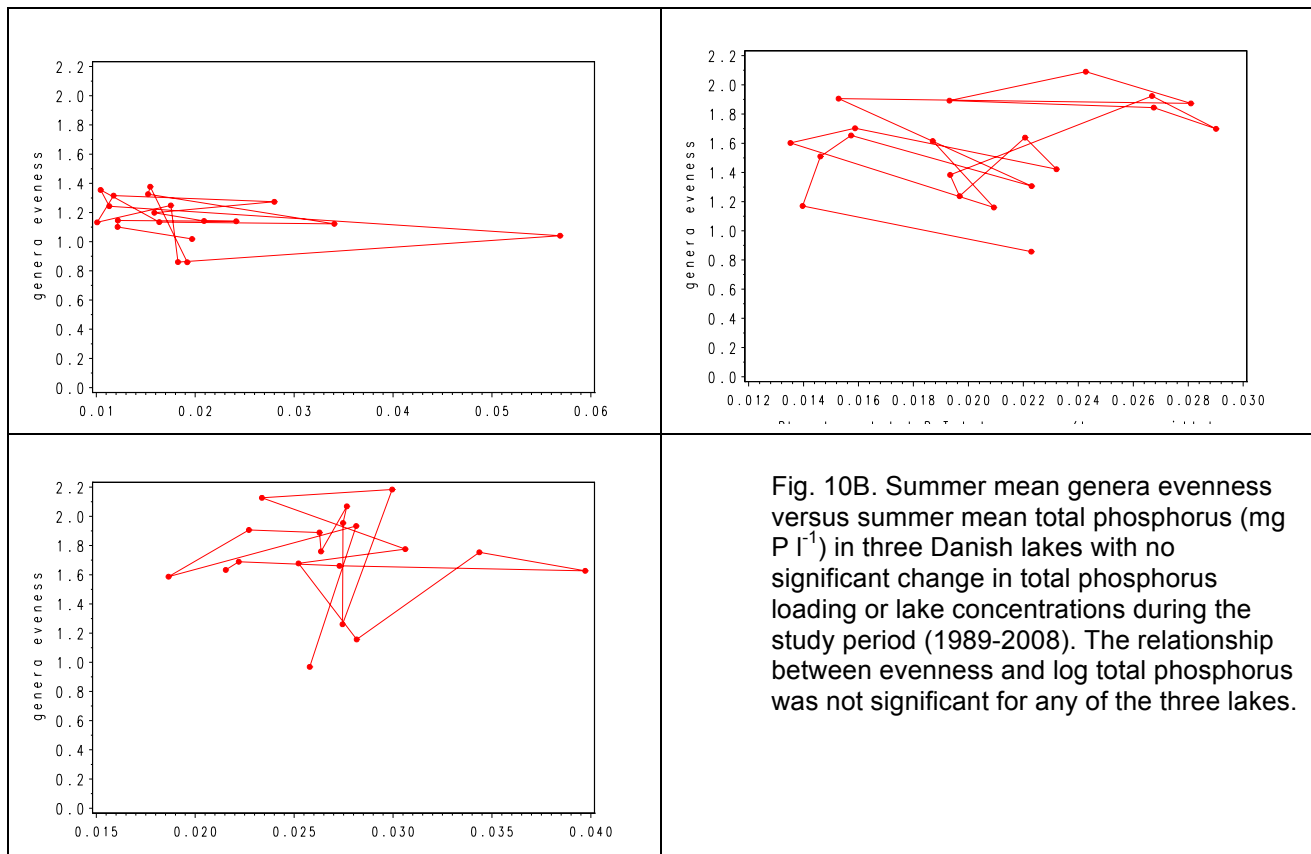


Fig 9. Changes in several phytoplankton metrics and total phosphorus phytoplankton in Lake Arresø subjected to major reductions in external nutrient loading during the past 20 years and Lake Holm with no changes in loading during the same period (Özkan, unpubl.).

Evenness is reacting significantly ($p < 0.05$, $n = 20$) to reduced TP concentrations in a number of Danish lakes (Fig. 10A), though the direction of change is not always the same. Most of the lakes show enhanced evenness, but one lake showed an opposite trend. Moreover, the year to year variation is high not only in these lakes, but also in lakes that have not been subjected to changes in nutrient loading or phosphorus concentrations (Fig. 10B). However, no change in evenness has occurred in the latter during the study period. Unless long time series are available or a large set of lakes are surveyed simultaneously, it therefore seems difficult to use this metric as a single indicator of change in ecological status.





Phytoplankton metrics sensitive to climate

Phytoplankton metrics are also sensitive to changes in climate. A multiple regression analysis of data from 250 Danish lakes sampled in August showed higher dominance of cyanobacteria in terms of biovolume, most notably potential N-fixing forms and also of dinophytes at higher temperatures (Jeppesen et al., 2009). There was also a tendency to increasing chl *a* concentrations and phytoplankton biovolume, while concomitantly diatoms became less important (Fig. 6). Thus, the risk of dominance by potentially toxic cyanobacteria will increase with warming, and the period when blooming cyanobacteria occurs will likely be longer (Romo et al., 2005; Blenckner et al., 2007) (Fig. 6). However, we did not find mean summer evenness to be related to temperature, when first accounting for total phosphorus.

The eutrophication-climate interaction has also been studied in a set of European lakes using a linear mixed effects chlorophyll *a* (chl *a*) model fitted to 461 European lakes (for more details see Deliverable 5.2-5/6). The effect of TP, TN and water temperature on chl *a* concentrations varied within the WFD affiliated lake types. The data structure was three-way nested as in every lake type there were several lakes and from every lake multiple chl *a* samples were taken. By using the linear mixed effects model for nested data we can substantially decrease the variation in data by selecting properly both the fixed effects and variance structure. For predicting the phytoplankton response to the reduction of nutrient load and climate change, a chlorophyll *a* model was developed and included in the LakeLoadResponse (LLR) Internet tool. The best linear mixed effects model for these data according to Akaike's Information Criterion AIC, likelihood ratio tests and residual analysis is the model that has random intercept and slopes for types and random intercept for lakes. The final chlorophyll *a* model is of the form:

$$\text{chla}_{ijk} = \underbrace{\text{totP}_{ijk} + \text{totN}_{ijk} + \text{temp}_{ijk}}_{\text{fixed effects}} + \underbrace{u_k + u1_{jk} + u2_{jk} + u3_{jk}}_{\text{randomeffectsof types}} + \underbrace{v_{jk}}_{\text{Randomeffectsof lakes}} + \underbrace{\varepsilon_{ijk}}_{\text{error term}}$$

chla_{ijk} is the log scale chlorophyll *a* concentration from sample *i* from lake *j* of lake type *k*
 Ltotp_{ijk} is the log scale total phosphorus concentration from sample *i* from lake *j* of lake type *k*
 totn_{ijk} is the log scale total nitrogen concentration from sample *i* from lake *j* of lake type *k*
 temp_{ijk} is the temperature from sample *i* from lake *j* of lake type *k*
 u_k is the random intercept of type *k*, allows for variation between the lake types, normally distributed with mean 0 and variance σ_{type}^2
 $u1_{jk}, u2_{jk}, u3_{jk}$ is the type specific random slopes for totP, totN and temp
 v_{jk} is the a random intercept of lake *j* of type *k*, allows for variation between the lakes, normally distributed with mean 0 and variance σ_{lake}^2
 ε_{ijk} is the model error term.

The hierarchical chlorophyll *a* model with temperature effect was implemented in the LakeLoadResponse (LLR) tool (<http://lakestate.vyh.fi/>). On the entire dataset the temperature effect on chl *a* in different lake types in July-August was not significant as it could have been anticipated. However, there were lake types that have significant temperature effect, such as the Northern GIG lake types (LN3a, LN3b and LN8a). An example is given in Fig. 11.

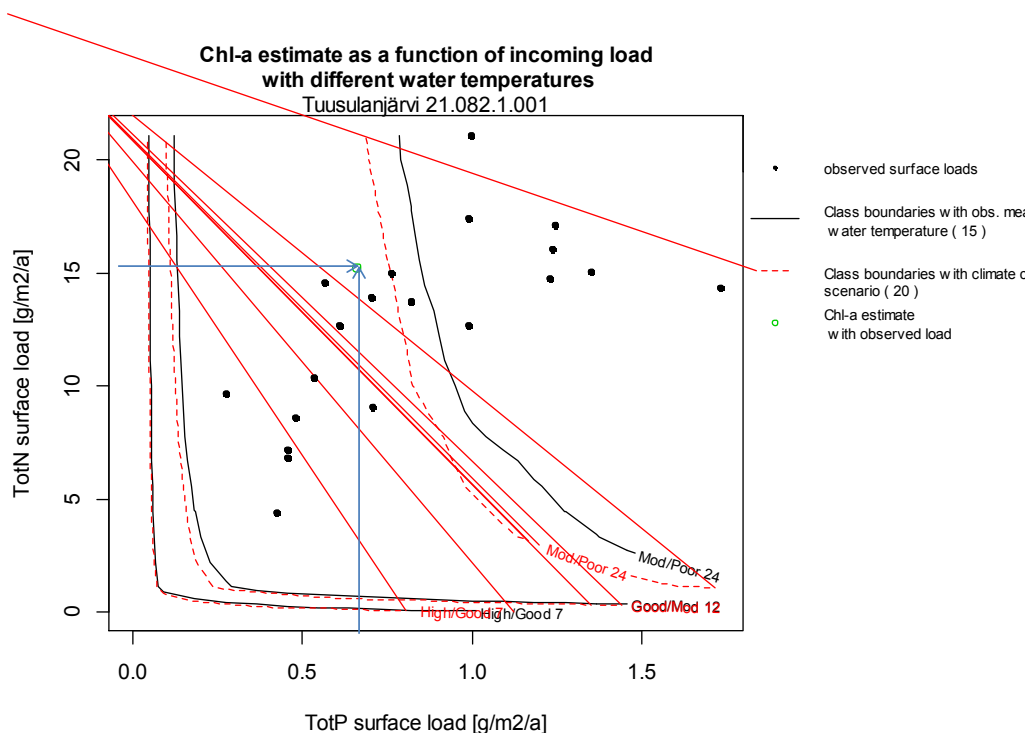


Fig. 11. Estimate for chlorophyll *a* concentration in Lake Tuusulanjärvi as a function of phosphorus and nitrogen loading ($\text{g/m}^2/\text{a}$) to the lake for the present temperature situation (black solid line) and for 5°C warmer water (red dashed line).

Bayesian network (BN) modelling

In this modelling study, we have focused on the combined impacts of restoration and climate change on ecological status based on phytoplankton (chlorophyll *a*). This model has applied data

from WISER WP3.1 (Lakes phytoplankton) from all of Europe, of which 337 lakes (types L-N2a and L-N5) were used for this analysis. We considered only climate impacts directly on lake processes and leave out potential climate impacts on river basin processes (such as water discharge and nutrient transport). In addition, we modelled the effect of lake restoration in terms of reduced P loading. A Bayesian network (BN) modelling approach was used for this study (see Deliverable 5.2-6 for details). The predicted levels of nutrients and chl a in this model exercise depend on many assumptions, of which not all can be justified. The most relevant result is therefore not the absolute probabilities, but the changes in probabilities (percentage points) across climate and restoration scenarios (Fig. 12). Since the ecological classification of lakes depends on lake type, two common lake types of Northern Europe were selected as examples (L-N2a: altitude <200 m; L-N5: altitude 200-800 m). The study considered altogether 9 scenarios: 3 levels of restoration (no change; -20% P loading; -40% P loading) combined with 3 levels of climate change (no change; +2 °C air temperature; +4 °C air temperature).

TP status class responded to restoration (reduction of P loading) by increased probability of High status. The highest restoration level resulted in a 30 percentage point (pp) increase in the probability of High status for lake types L-N2a, and 32 pp increase for L-N5. TP was not affected by the climate scenarios, which is in accordance with the model settings. Chl a status class also responded to reduction of P loading by an increase in the probability of High status, although to a lesser degree than for TP (L-N2a: 10-12 pp increase; L-N5: 8-11 pp increase). At the same time, the risk of less-than-good status was reduced by 14-19 pp. Chl a was also impacted by climate change: +4 degrees resulted in an 18-20 pp reduction of High status probability for L-N2a, and 22-23 pp reduction for L-N5. The increase in risk of less-than-good status, however, was only 1-7 pp in this model.

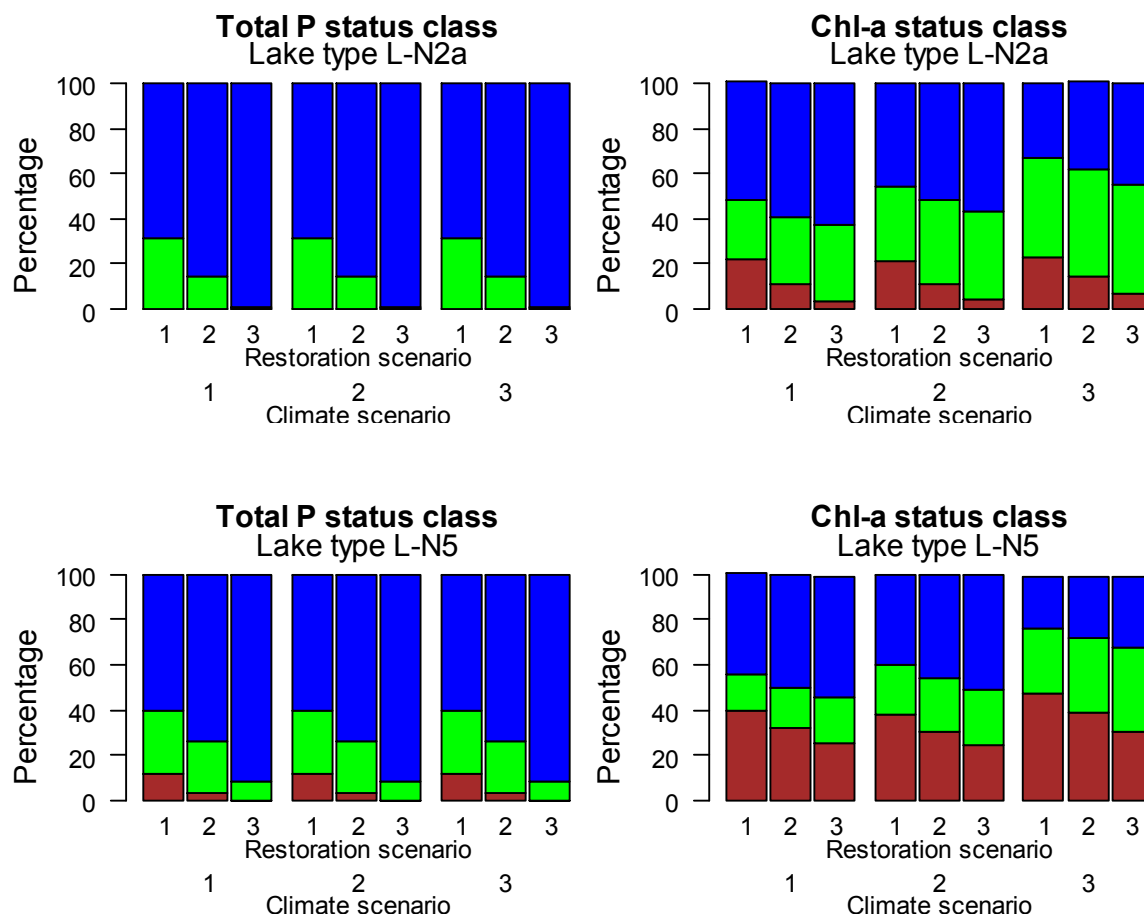


Fig 12. Probability distribution of status classes for total phosphorus (upper panel) and chlorophyll a (lower

panel), for low-altitude (left) and high-altitude (right) lakes, respectively. Each plot shows the outcome of the 3x3 scenarios for Restoration and Climate change levels.

Deterministic dynamic model of Lake Veleuwe, The Netherlands

Further studies have been done on shallow Lake Veleuwe, The Netherlands, using a deterministic water quality and primary productivity model DELWAQ-G - BLOOM. Different scenarios were investigated for both long term (1976-1993, DELWAQ-G – BLOOM) and short term modelling applications (1985, stand alone BLOOM) (model description see WISER Deliverable 5.2.1).

For the long-term simulation, the effects of water management measurements were assessed by hind casting the lake's water quality both with and without the implemented measures of an increased flushing regime and the increased removal of phosphate of the WWTP that discharges in the lake. Additionally, the effect of increased temperatures (deduced from climate scenarios of the Royal Dutch Meteorological Institute (KNMI) and set to +0.9°C (G) and +2.6°C (W+)) was superposed on the original run and on the run with no increased efficiency of the WWTP to investigate whether or not increased temperatures are of importance in comparison with the taken measure.

It was evident that increased efficiency of WWTP has improved water quality more than the increased flushing measure. The effect of increased temperatures has less impact on water quality compared to the two water management measures taken. Next to the difference in the extent of effect between the two water management measures on chl a concentrations, the composition of phytoplankton also differs, especially in late summer, with more cyanobacteria (*Planktothrix*) occurring in the calculations where measures were not carried out.

The results of the BLOOM stand alone simulations show that for the summer half year an increase in temperature, a change in nutrient concentrations as well as the combination of increased temperature and reduced nutrient concentrations result in different phytoplankton biomass and to a larger extent to changed phytoplankton composition. In the summer half year in almost all scenarios diatom biomass is less than in the reference run. In the period April until June, an increase in temperature leads to comparable cyanobacteria biomass as the reference run (+1.8°C) or lower (+2.6°C) or no cyanobacteria at all (+0.9°C and +1.3°C). Changes in nutrient concentrations lead to higher (plus 10%N, minus 10%P and minus 10%N) or lower (plus 10%P and plus and minus 10%P and N) biomasses in cyanobacteria. In late summer, all scenarios lead to higher cyanobacteria biomass than the reference run. Increased temperatures and reduced nutrient concentrations lead to a phytoplankton composition that cannot be ascribed to either increased temperatures or reduced nutrient concentrations: it is an interplay between those two steering factors. This study also shows synergetic and antagonistic effects of the combined climate change and water management scenarios (see Deliverable 5.2-6).

As for fish, it is therefore important to consider the combination of climate change and water management options, as the effect of climate change can cause a change in the ecological status which can be amplified (positively or negatively) by certain water management measures or the full potential of water management measures is not reached because of interference of climate change.

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Appendices

Appendix 1: Paper: Jeppesen E., P. Nöges, T. A. Davidson, J. Haberman, T. Nöges, K. Blank, T.L. Lauridsen, M. Søndergaard, C. Sayer, R. Laugaste, L.S. Johansson, R. Bjerring & S.L. Amsinck, 2011. Zooplankton as indicators in lakes - a plea for including zooplankton in the ecological quality assessment of lakes according to the European Water Framework Directive (WFD)- *Hydrobiologia* 676:270-297.

Appendix 2: Paper: Davidson T.A., G. H. Henderson, H. Bennion, E. Jeppesen, D. Morley, B. Odgaard, R. Rawcliffe, J. Salgado & C. Sayer, 2011. The role of Cladocerans in tracking long-term change ecosystem structure and function in shallow lakes – *Hydrobiologia* 676:299-315

Appendix 3: Abstract: Jeppesen E., T. Mehner, I. J. Winfield, K. Kangur, J. Sarvala, D. Gerdeaux, M. Rask, H. J. Malmquist, K. Holmgren, P. Volta, S. Romo, R. Eckmann, A. Sandström, S. Blanco, A. Kangur, H. R. Stabo, M. Meerhoff, A.-M. Ventelä, M. Søndergaard, T. L. Lauridsen, submitted. Impacts of climate warming on lake fish assemblages: evidence from 24 European long-term data series