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WISER

Water bodies in Europe: Integrative Systems to assess Ecological status and Recovery

DELIVERABLE

Deliverable D5.2-6: Synthesis paper on options for lake management to improve ecological status – Resistance to climate change in focus

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PP	Restricted to other programme participants (including the Commission Services)	
RE	Restricted to a group specified by the consortium (including the Commission Services)	
CO	Confidential, only for members of the consortium (including the Commission Services)	

Content

Non-technical summary

It is important to understand the mechanisms of climate induced effects on recovery of lakes to identify cost efficient management measures and to set up reasonable reference conditions and ecological classification. These mechanisms were studied by Work package W5.2 using extensive data on ecological quality indicators and key environmental variables.

Based on data and models the following results were drawn:

A. Palaeoecological studies
   - Lake sediments provide a valuable means by which degradation and recovery may be tracked and reference conditions identified.
   - A reduction in one or more environmental stressors may not ultimately return a lake to reference conditions, but instead other processes such as internal nutrient loadings and climate change may determine the rate and direction of recovery
   - A range of sub-fossil organisms may be used to determine both the degradation and the recovery process inlcuding diatoms, cladocerans, chironomids and macrofossils.
   - In terms of lake management, recovery may not simply be the reverse process of the degradation pathway and the reference state may perhaps never be achievable in some lakes.
   - Recovery is more predictable in deep stratified lakes than shallow lakes, where top-down processes exert a major environmental control
   - Multi-proxy palaeoecological techniques have an important role to play in assessing degradation and recovery pathways and informing lake management in order to satisfy the aims of the Water Framework Directive.
   - Palaeoecological techniques are being developed to assess alternative recovery targets using habitat structure and functional process to determine reference conditions rather than species composition.

B. Fish and zooplankton as a BQEs
   - Changes in fish assemblage composition, size and age structure in recent decades are profound.
   - The most obvious alterations encompass a decline in cold-stenothermal species, particularly in shallow lakes, an increase in eurythermal species even in deep, stratified lakes, and several case studies show a decrease in the average size of the dominant species roach and perch. This development 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.
   - Warming will result in fish-mediated increase in eutrophication partly counteracting the effect of nutrient loading reduction. This implies that it will be more difficult to attain the good ecological status required by the Water Framework Directive in lakes facing climate warming, and nutrient inputs to lakes will need to be reduced even further than planned under the present-day climate.
   - The response of the fish to the warming in recent decades has been surprisingly strong, making fish ideal sentinels for detecting and documenting climate-induced modifications of freshwater ecosystems.

C. Effects of temperature and nutrient load to phytoplankton
   - A limited effect of climate warming on eutrophication was noticeable, mainly in central European lakes. The effect may have been masked by other pressures. However, the

study gave an indication that in warmer climatic conditions, a greater reduction of nutrients may be needed to achieve a good ecological condition in a lake.

- The LLR (LakeLoadResponse) internet tool (http://lakestate.vyh.fi/cgi-bin/frontpage.cgi?kieli=ENG) can be used by managers to estimate of the required level of nutrient load reduction of at present and under changed climate conditions.
- LLR produces water quality predictions with statistical confidence intervals to inform management strategies.

D. Impacts of climate change and restoration on ecological status class: a management-oriented modelling approach

- A temperature increase of +2 °C would almost counteract the benefit from 20% P loading reduction, while +4 °C would more than outweigh the benefits from 40% P loading reduction.
- The Bayesian network (BN) modelling approach is generic in nature, and can easily be extended to include more lake types and other biological quality elements, as well as different scenarios.
- The BN approach is particularly useful in relation to environmental risk assessment and management: It can
  1. combine data or other information from different sources
  2. explicitly model uncertainties (as probability distributions)
  3. predict the probability of different outcomes of interest (such as different status classes).

Management options for recovering shallow lake ecosystems under stress: case Lake Veluwe

- Direct measures to improve water quality can have a strong and pronounced effect on primary productivity and ecological status. Climate change (expressed as an increase in average temperature), however is more gradually and can have synergetic as well as antagonistic effect when combined with direct measures taken and therefore need to be considered while planning water management measures.
- Future increase in water temperature due to changing climate, will have a significant impact on both ecosystem processes and species diversity and abundance.
- Well calibrated and validated deterministic models can help provide insight in the effect of direct measures and assess the impact of longer term scenarios for both land use changes and climate change

Recommendations for the management of Lake Pyhäjärvi, Finland.

- Good ecological status in terms of phytoplankton biomass can be achieved most efficiently by reduction of external nutrient load and fisheries management.
- Increased temperature will have an adverse effect which can be compensated with additional load reduction and fisheries management.

E. Guidelines for model usage

- General recommendations for model usage were formulated based on the feedback from stakeholders:
  1. Close co-operation with end users in the formulation of management questions and the conceptual framework.
  2. Designing conceptual models together with stakeholders to have a common understanding and to create a clear picture about the issues that should be addressed
  3. Non scientific reporting of impacts of management measures directly to the end users
  4. More time and resources to collateral data collation and analysis together with end user
  5. Flexible selection of temporal and spatial scales

6. Use of bayesian networks throughout the modeling process in the analysis and dissemination of ecological and social impacts.
7. Securing input and output data into in a generic data base.

- Good modelling practise should be applied carefully

Introduction

Throughout Europe the majority of lakes have been modified to some extent by human activity with agriculture and sewerage being the major contributors to eutrophication, most notably since the mid-twentieth century. As a consequence, higher algal productivity has lead to filtration problems for the water industry, oxygen depletion, recreational impairment, loss of biodiversity and an overall decline in habitat quality.

The Water Framework Directive (WFD), adopted in 2000 (European Commission, 2000) has, at its centre, the requirement that all waters (rivers, lakes, groundwater, coastal and transitional waters) should achieve 'good ecological status' by 2015 and that there should be no future deterioration from this state. Good ecological status is defined as 'slight' deviation from 'high' status which represents the biological, chemical and morphological conditions associated with no, or very minor, alteration as a result of human activities. This represents the baseline or 'reference condition. Ecological status is determined using a series of biological quality elements BQEs), supported by hydromorphological and physico-chemical quality elements. Although the Directive specifies the BQEs required for assessment (macroinvertebrates, fish and macrophytes phytoplankton) and the characteristics that should be assessed (e.g. composition and abundance of aquatic flora or composition, abundance and age structure of fish) it does not indicate which indices or metrics should be used for the different BQEs and this has been left to individual member states.

To assess ecological status and to determine whether restoration efforts have been successful knowledge of the baseline conditions that occurred prior to the human-induced stress is required. With 'good ecological status' defined as only slightly different from the high ecological status of the reference conditions, the way the reference state is defined will be of critical importance. There are few, if any systems, remaining in Europe that are unaltered or only negligibly altered by human activity. There are numerous techniques available to establish reference conditions including temporal (the use of historical records or, for lakes, palaeoecological reconstructions) spatial (using survey data and space-for-time substitution) and modelling approaches (hindcasting) as well as expert judgment.

However, the WFD is not currently explicitly formulated to allow for the potential effects of climate change. Assessment metrics are generally focused on 'traditional stressors' such as morphological change, acidification, eutrophication) and there are currently no specific metrics for the effects of emerging stressors, in particular changing climate. In fact climate change is not deemed as an anthropogenic pressure under the WFD due to the perception that the impacts cannot be tackled by water managers (Quevauviller, 2011). However, it is now recognised that climate change may have significant consequences for the implementation of the WFD at all stages (Wilby et al., 2006). Given the Directive embraces the principle of protecting ecosystems from deterioration, there is a pressing need for it to be amended to accommodate the likely effects of climate change on freshwater ecosystems. Climate change will inevitably undermine ecological assessment schemes as species become eliminated or migrate into previous cooler habitats. This does not invalidate the reference condition concept. A reference condition is not necessarily a static state and can change over time, within the constraints of its ecological envelope (i.e. physical conditions determined by location and catchment characteristics, the biological potential of the water body and the interactions between populations) (Battarbee et al., 2008). Reference conditions are not only dynamic but subject to directional change where the baseline is continually changing (Battarbee et al., 2005 - Figure 1)

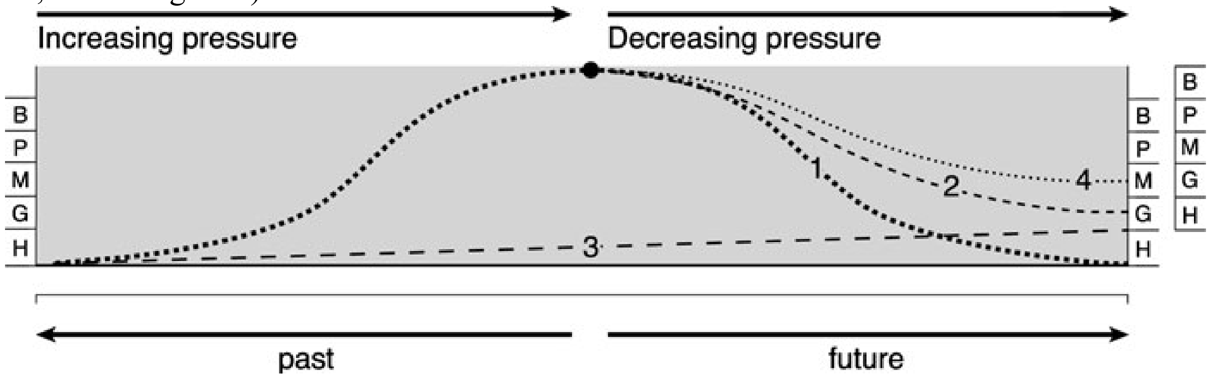


Figure 1: Conceptual diagram to show the potential impact of climate change on reference conditions and its implication for the Water Framework Directive: 1. Ideal return trajectory to the palaeo-defined reference; 2. Response trajectory to measures designed to achieve 'good' ecological status by a specified time; 3. Shift in reference baseline as a result of climate change; 4. Altered trajectory caused by the reduced effectiveness of the measures taken as a result of climate change. In this example climate change prevents restoration to 'good' status. Options are either to intensify the measures taken or to shift the recovery target and re-classify the 'good/ moderate' boundary as indicated by the right hand panel (from Bennion et al., 2011).

Here, lake sediment analysis provides unique insights into the history of lake ecosystems, including evidence for the nature and timing of ecosystem change resulting from human impact. Palaeoecological methods can reveal pre-impact conditions as well as identifying any signs of recovery and have played a key role in the WFD in determining pre-enrichment reference conditions. Diatom records have proved especially valuable in this respect, largely due to their sensitivity to shifts in trophic status. In the absence of long-term chemical monitoring, analysis of lake sediments can provide evidence not only of the pre-eutrophication baseline conditions, but also help track degradation and recovery pathways and thus provide a valuable tool for informing restoration programmes. Furthermore, where restoration programmes are underway, there is evidence that the recovery pathways are not simply a reverse of the degradation process and therefore indicate that other factors such as climate change may be influencing the rate and direction of recovery.

Many lake ecologists were surprised that zooplankton were not included as a biological quality element (BQE) in lake assessment according to the EU Water Framework Directive (WFD), given that they are considered to be an important and integrated component of the pelagic food web. Fish play also a key role in the trophic dynamics of lakes. With climate warming, complex changes in fish assemblage structure may be expected as a direct result of temperature increases and indirectly through eutrophication, water level changes, stratification and salinisation. The response of fish to the warming in recent decades has been surprisingly strong, making fish ideal sentinels for detecting and documenting climate-induced modifications of freshwater ecosystems.

Fish play also a key role in the trophic dynamics of lakes. With climate warming, complex changes in fish assemblage structure may be expected owing to the direct effects of temperature and indirect effects of eutrophication, water level changes, stratification and salinisation. The response of fish to the warming in recent decades has been surprisingly strong, making fish ideal sentinels for detecting and documenting climate-induced modifications of freshwater ecosystems.

Mathematical models are also needed to estimate individual and combined effects of climate change and management. However, the use of models in lake and river basin management is often cumbersome and confusing due to insufficient monitoring data, the opacity of complex models and the uncertainty of predictions. Thus, clear guidance and criteria for model selection and use in lake management is needed.
For example, how will the assessment of ecological status and the outcome of restoration efforts be affected by climate change? How can uncertainties in assessment be quantified? These are the two of key questions, which have been addressed in many of the project's deliverables and publications. However, the scientific output may not directly address the questions being asked by river basin managers, such as "how will +2° temperature increase affect the probability of failing to achieve good ecological status?" For this purpose we have applied a probabilistic modelling approach - Bayesian networks - to summarise information on relationships between climate change, restoration, typology, pressure, biological response and ecological status class.

In the present study we reviewed paleaoecological data as well as analysed phytoplankton, zooplankton and fish data from a number of lakes to obtain basic understanding of the response of lake ecosystems to changes in nutrient in a world with changing climate. In addition, based on a simple retention model, a linear mixed effects chlorophyll a model and on data from 351 European lakes we developed a simple loading response model. The resulting LLR (LakeLoadResponse) internet tool (http://lakestate.vyh.fi/cgi-bin/frontpage.cgi?kieli=ENG) was used to study the effect of total phosphorus, total nitrogen and water temperature on chlorophyll a concentrations in WFD affiliated lake types and lakes.

We also present data from two intensively studied case studies i.e. Lake Veluwe, Netherlands and Lake Pyhäjärvi, Finland. Both lakes are shallow and eutrophic clear-water lakes with moderate to high alkalinity situated in Northern and Central Europe, respectively. Data and models were used to assess the impacts of climate change and catchment management measures on ecological status. Based on these modeling efforts and earlier experience, the guidelines on the use of different modeling approaches for the design of program of measures was developed.

Using aforementioned methods and physical, chemical, biological quality indicators from several European lakes and their sediments were analyzed (Figure 2).

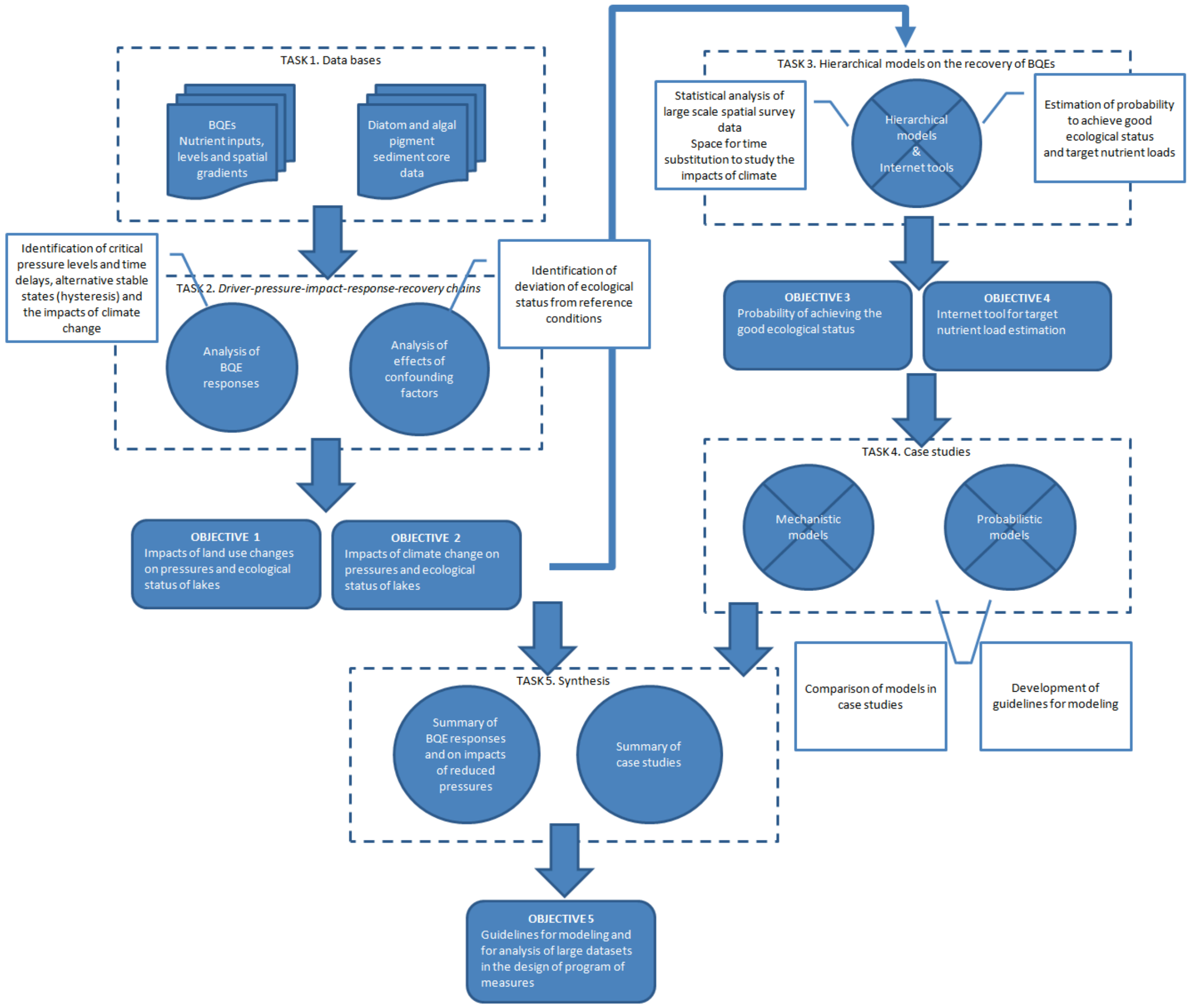


Figure 2: Workflow of WP5.2

Objectives

The goal of this deliverable was to summarize the options for lake management to improve ecological status with specific focus on climate change effects. Lessons learnt were discussed and guidelines for model usage in river basin management were outlined.

The data analysis and model applications addressed (Figure 3):

1. *Land use*: Assess the impacts of catchment management strategies on BQEs and ecological status of lakes. The pressures addressed include eutrophication and hydro-morphological alterations (mainly lake level regulation).
2. *Climate change*: Assess the impacts of climate change on ecological status of lakes, focusing on impacts on the thresholds used to set the good/moderate class boundary for the various BQEs.
3. *Uncertainties*: Assess the uncertainty and risks of failing to achieve and maintain the good status objective under various climate change scenarios.
4. *Target nutrient loads*: Develop and apply models and tools that can be used for estimating the required pressure levels to achieve good ecological status.
5. *Guidelines*: Develop guidelines for using case-specific mechanistic models/empirical analysis of large datasets for designing the program of measures under climate change.

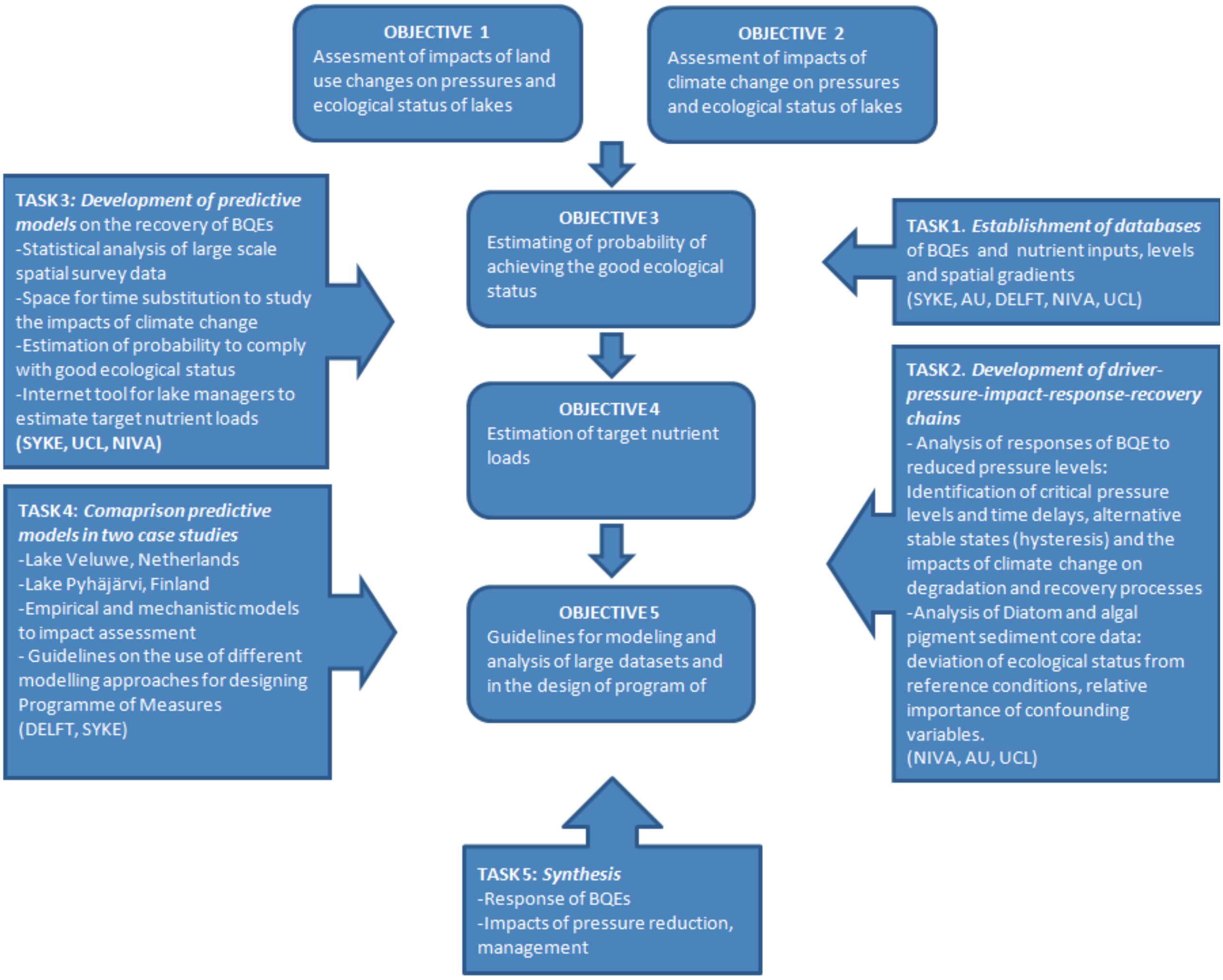


Figure 3: Objectives of WP 5.2.

Paleoecological studies

Lakes in Europe have been widely affected by a range of anthropogenic stresses including eutrophication, acidification, pollution by toxic substances, changes in water level and introductions invasive species. The scale of these impacts ranges from low-level contamination by long-distance transported air pollutants in remote regions to severe degradation from catchment sources in agricultural and urban areas. Efforts to manage or restore lake ecosystems to pre-impact conditions have been given impetus by the EU Water Framework Directive (WFD) (European Union 2000). The WFD requires that the ecological quality of surface water bodies should not be significantly different from those found prior to anthropogenic impacts, i.e. the reference condition. The reference condition concept is thus at the heart of the WFD and there are number of approaches that can be used to establish this including temporally based approaches using historical data, the use of survey data for time-space substitution, modelling approaches (such as hindcasting) and expert judgement.

Palaeoecological applications in lake management

For lakes, palaeolimnology offers an additional approach, using the sub-fossil record from sediment cores to highlight species assemblages and infer environmental conditions prior to impact and during both degradation and post-restoration recovery phases (e.g. Birks 1996; Battarbee 1999; Smol 2008). Applications have included the use of radiometric dating to determine sediment accumulation rate changes through time (Rose et al. 2011) deriving references for nutrient status (Bennion and Simpson 2011), acidity (Battarbee et al. 2011), organic carbon (Cunningham et al. 2011) and trace metals (Bindler et al. 2011). Further refinement and development of palaeoecological techniques have enabled the approach to be used specifically to support management within the context of the WFD (Bennion and Battarbee 2007). The palaeolimnological record has been used to define chemical (Bennion et al., 2004; Battarbee et al., 2005) as well as ecological reference conditions and ecological response to environmental pressures (Sayer et al., 1999; Davidson et al., 2005). Palaeolimnology can be used to define the 'reference point' for systems impacted over 1-2 hundred years (e.g. Bennion et al., 2004) or for over centennial to millennial timescales (e.g. Bradshaw et al., 2006). Ecological classification based on deviation from baseline conditions can be supported by palaeolimnology (e.g. Smol et al., 2008) and the sub-fossil record can also supplement monitoring under the WFD, identifying where ecological thresholds have been crossed (Battarbee. 1999).

Several areas for future improvement have been proposed including; i) the wider use of multi-proxy approaches to assess reference conditions and status, ii) amalgamating results from different organsim groups (such as diatoms, cladocerans, chironomids and macrofossils to provide an holistic assessment of change over time, iii) development of floristic and faunistic indicators for setting ecological status boundaries, iv) further development of analogue matching techniques to identify reference sites for a broader range of lake types and pressures,) incorporating surface-sediment sampling into WFD monitoring programmes and vi) integrating palaeoecology with modelling time-series analysis. Some of the difficulties associated with the role of lake sediments in establishing reference conditions and defining recovery targets have been addressed by Bennion et al., 2010b) and include i) distinguishing signals representing human activity from those resulting from natural causes or random variability, ii) quantifying the degree and rates of change and iii) accounting for state changes that may have taken place between pre-impact and the present that may limit the use of the reference condition as a restoration target.

Climate Change impacts on lakes and lake management

It is clear that climate is changing rapidly, beyond the range of previous natural variability. Lake ecosystems, already under stress from land-use change and pollution, now face additional pressures from climate change, both directly and through interaction with other drivers of change. Evidence from long-term data-sets shows that many of the effects of changing climate are already occurring including increases in surface water temperature, particularly at high altitudes and latitudes, strengthening and lengthening of lake stratification in summer, increases in hypolimnetic temperature of large deep lakes and reductions in lake ice-cover. These are likely to have significant ecological consequences through modifying species life-histories, impacts on species abundance, composition and distribution, alterations to food-web structure, and changes in functional aspects such as the rates of species and community metabolism.

Palaeoecological evidence has been used to make inferences about the response of lake ecosystems to climate change over different time-scales, from multi-millennial to seasonal focusing (Battarbee, 2010). Impacts studies include those driven by temperature changes and those resulting from changes in effective moisture. Palaeolimnological methods can be used to assess decadal and centennial variability in the past which, when projected forward, allows the identification of climate-related risks of failing comply with WFD objectives.

However, it is now recognised that the increasingly apparent effects of climate change (and other global pressures such as nitrogen deposition) may mean that recovery targets may need to be modified (Battarbee et al. 2005; Wilby et al. 2006). It follows that the WFD will need to be refined to accommodate such change and the programmes of measures identified for achieving good ecological status (and permitting no further deterioration) will need to be climate proofed. A key consideration for the WFD is how to ensure compliance with the mandatory standards set by the WFD in situations where climate change exacerbates water quality problems, especially those of eutrophication and toxic substance pollution. The effects of climate change on the definition of reference conditions, especially with respect to the use of the reference state as a restoration target is also a concern as climate change will affect the status of sites used as references. Climate change is also likely to affects the ecological thresholds currently used to set the WFD target (good/moderate status class boundary).

Palaeolimnology as a management tool under climate change

A major challenge with using palaeolimnology as a tool to incorporate climate change impacts into assessments of reference conditions and recover targets is that it can be very difficult to disentangle the individual signals when multiple stresses are acting on a system. Climate change can act in tandem with other pressures and exacerbate the impacts. One of the consequences of climate warming is likely to be increased eutrophication (Jeppesen et al. 2010) which will act in a very similar way to nutrient pollution. Reduced precipitation may cause increases in lake water salinity which will be manifested in the sediment record in the same way abstraction for irrigation (cf. Austin et al. 2007). However, recent developments with numerical quantitative techniques now allow the independent effects of different pressures to be identified data using sophisticated statistical methods (Simpson and Anderson 2009).

Where multiple stresses are acting the use of additional proxies from the sediment core can support inference and be used to identify the dominant driver of change. Stable isotopes of nitrogen and lead concentrations have been used in tandem with the diatom record to show that ecological change at some sites in the Rocky Mountains is being driven by nitrogen deposition rather than climate change.(Wolfe et al., 2001) . Similarly analysis of pollen and heavy metals from sediments in Scotland showed that the acidification of the site resulted from atmospheric deposition of acidifying compounds rather than land-use change (Battarbee et al.1985)

Palaeolimnology is particularly valuable in identifying ecological changes over time and this can be used to highlight the notional reference condition. What is becoming increasingly clear is that the reference condition is not static but varies naturally in the short term (at interannual to decadal) longer term perturbations at centennial to millennial timescales. So, although palaeolimnology can be used to identify a reference condition relative to the onset of some anthropogenic impact, whether that baseline is the most appropriate reference for lake restoration purposes depends on the extent of environmental degradation the impacts of confounding factors such as pressures from climate change which may shift the potential recovery target.

Developments with palaeolimnological methods mean this approach can now be used to assess alternative recovery targets. The comparison of sub-fossil assemblages with contemporary species composition is very site specific and, in management terms, the approach implies the requirement to return to the same species assemblage as the reference. Even without the influence of climate change, this is a very ambitious target. An alternative to the site specific approach is to use reference assemblages from more than one site, similar in type to the lake of interest. This was done for acidified lakes by examining diatom assemblages from a series of low alkalinity systems to identify species to be used as indicators for reference conditions at such sites (Battarbee et al., 2010). However, even within the low alkalinity typology there is considerable variation in reference assemblages, naturally acidified systems dominated by benthic taxa to circumneutral sites where planktonic diatoms were more abundant. By classifying sites into clusters it was possible to derive reference assemblage characteristics of a type of lake rather than a specific site, which matches more closely the use of metrics in the WFD.

Another approach is to use habitat structure and functional process to determine reference conditions rather than species composition. This can be applied in a palaeolimnological approach by the use of contemporary ecological training sets representing functional attributes and habitat characteristics of different organisms groups (Bennion et al. 2011a; Sayer et al. 2011a). This relatively new approach has been employed to reconstruct zooplanktiverous fish densities from sub-fossil cladocera (Jeppesen et al. 1996, 2001a) (Jeppesen et al. 2001b; Bjerring et al. 2008) and the characteristics of submerged macrophytes from plant macrofossil and pollen remains (Davidson et al. 2005; Ayres et al. 2008; Salgado et al. 2011; Sayer et al. 2011b). This approach aims to describe the physical habitats within sites during the reference period rather than specific species assemblages. The approach is demonstrated by multi-proxy palaeolimnological studies of eutrophication at Felbrigg Hall Lake, England (see Sayer et al. 2010b; Davidson et al. 2010) whereby the recovery of the lake was not measured by a return to the species assemblage prior to impact but the restoration of habitat attributes and lake functions. Further work is required to determine whether this functional based approach can be defined for all lake types. The approach can be enhanced using analogue matching techniques (e.g. Flower et al. 1997; Simpson et al. 2005; Bennion et al. 2011b; Rawcliffe et al.2011) where the reference assemblages from impacted lakes can be used to identify population of unimpacted sites that can act as reference sites for the damaged system, combining information from palaeoecological and spatial sources.

Ultimately the value of a specific reference point depends on whether it can be achieved by removing or reducing the relevant pressures or managing the response. Where existing pressures or the impacts of climate change have led to ecological thresholds being crossed and state changes occurring, restoring lakes so that pre-impact species assemblages are returned may not be feasible. In these circumstances restoring past habitats and ecological processes and functions may be more feasible. Thus for palaeolimnology the reference condition equates to a benchmark rather than the recovery target. The priority therefore is to continue developing the methods so that past ecosystem functioning as well as ecosystem structure can be reconstructed (Bennion et al., 2011c).

Fish respond fast to climate warming and counteract recovery from eutrophication

An analysis of the effect on fish assemblages to climate change and climate variability have been conducted based on long-term (10 to 100 years) data series from 24 European lakes (Jeppesen et al, submitted). European lakes constitute an appropriate and tractable sample of the world’s lakes since many of them have been monitored more intensively and for a longer period of time than have most lakes elsewhere. Profound changes in fish assemblage composition, size and age structure were found during the last decades and a shift towards higher dominance of eurythermal species. The cold-stenothermic Arctic charr has been particularly affected and its abundance has decreased in the majority of the lakes where its presence was recorded. The harvest of cool-stenothermal trout has decreased substantially in two southern lakes (Lake Geneva and Lake Maggiore).Vendace, whitefish and smelt has shown a different response depending on lake depth and latitude, with a drastic reduction in the Estonian Lake Peipsi.

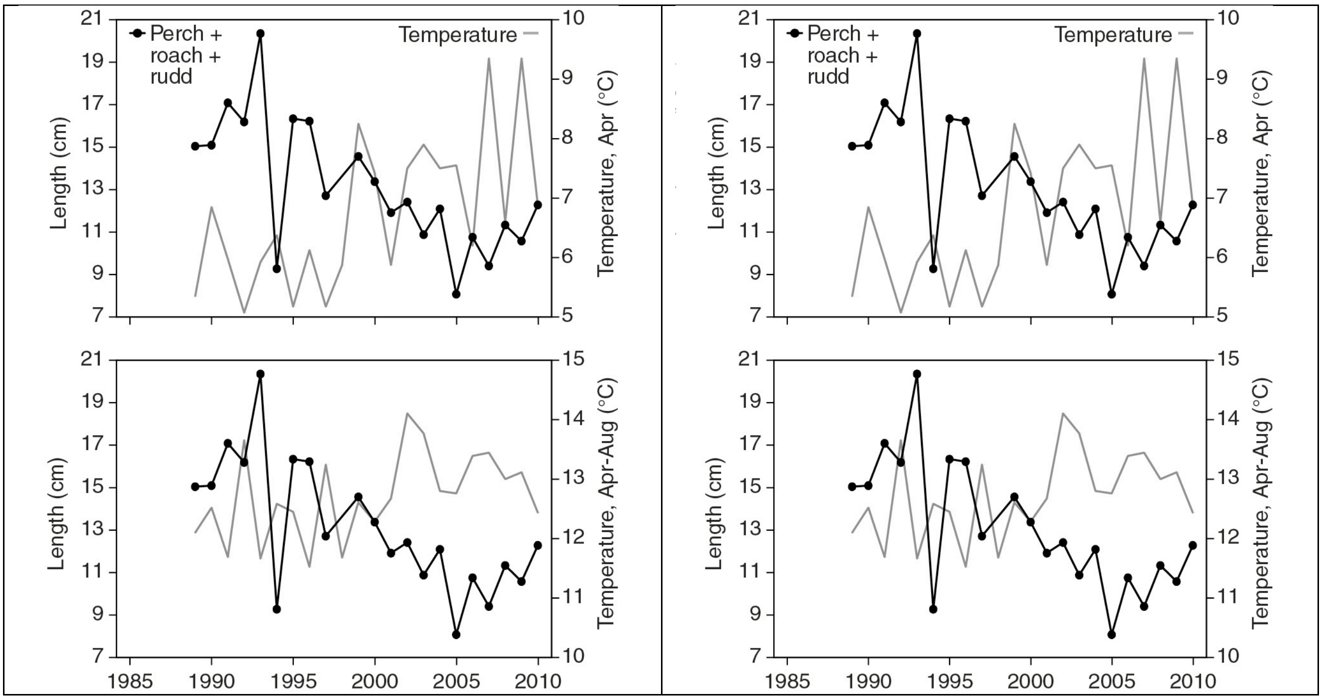


Figure 4: Lake Søbygård. A: Summer mean total phosphorus and chlorophyll. Left: CPUE by weight of various key fish species. Right: Mean per capita biomass of roach, rudd and perch (pooled together) and average air temperature in April and summer (Apr. 1 to Sept. 1). Since 1989, total CPUE in terms of biomass has shown a declining trend in Lake Søbygaard, coinciding with a decrease in nutrient concentrations. A major change has occurred from roach dominance to dominance by a mixed assemblage of roach and perch. This is to be expected when lakes recover from eutrophication. A major decline has occurred in average size of both roach and perch during the study period, leading to a significant reduction in the average size of the roach, perch and rudd pooled together). This decline coincides well with the change in April air temperatures), and even better with the average summer air temperature (April-September), indicating that the changes might reflect an increase in ambient temperatures. In a multiple regression including also phytoplankton chlorophyll a and TP, only temperature was retained in the final model, further emphasising the key role of temperature for the size change. (Jeppesen et al, submitted)

Perch was apparently stimulated in the north, with stronger year classes in warm years, but its abundance has declined in southern Lake Maggiore, as judged from the harvest. 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 are on the increase in several of the lakes. The climate effects have overall been larger in shallow lakes. Moreover fish size has declined in several of the lakes (see example in Fig. 4), following the pattern of declining fish size observed along a north-south a gradient in Europe (Emmerich et al, submitted; Brucet et al, submitted, Jeppesen et al, 2010b).

Changes in lake fish communities may enhance the risk of turbid conditions and dominance of cyanobacteria (Jeppesen et al. 2010a,2012). 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. 5). This usually suggests enhanced predation by fish. A tendency to a decrease in the zooplankton:phytoplankton 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 that also are stimulated by warming (Fig 5). 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.

Moreover, a reduced ice cover in winter will enhance fish survival, with potential cascading effects within the food web (Balayla et al, 2010, Ruuhijärvi, et al. , 2010), also reinforcing eutrophication. Thus Monitoring data from Danish lakes show clear 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 (Balayla et al. 2010). The size structure of the main microcrustaceans was displaced towards larger size classes in the summer of 1996, resulting in a significantly greater grazing capacity on phytoplankton. Accordingly, phytoplankton biomass (as chlorophyll *a*) was lower and grazing apparently higher. All these effects were stronger in shallow than in deep lakes (Balayla et al. 2010). In a Finnish lake, Ruuhijärvi et al. (2010) recorded similar results of winter fish kills. In combination, the results from the long-tem study of Lake Søbygård and less frequent samplings from numerous Danish lakes strongly indicate that despite a reduction in external nutrient loading and a subsequent reduction in the overall biomass of fish, fish density is increasing and the average body size decreasing, with potentially strong cascading effects. The results from Danish lakes concur with a latitude gradient study by Gyllström et al. (2005) of shallow European lakes that showed that the ratio of fish biomass (expressed as catches per net in multi-mesh sized gillnets) to zooplankton biomass increased from north to south and that the zooplankton:phytoplankton biomass ratio decreased, both substantially. Moreover, higher-latitude fish species are typically larger, grow more slowly, mature later, have longer life spans and allocate more energy to reproduction than populations at lower latitudes (Blanck and Lammouroux 2007). Even within species such changes are seen along a latitude gradient (Blanck and Lammouroux 2007; Jeppesen et al. 2010b).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.

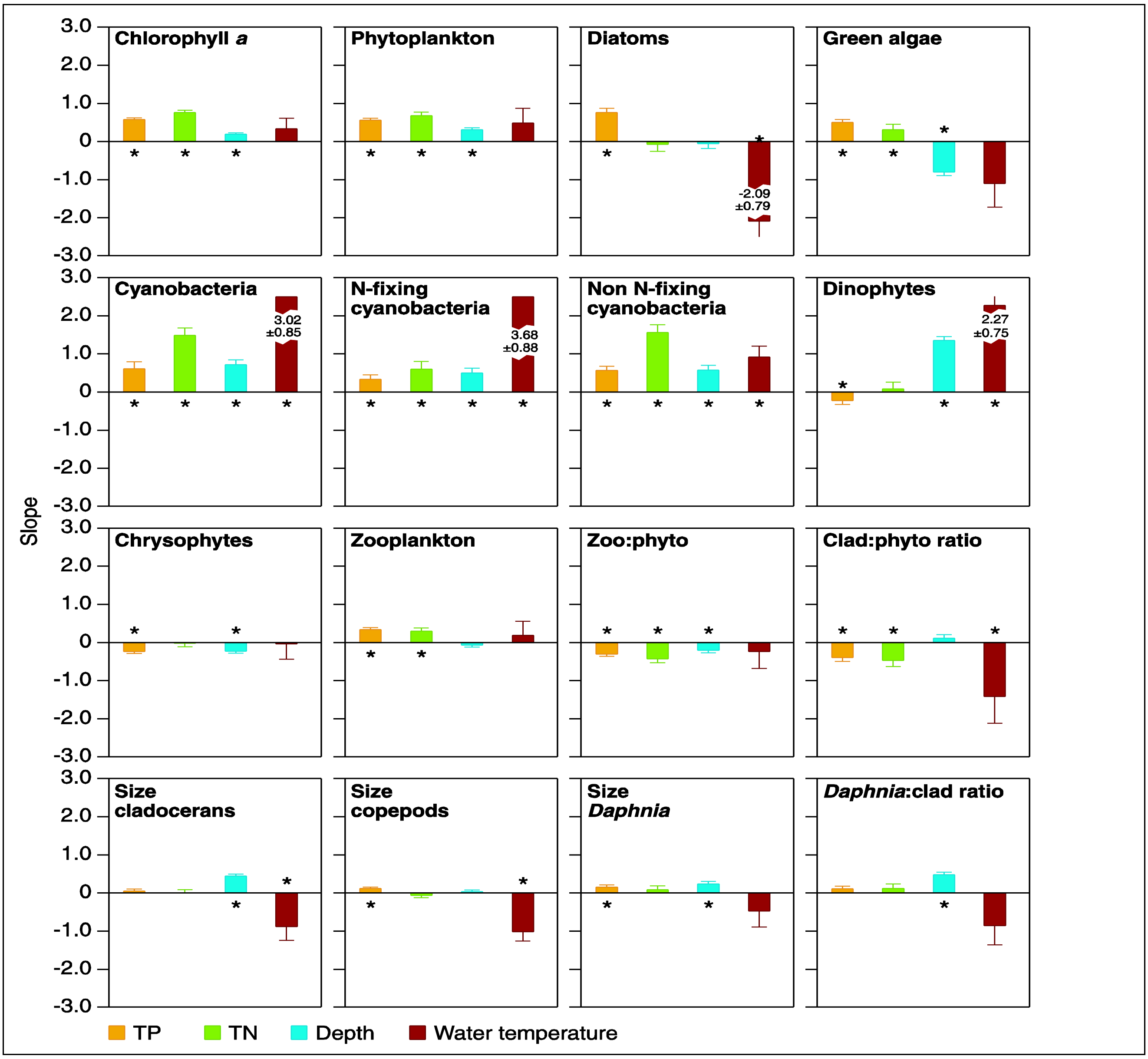


Figure 5: Multiple regressions between different plankton variables (log$_e$-transformed) and total phosphorus, total nitrogen and water temperature of the surface layer and lake mean depth – all log$_e$-transformed. When the value is positive, there is a positive effect of a given variable (when including also the other variables) and the opposite, if negative. All data are from August. Significant values are marked with an From Jeppesen et al. (2012).

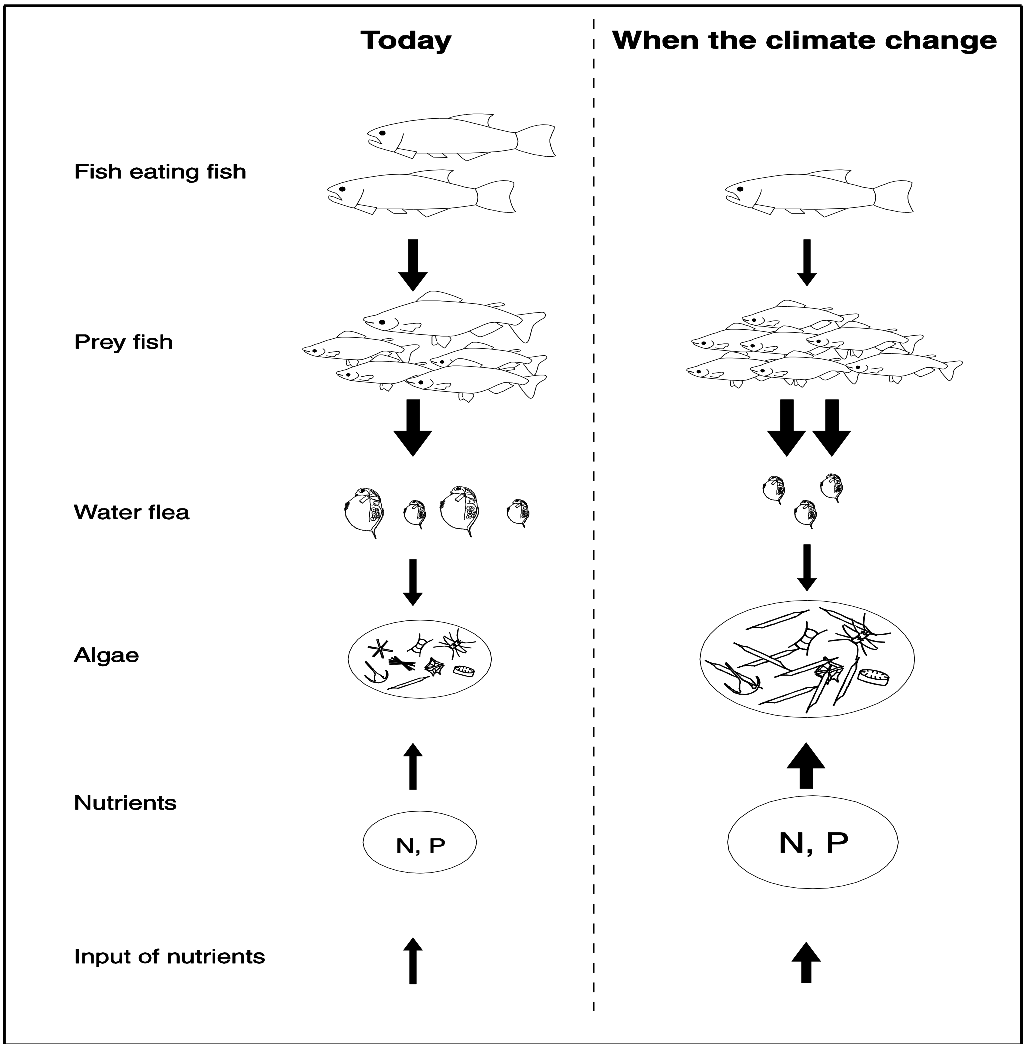


Figure 6: Conceptual model showing trophic structure in meso-eutrophic Danish lakes and the suggested changes in a climate change perspective. Today top-down control by piscivorous fish are medium to low depending on nutrient level and cyprinids feeding on large-bodied zooplankton, conversely, medium to high. Accordingly, there are moderate to few Daphnia and moderate to low grazing on phytoplankton. Such lakes are sensitive to additional nutrient loading but somewhat buffered by grazing by zooplankton, if the nutrient level is not high. When the lakes become warmer they will to a higher degree be dominated by small fish, and zooplankton will be less abundant and smaller. Accordingly, the lakes will be more sensitive to addition of nutrients. Moreover, cyanobacteria will be stimulated by both higher temperature and higher external and internal nutrient loading (from Jeppesen et al, 2012).

In summary, we have found profound changes in fish assemblage composition, size and age structure during the last decades. The most obvious alterations encompass a decline in cold-stenothermal species, in particular in shallow lakes, an increase in eurythermal species even in deep, stratified lakes, and several case studies show a decrease in the average size of the dominant species roach and perch. This development 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. Such chnages have cascading effect down the food-web and lead to fish-mediated increase in eutrophication partly counteracting the effect of nutrient loading reduction (Fig. 6).

This implies that it will be more difficult to obtain the good ecological status required by the Water Framework Directive 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. The response of the fish to the warming during recent decades has been surprisingly strong, making fish ideal sentinels for detecting and documenting climate-induced modifications of freshwater ecosystems.

Zooplankton a good indicator of eutrophication and climate change - should be included as a BQE.

Surprisingly to many lake ecologists, zooplankton were not included as a biological quality element (BQE) in lake assessment according to the EU Water Framework Directive (WFD), the despite 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 UK lakes and time series following changes in pressures (eutrophication and top-down control e.g. as a result of climate change or biomanipulation) it was shown that contemporary zooplankton as well cladoceran remains in the sediment have a strong indicator values and when selecting the right metrics zooplankton are cost-efficient indicators of the trophic state and ecological quality of lakes (Jeppesen et al.,2011) and via the indirect effect on fish community structure described above also of changes in climate (Jeppesen et al, 2011, 2012). Moreover, they can be important indicators of the success/failure of measures taken to bring the lakes to at least good ecological status. We found that zooplankton size structure, proportion of large zooplankton, cladoceran size and the Zooplankton:Phytoplankton biomass ratio are suitable indicators of ”top-down” processes in lakes. Important indicators of ”bottom-up” processes could be zooplankton biomass, the proportion of rotifers by numbers and the proportion of calanoid copepods 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. 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 also 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.

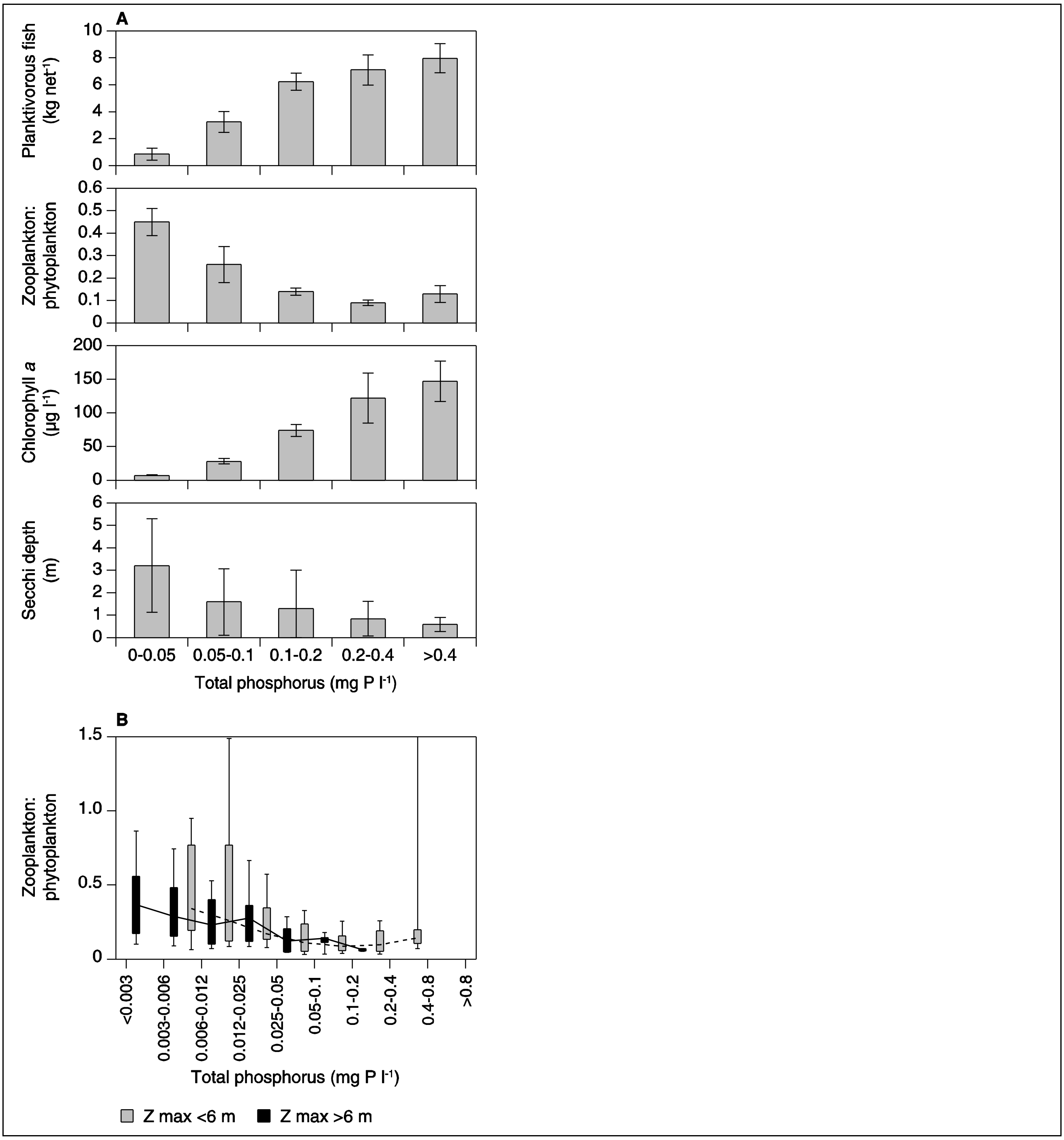


Figure 7: A: August biomass of zooplanktivorous fish (CPUE, catch in multiple mesh-size gill nets, 14 different mesh sizes 6.25-75 mm in late summer) versus summer mean lake water concentrations of TP. Also shown are summer mean (1 May - 1 Oct) of the zooplankton:phytoplankton biomass ratio, the epilimnion chlorophyll a concentration and Secchi depth versus TP. Mean±SD of the five TP groups is shown. B: Box-plots showing the biomass ratio of zooplankton to phytoplankton (means for July - August) in 466 deep (more than 6 m) and shallow (less than 6 m) Danish and Norwegian lakes. (From Jeppesen et al., 2011).

Effects of temperature and nutrient load to phytoplankton

The eutrophication of European lakes was studied using a linear mixed effects chlorophyll a model which was fitted to 461 European lakes. The effect of total phosphorus, total nitrogen and water temperature on chlorophyll a concentrations varied within 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 chlorophyll a samples were taken. By using the linear mixed effects model for nested data we can substantially decrease the variation of data by selecting both the fixed effects and variance structure properly. Based on the data analysis of the European data set, the effect of climate warming on eutrophication proved to be positive. Thus, in warmer climatic conditions, a bigger reduction of nutrients is needed to achieve good ecological condition in a lake. For predicting 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 data set that we used to model the chlorophyll a (Chla) responses to total phosphorus (totP), total nitrogen (totN) and water surface temperature (temp) has total of 2265 observations, from 12 countries, from 461 lakes and from 21 WFD affiliated lake types. In order to get as comprehensive picture as possible from the water temperature effects to chlorophyll a, we examined the data from two different temporal angles. The whole data set has observations from all year round and the late summer data have values only from July and August (referred as "Jul-Aug"). The July-August data set has 1411 observations from 426 lakes.

The whole data set characterizes more of the dynamic variation of species and phytoplankton production annually, whereas the July-August situation is about the summer primary production maximum and variation of phytoplankton abundance there. Obviously water temperature does have an effect to phytoplankton abundance. The effect in late summer is more straightforward, as for the whole data set the temperature effect is mixed with inter annual dynamic variation. In addition the variation in the temperature in summer time is smaller and the possibility to get the effect of changing temperature is thus harder.

Histograms, scatter plots and correlations between the variables are shown in Figure 8 a). The correlations between log-scaled chlorophyll a and the predictors are relatively strong. The correlation between surface water temperature and log(Chla) is lower (0.23) but it can roughly be assumed that when the water gets warmer the concentration of chlorophyll a grows somewhat linearly. The substantial variation in chlorophyll a concentrations between the lake types can clearly be seen in Figure 8 b) that shows box plots of chlorophyll a conditional on lake type. The lowest medians are in Northern GIG type LN7 and the highest is in Central European and Baltic GIG types (LEC, LCB). There are also some differences of chlorophyll a concentrations between the data sets. E.g. for Central Europe and Baltic countries' types (LCB) the median concentrations and variations are somewhat different in late summer and in the whole data set.

a.) *b.)*

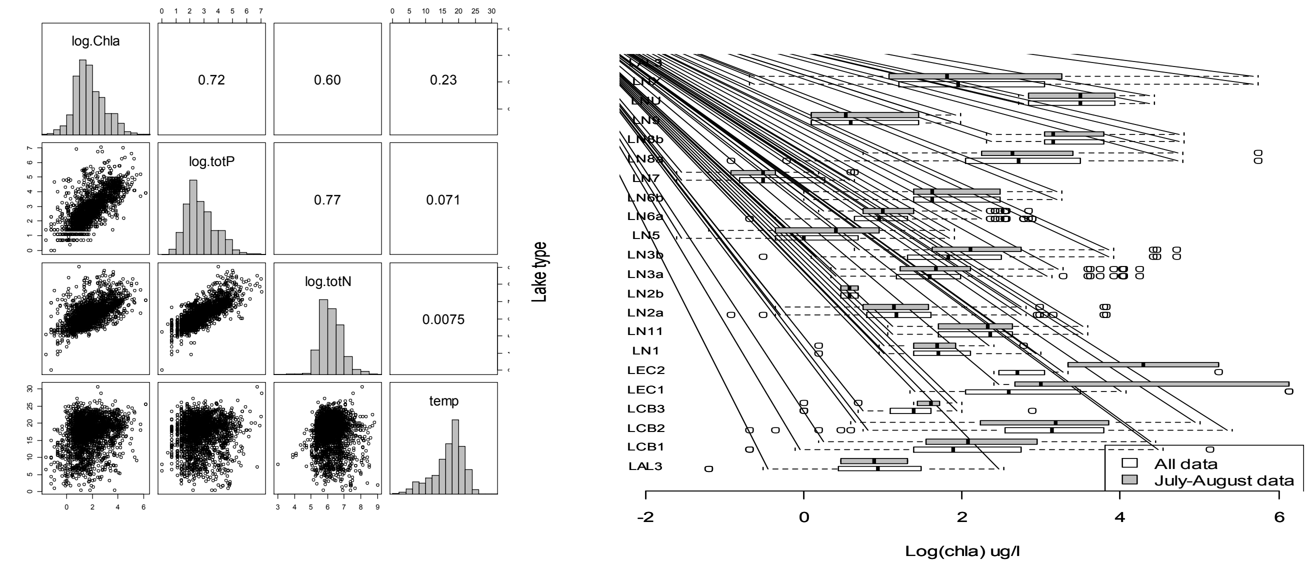


Figure 8: Figure 1 a.) Histograms, scatter plots and correlations between the variables for the whole data se and b.) box plots of of log(Chla) [µg/l] concentrations in each lake type for the whole data set (white boxes) and for July-August data (grey boxes).

Temperature effects to chlorophyll a concentrations for different lake types can be seen in Figure 9. For July-August it seems that for LEC types the effect is very strong. But as there are only few data points we can not say anything about this relationship. Nevertheless there seem to be some kind of linear relationship between chlorophyll a and water temperature for some lake types. And the relationships seem to be stronger fort he whole data set than fort he late summer data. From the scatter plots and linear regression lines drawn for chlorophyll a and the total nutrients it is clear that phosphorus affects the chlorophyll most, and this is the case for most of the lake types in both data sets. Same stands for totN (these figures are not shown here).

As we could see the chlorophyll a response to nutrients and water temperature can be assumed to be linear. Although using linear regression models requires several assumptions concerning the normality and homoscedasticity of the variables, independence of observations and deterministic nature of variables. There were differences in the responses of the nutrients and temperature to chlorophyll a in several lake types. This heteroscedasticity and different responses in the lake types can be dealt with by using (hierarchical) mixed effects models. Mixed effects models allow different lake types to have different variation. Also the observations from the same lake within the lake type may correlate and this violates the assumptions of the traditional regression analysis. With mixed effects models this correlation can be taken considered.

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:

$$\mathrm{chla}_{ijk} = \underbrace{\mathrm{totP}_{ijk} + \mathrm{totN}_{ijk} + \mathrm{temp}_{ijk}}_{\textit{fixedeffects}} + \underbrace{\mathrm{u}_{k} + \mathrm{u1}_{j|k} + \mathrm{u2}_{j|k} + \mathrm{u3}_{j|k}}_{\textit{randomeffectsoftypes}} + \underbrace{\mathrm{v}_{j|k}}_{\textit{Rrandomeffectsoflakes}} + \underbrace{\varepsilon_{ijk}}_{\textit{errorterm}}$$

$chla_{ijk}$ 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

$u1_{j|k}, u2_{j|k}, u3_{j|k}$ is the type specific random slopes for totP, totN and temp

$v_{j|k},$ 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.

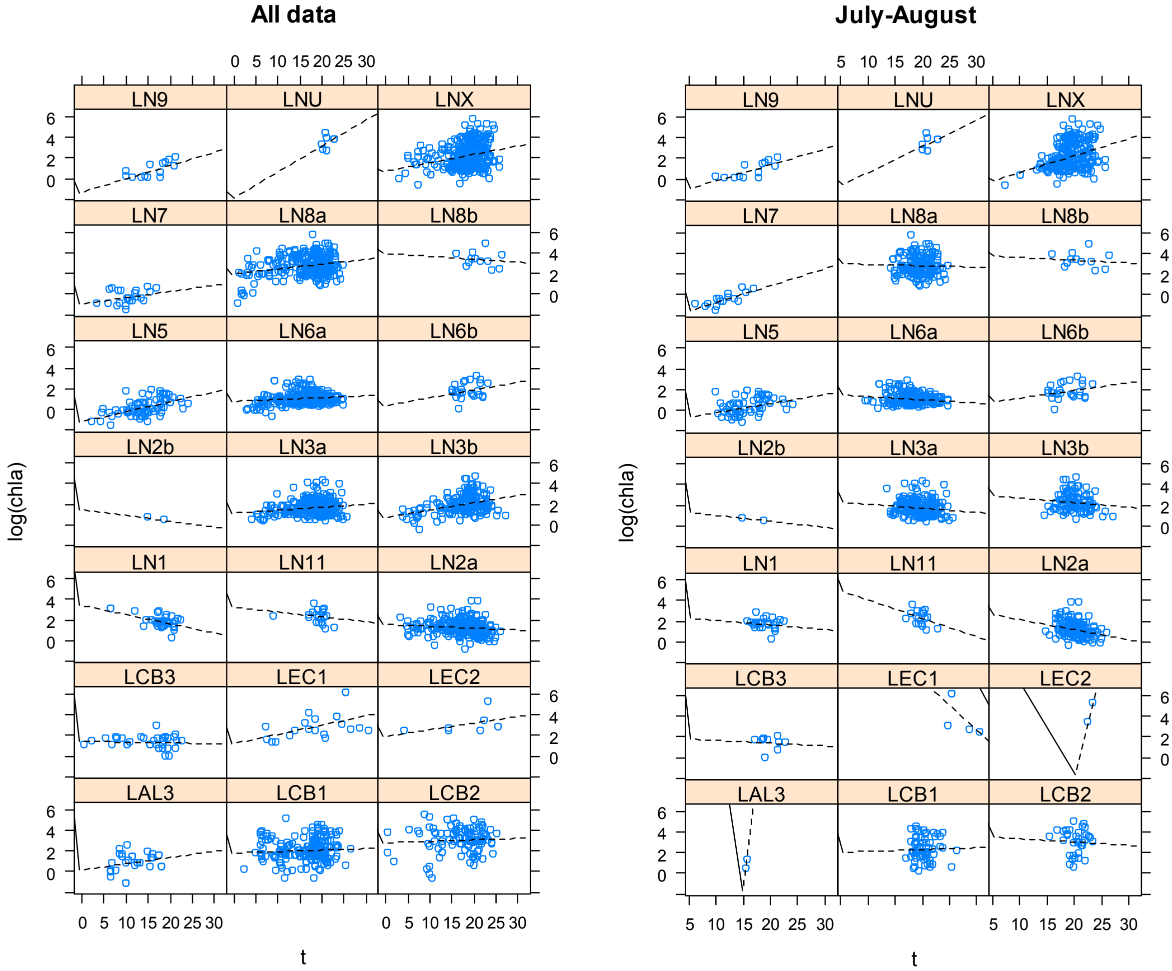
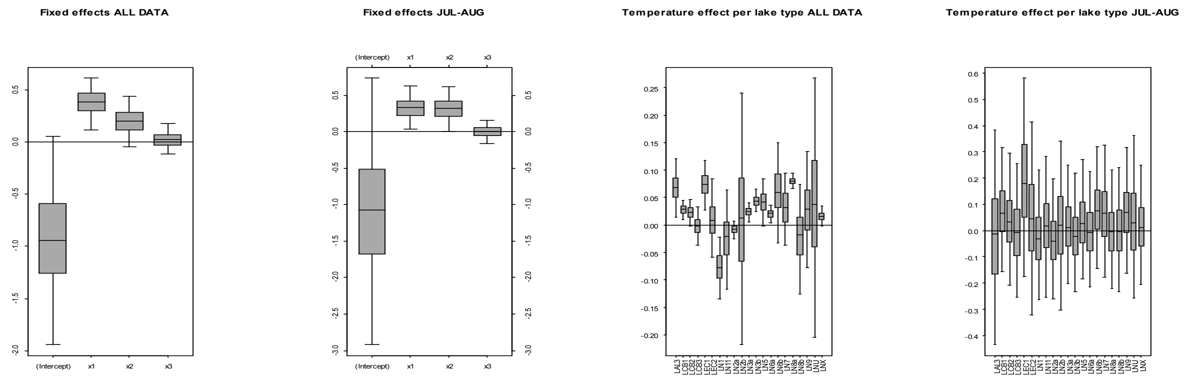


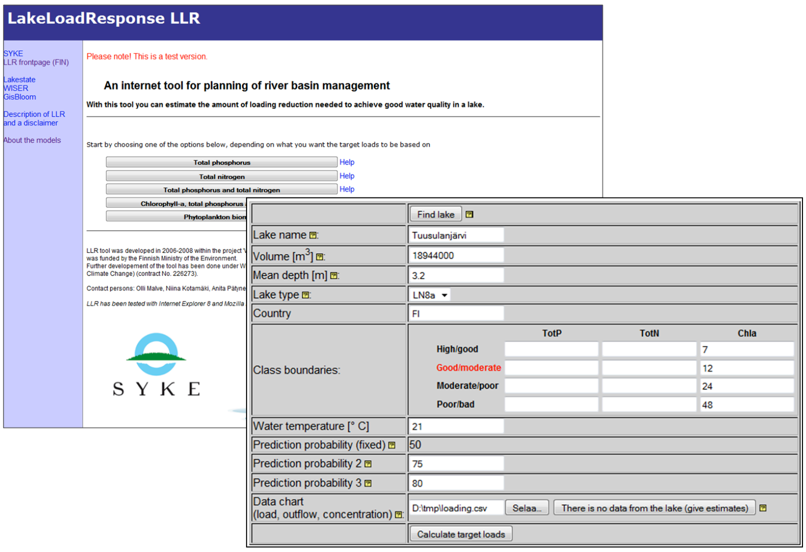
Figure 9: Figure 2 Scatterplots and linear regression lines (dashed line) of log(Chla) [µg/l] and temperature [°C] for lake types in both data sets.

The mixed effects models with random intercept and slope for lake types and random intercept for lakes were fitted to the whole data set and to July-August data in a Bayesian framework using Markov Chain Monte Carlo (MCMC) methods. This way the uncertainties of the model results can be properly taken into account. The box plots of the posterior distributions of MCMC runs separately for both data sets are shown in Figure 10. The fixed effects differ in the way that in the whole data the phosphorus effect is stronger than in the late summer data. Also the fixed global temperature effect is slightly clearer in the whole data than in the July-August data. Temperature effect in different lake types in July-August is not significant as could be anticipated. Although for the whole data set there are lake types that have significant temperature effect on chlorophyll a. E.g. for Northern GIG lake types (LN3a, LN3b and LN8a) the temperature effect is significant and positive.



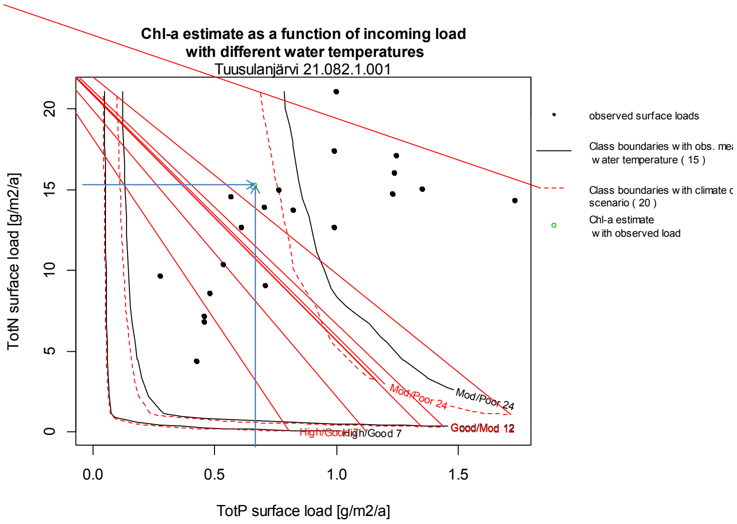
gure 10: Box plots of fixed effects (Intercept=constant, x1=log(totP), x2=log/totN9, x3=temp) and temperature effect in lake types for the whole data set and for July-August data.

The hierarchical chlorophyll a model with temperature effect was implemented in the LakeLoadRespose (LLR) tool. Open access internet tool LLR makes it easy to estimate needed reduction of nutrient load in a variety of climatic conditions. In Figure 11, there is a view of the LLR tool's front page and input form. First the user chooses which variable he wishes the loading reductions to be estimated with. The phytoplankton (chlorophyll a) model can be used to estimate the effects of the nutrient loading and climate change scenarios. At the moment the beta version of European wide LLR tool is available (http://lakestate.vyh.fi/).



gure 11: A view of the LLR user interface: the front page and the input form.

As a result LLR gives several figures that show the nutrient loading reduction with different risk levels and with different variables. The information of the loading reductions is also shown in table format. The chlorophyll a model results are shown with the model fitted to the whole data set. In Figure 12 is the main result of chlorophyll a model. From the set of level curves it can be seen the phosphorus - nitrogen loading combinations with which the chlorophyll a concentration stays below the good/moderate status class limit. The arrows indicate the model estimate of the chlorophyll-a concentration with present loading. The black solid lines denote the present water temperature situation (observed median for the lake) and the red dashed curves are in the situation of climate warming in the sense of +5°C warmer temperature. As can be seen the loading reduction must be bigger in the warmer lake water temperature.



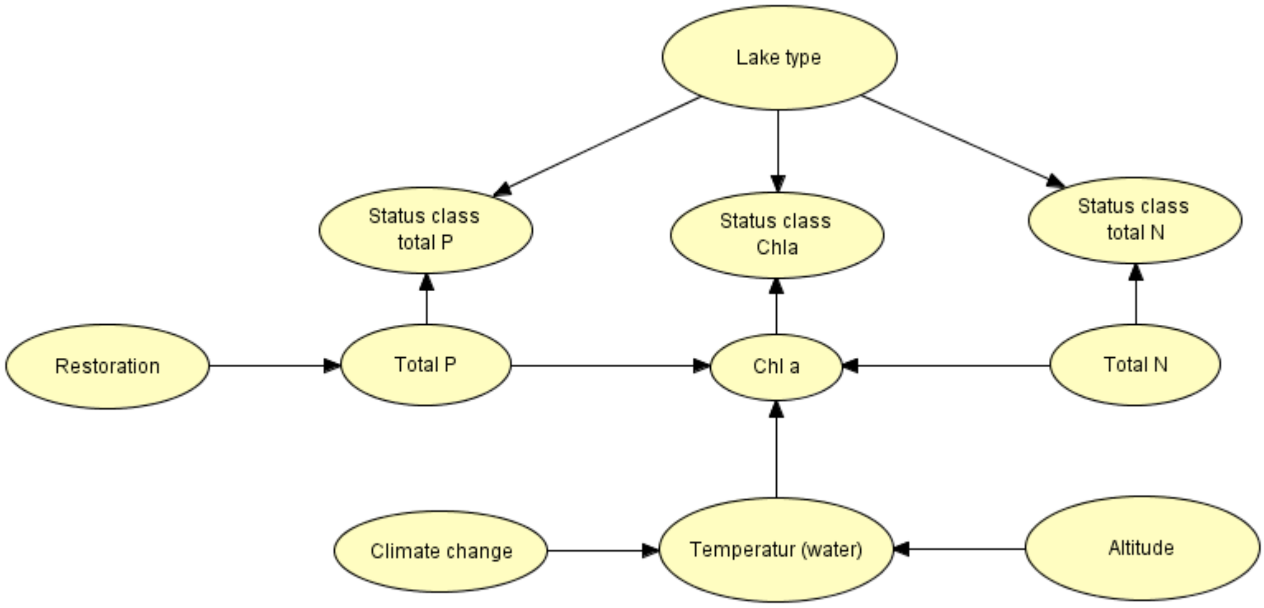
gure 12: *LLR result of chlorophyll a model: Estimate for chlorophyll-a concentration in Lake Tuusulanjärvi as a function of phosphorus and nitrogen loading (g/m²/a) to the lake for present temperature situatuin (black solid line) and for 5°C warmer water (red dashed line).*

Impacts of climate change and restoration on ecological status class: a management-oriented modelling approach

Climate change may impact ecological status directly and indirectly in multiple ways (Moe et al. 2010): e.g. by increasing physico-chemical pressures, by impacting the ecological baseline, by reinforcing the ecological response to a pressure gradient, or by reducing the ecological ability to recover. In this modelling study, we have focused on the combined impacts of restoration and climate change on ecological status based on phytoplankton (chlorophyll *a)*. 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. Our 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). We explored the impacts of these scenarios on the lake status class according to a biological quality element (phytoplankton) as well as to supporting physico-chemical elements (total P and total N). Since ecological classification of lakes is dependent on the lake type, we have selected two common lake types of Northern Europe as an example (L-N2a: altitude <200 m; L-N5: altitude 200-800 m).

Modelling approach

For this study we used a Bayesian network (BN) modelling approach. One of the proposed common approaches for all water categories (WPs 5.1-5.3) was development of conceptual models representing driver-pressure-impact-response-recovery chains. A Bayesian network can be developed as a conceptual model (Figure 13), but can also be parameterised and used as a simulation model. In brief, each variable (e.g. Total P, Chl-a) is illustrated by a node, which represents a discrete probability distribution (Figure 2). The cause-effect links are illustrated by arrows, which represent contingent probability tables (CPTs, Table 1). This modelling approach has many benefits, especially in relation to environmental risk assessment and management (Moe 2010): it can easily combine data or other information from different sources; it can explicitly model uncertainties (as probability distributions); and it can predict the probability of different outcomes of interest (such as different status classes).



gure 13: *Bayesian network representing Restoration (reduction in Total P loading), Climate change (increase in air temperature), and lake-type-specific assessment system for Total P, Total N and Chlorophyll-a.*

Table 1: Conditional probability tables. (A) Probability distribution of Temperature depends on levels of Altitude and Climate change. (B) Probability distribution of Status class for Chl-a depends on levels of Chl-a and Lake type.

(A)

Status class Chla

Lake type	L-N2a						L-N5					
Chl a	0 - 3	3 - 4	4 - 5	5 - 7	7 - 10	10 - 20	0 - 3	3 - 4	4 - 5	5 - 7	7 - 10	10 - 20
H	1	1	0	0	0	0	1	0	0	0	0	0
G	0	0	1	1	0	0	0	1	1	0	0	0
MPB	0	0	0	0	1	1	0	0	0	1	1	1

(B)

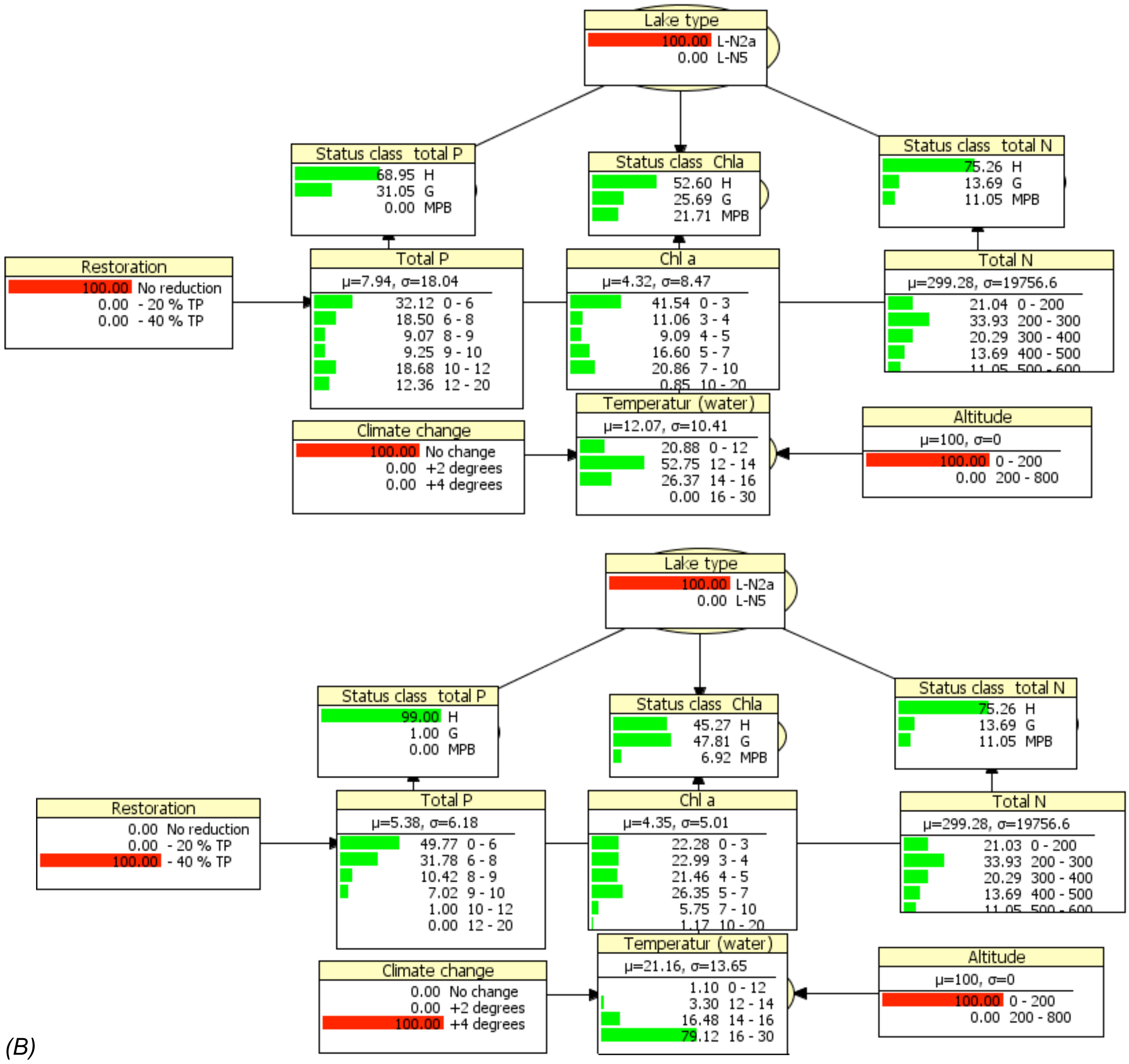
Temperatur (water)

Climate cha...	No change		+2 degrees		+4 degrees	
Altitude	0 - 200	200 - 800	0 - 200	200 - 800	0 - 200	200 - 800
0 - 12	0.209	0.566	0.066	0.242	0.011	0.121
12 - 14	0.527	0.434	0.099	0.242	0.033	0.091
14 - 16	0.264	0	0.527	0.515	0.165	0.242
16 - 30	0	0	0.308	0	0.791	0.545

Model construction and simulation

The construction of this Bayesian network model integrated two different lake models and data sources, from NIVA and SYKE respectively. The first part of the BN (effect of restoration and climate change on total P and temperature in lakes) was based on the "NIVA model", while the second part (effects of total P and temperature on Chl-a) was based on the "SYKE model" (http://lakestate.vyh.fi/).

(A)



(B)

gure 14: The Bayesian network (Figure 4) with probability distributions of each node. Red bars indicated levels selected by the user. (A) Model scenario with no restoration and no climate change (low-altitude lakes). (B) Model scenario with highest level of restoration and climate change (low-altitude lakes).

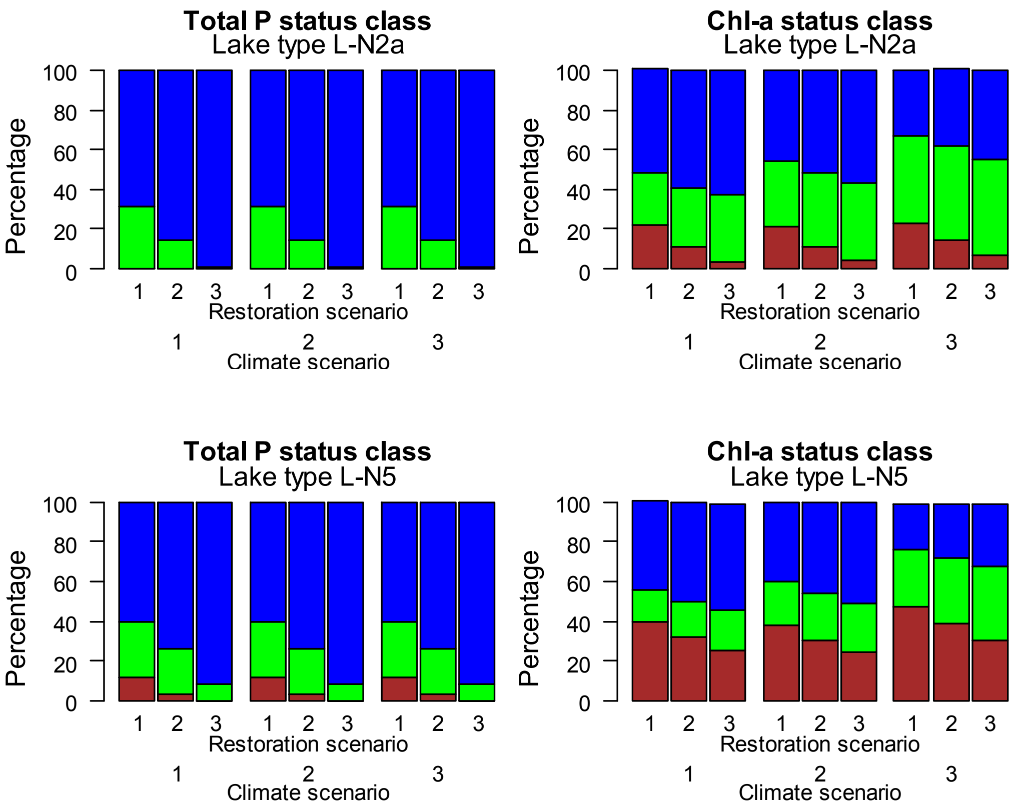
The NIVA model used the model code MyLake (Saloranta & Anderson 2007) to simulate climate and restoration impacts for a large number of Norwegian lakes, of which 124 lakes (types L-N2a and L-N5) were selected for this analysis. More detailed information about the simulations can be found in WISER deliverable 5.2-1 (pp. 76-78).

The SYKE model is a hierarchical model that simulates chlorophyll-a from nutrients and temperature observations, based on lake-type-specific estimations. 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. Only lakes with complete set of observations were used, which means that the dataset is not representative for the region in general (high-status lakes are overrepresented). For both data sources, average values for summer months were used in this study. Discretisation of continuous variables was based on (1) official class boundaries for TN, TP and chl-a (Finland), and (2) regression tree analysis for identifying threshold responses in the variables. The entries of the conditional probability tables (Table 1) were calculated as the proportion of data points falling into each combination of the parent node levels.

The BN model simulations were run by selecting a scenario and a lake type (marked red in Figure 14), and recording the resulting status classes according to TN, TP and chl-a. The model also provides expected value (μ) of e.g. chl-a, but with this modelling approach the probability distribution is more interesting than a point estimate.

Results

The predicted levels of nutrients and chl-a in this model exercise depends 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 (Figure 615). TN was not affected by restoration or climate scenarios in this model, and remained as shown in Figure 5 across all scenarios. TP status class responded to restoration (reduction of P loading) by increased probability of High status. The highest restoration level resulted in 30 percentage point (pp) increase in probability of High status for lake types L-N2a, and 32 pp increase for L-N5. TP was not affected by the climate scenarios, in accordance with the model settings. Chl-a status class also responded to reduction of P loading by increase in 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.



gure 15: Probability distribution of status classes for Total P (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.

Concluding remarks

This study has focused on high status of lakes, because the source dataset was dominated by good- and high-status lakes. According to the phytoplankton indicator in this BN model, a climate change of +2 °C would almost counteract the benefit from 20% P loading reduction, while +4 °C would more than outweigh the benefits from 40% P loading reduction. The risk of not meeting good ecological status was less affected in this study. Although "good ecological status" is the main WFD management goal, preservation of high status is also a WFD requirement.

As mentioned, this modelling exercise considers only climate change impacts on lake processes, notably phytoplankton growth rate. In reality one can expect additional impacts on river basins such as increased P loading. This study has therefore only explored one of many potential climate change impacts on ecological status of lakes.

The Bayesian network modelling approach presented here is very general, and can easily be extended to include more lake types and other biological quality elements, as well as different scenarios.

Management options for recovering shallow lake ecosystems under stress: case Lake Veluwe

Globally, efforts are made to improve the water quality and ecological status of water bodies. Often a range of management options is considered and cost benefit analyses are made to select the most appropriate set of measures. Unfortunately, the effects of climate change are often ignored when selecting measures for improvement of the ecological status of a water body. However, a myriad studies show that climate change interferes significantly with the physical, biogeochemical and biological aspects of ecosystems. In freshwater shallow lake ecosystems this leads to an increased risk of eutrophication and an expected decrease in piscivorous fish thereby favouring a turbid ecosystem characterized by small white fish (bream, perch) and an increased risk of cyanobacterial dominance in the phytoplankton composition (Jeppesen et al 2007; Kosten et al 2009; Kosten, 2011 and references therein).

Implemented measures are often expected to have (near-) immediate impact on the aquatic ecosystem. To determine the impact of the considered measures in relation to the autonomic developments (including climate change and land use changes) models offer a good option for analyses of longer time periods. Put this together, models have the potential to be a powerful tool to get insight in how effective current management is, what the impact of longer term climate change is and what the combined outcome is.

This study aims to demonstrate how deterministic modeling can be used to assess the relative impact of both the implementation of (short term) management measures and (long term) climate change on the water quality of a large shallow lake in the Netherlands, Lake Veluwe (system description see WISER Deliverable 5.2.1). We assessed the relative role of water management and climate change, expressed as a change in average temperature over the year, 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 modeling application (1985, stand alone BLOOM) (model description see WISER Deliverable 5.2.1).

Scenarios
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 on +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.

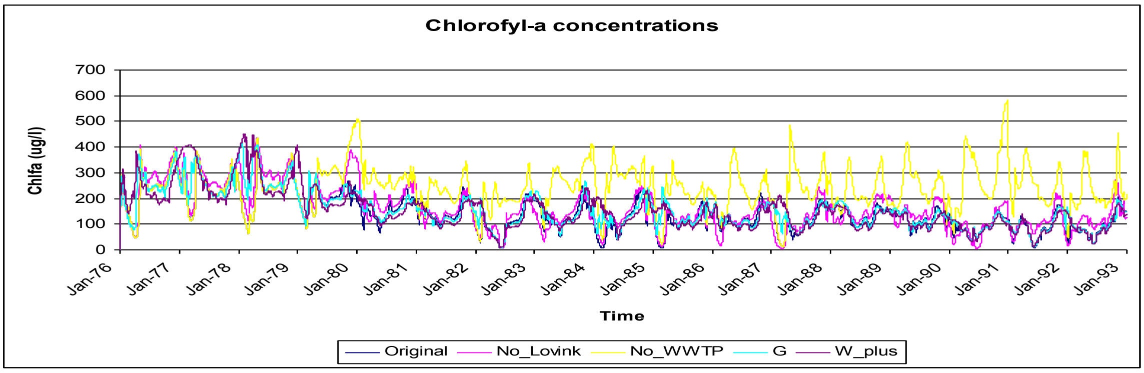
In the short-term runs three scenario types were performed: only an increase in temperature, only a change in nutrient concentrations and the combination of those two types of scenarios. Temperature scenarios that were used were: 1. +0.9°C (G), 2. +1.3°C (G+), 3. +1.8°C (W) and 4. +2.6°C (W+) (KNMI climate scenarios).

Nutrient scenarios were set for both increased and reduced concentrations of nitrogen and/or phosphorous relative to the reference run. Increased nutrient concentrations stands for increased eutrophication due to climate change and reduced nutrient concentrations for water management efforts. Increased nutrient concentrations were set on +10% nitrogen and/or +10% phosphorous and reduced nutrient concentrations on -10% nitrogen and/or -10% phosphorous

The combined effect of temperature increase and change in nutrient concentrations resulted in six scenarios, being: 1. +0.9°C and -10% nitrogen reduction, 2. +0.9°C and -10% phosphorus reduction, 3 +0.9°C and -10% nitrogen and -10% phosphorus reduction, 4. +2.6°C and -10% nitrogen reduction, 5. +2.6°C and -10% phosphorus reduction, 6 +2.6°C and -10% nitrogen and -10% phosphorus reduction,

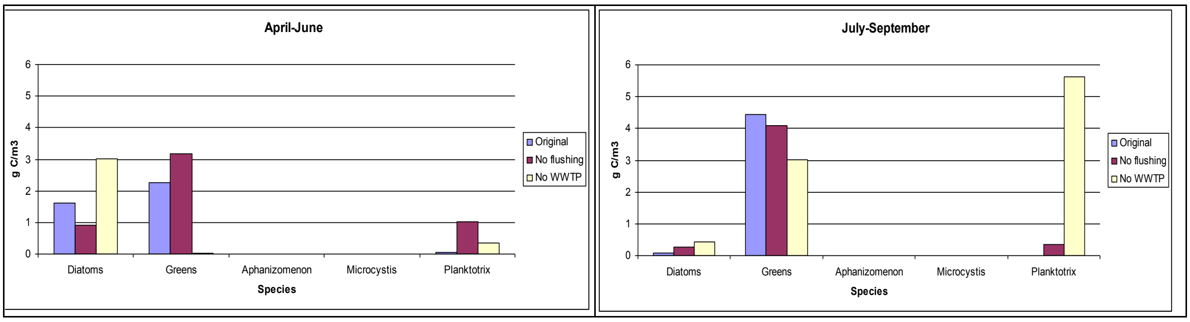
Boundary conditions
As this model study is meant merely to explore the relative contribution of climate change and water management measures to the water quality, some simplifications are made. Changes in solar radiation patterns, precipitation patterns, residence time (both climate and management induced), macrophyte growth and dynamic grazing were not taken into account.

Results
The hind cast of executed water management measures and the isolated effect of increased temperature are depicted in 16. It is evident that increased efficiency of WWTP has improved water quality more than the increased flushing measure. The effect of increased temperatures has less effect on water quality compared to the two water management measures taken.



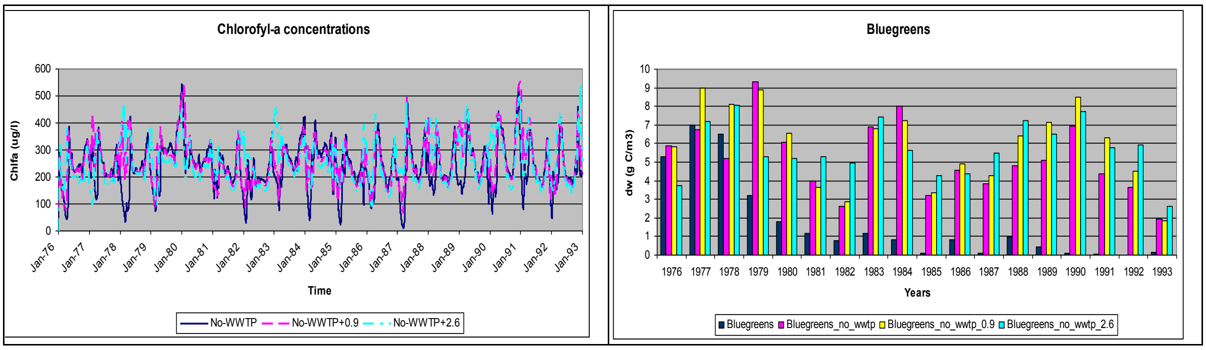
gure 16: *Chlorofyll-a concentrations (µg/l) for the original long-term run, this run without flushing by pumping station Lovink (NO_Lovink), without increased efficiency of the WWTP(NO_WWTP), a temperature increase of 0.9°C(G) and a temperature increase of 2.6°C (W_Plus).*

Next to the difference in the extend of effect between the two water management measures on chlorofyl-a concentrations, the composition of phytoplankton also differs, especially in late summer (**Fehler! Verweisquelle konnte nicht gefunden werden.**17), with more blue-green algae (*Planktothrix*) occurring in the calculations where measures were not carried out.



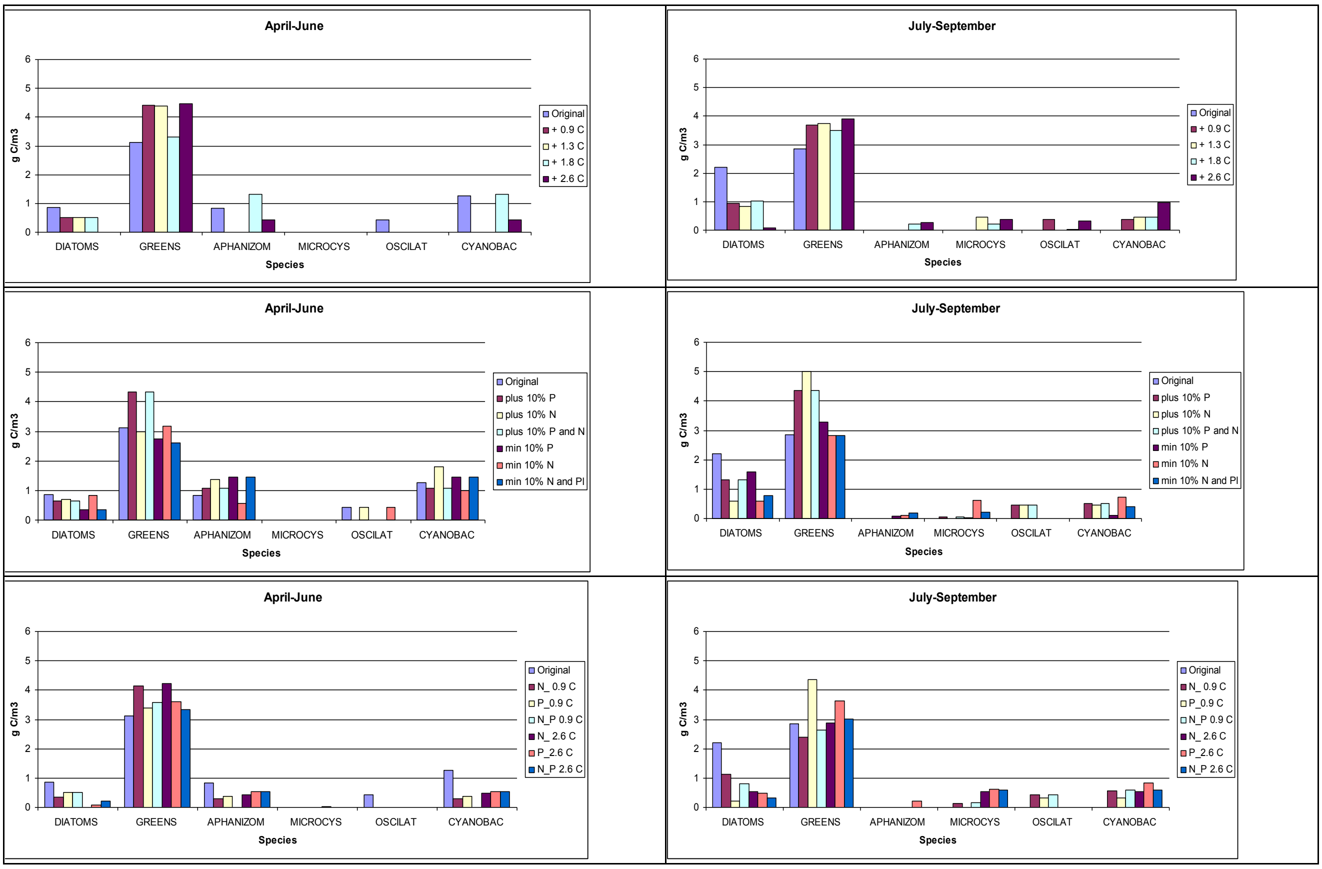
gure 17: *Changes in phytoplankton communities as a result of not carrying out flushing of the lake (red) or increased WWTP efficiency (yellow) in comparison to the original situation (purple) where measures were included.*

The combination of no increased efficiency of the WWTP and increased temperatures on chlorofyl-a concentration and the amount of bluegreen algae are depicted in **Fehler! Verweisquelle konnte nicht gefunden werden.**18. In the conducted scenarios, the biomass of bluegreen algae show deviations compared to the reference run. In the scenario runs bluegreens remain in the lake during the simulation period. **Fehler! Verweisquelle konnte nicht gefunden werden.**18 illustrates that the management measure has more effect on the algae composition than temperature increase. The latter has an additional effect most of the years. Remarkable is that the biomass of bluegreens is not per se higher in the +2.6°C scenario than in the +0.9°C scenario.



gure 18: chlorofyl-a concentrations (ug/l) for the scenarios: no WWTP, no WWTP in combination with a temperature increase of 0.9°C and no WWTP in combination with a temperature increase of 2.6°C (left) and the biomass of bluegreens (dry weight in gC/m^3) in the reference run and in the scenarios: no WWTP, no WWTP in combination with a temperature increase of 0.9°C and no WWTP in combination with a temperature in crease of 2.6°C (right).

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 in temperature and reduced nutrient concentrations result in different phytoplankton biomass and to a larger extend to changed phytoplankton composition (**Fehler! Verweisquelle konnte nicht gefunden werden.**19). In the summer half year in almost all scenarios of diatom biomass are less than in the reference run. In the period April until June, an increase in temperature leads to comparable bluegreens biomass as the reference run (+1.8°) or lower (+2.6°C) or no bluegreens 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 bluegreens. In late summer, all scenarios lead to higher bluegreen biomass than the reference run. Increased temperatures and reduced nutrient concentrations lead to a phytoplankton composition that is not evidently due to increased temperatures or reduced nutrient concentrations: it is an interplay between those two steering factors.



gure 19: Effect on phytoplankton biomass and composition in the period April to June (left) and July to September (right), due to increased temperatures (upper panel), changed nutrient concentrations (mid panel) and increased temperatures and reduced nutrient concentrations (lower panel).

Conclusions

We showed that a long-term deterministic model can be used to assess the effects of excluding measures from the calculation by running scenarios in which a certain measure was not carried out in comparison to runs validated with monitoring data in which the measure was implemented. The results of this omission of measures showed effects on the water quality that are in line with the expectations, with higher chlorophyll-a levels and changes in phytoplankton community structure. Furthermore, it enables us to make a comparison of the relative importance of climate change (expressed as an increase in temperature).

Stand alone BLOOM simulations show that in winter half year an increase in temperature is more important than nutrient reductions. However, during summer half year there is interplay of increased temperature and reduced nutrients, but changes in nutrients seem to have a larger effect. The switch in relevance of temperature and nutrient changes can be explained by the concept of limiting factors. In winter half year temperature is often an important factor in regulating phytoplankton growth in combination with phosphorous limitation. In summer half year, mostly nitrogen and phosphorous are limiting algae growth. However, it depends on the limitations in a lake system, how the effects of climate change and water management on water quality work out. In this case study, we examined a most of the time nutrient limiting lake, but lakes that have light limitation are also common and the same scenarios as presented in this study can lead to other outcomes.

This study demonstrated that deterministic models can be a powerful tool to examine the effect of water management measures in combination with climate change, used here as being temperature increase. It helps to assess the relevance of climate change for the success of an intended water management measure to improve a water body's ecological status. This is of meaning as the effect of climate change can be of minor or at least as important for water quality as water management measures. Moreover, we demonstrated that there are synergetic and antagonistic effects of the combined climate change and water management scenario's. Therefore it is key to consider the combination of climate change and water management option, e.g. as the effect of climate change can cause a change in the ecological status which can be amplified (positive or negative) by certain water management measures or the full potential of water management measures is not reached because of interference of climate change. Thus, deterministic models are of use exploring the extend of effect of intended water management measures in the light of climate change as both processes are highly likely to interact on water quality level..

Key messages:

- Direct measures to improve water quality can have a strong and pronounced effect on primary productivity and ecological status. Climate change (expressed as an increase in average temperature), however is more gradually and can have synergetic as well as antagonistic effect when combined with direct measures taken and therefore need to be considered while planning water management measures.
- Future increase in water temperature due to changing climate, will have a significant impact on both ecosystem processes and species diversity and abundancy.
- Well calibrated and validated deterministic models can help provide insight in the effect of direct measures and assess the impact of longer term scenarios for both land use changes and climate change.

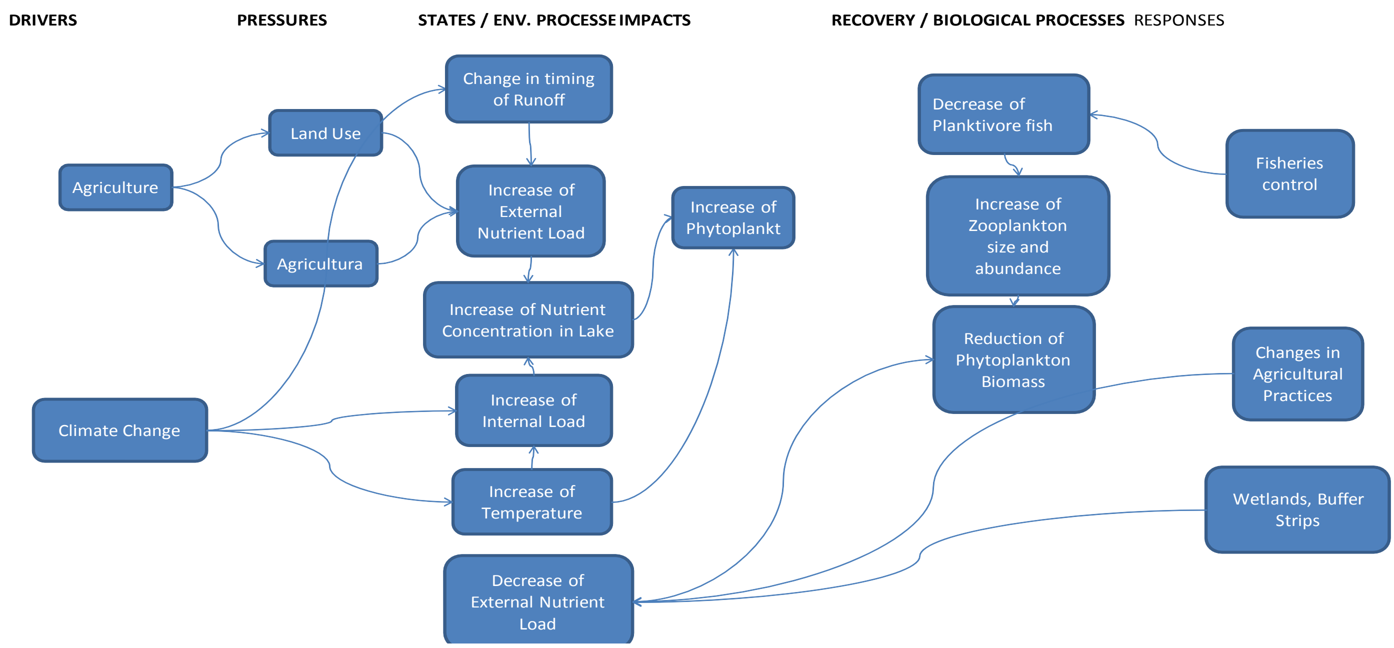
Recommendations for the management of Finnish case study Lake Pyhäjärvi

Lake Pyhäjärvi: the good ecological status in terms of phytoplankton biomass can be achieved most efficiently by reduction of external nutrient load and fisheries management. Warming climate will have an adverse effect which can be compensated with additional load reduction and fisheries management.

Lake Pyhäjärvi is shallow (mean depth 5.4), mesotrophic lake and it has regular algal blooms. The lake is loaded by phosphorus load from agricultural practices and warming climate is likely to intensify algal blooms. A strong correlation between Planktivorous fish stocks, zooplankton size and abundance and phytoplankton biomass has been detected. The management of algal blooms has been based on control of agriculture and fisheries.

The main drivers in the conceptual model (Figure 20) are agriculture and climate change. The former is increasing external phosphorus load and the later is increasing temperature of lake water. As a result, internal loading, phosphorus concentration and phytoplankton growth are accelerated. Responses are the control of fisheries and agriculture and the construction of wetlands and buffer strips which have direct effects to phosphorus concentration and fish stock and mediated effect to phytoplankton blooms.

Evidence for causal inference is based on the experience gained from 30 years of management, monitoring, research and modelling of lake.



gure 20: Drivers, pressures, states, impacts, responses and recovery of Lake Pyhäjärvi.

The main degradation process observed and validated using Coherens and LakeState models are the accumulation of phosphorus in sediment, the intensification of internal load and increase of phytoplankton biomass. As shown in numerous studies the size of planktivorous fish stock has also a definite effect on phytoplankton via decreased size and abundance of zooplankton. Internal load of phosphorus buffers effectively the impact of external load reduction measures.

The warming climate increases detrimental algal blooms via intensifying internal load and phytoplankton production.

The increase of daily main air temperature in South-Western part of Finland will be 5 0C by the end of this century. The increase will be more pronounced in winter time as in summer. The amount of annual rain will increase by 17 %. The increase will be more significant during the winter time than in summer. Some models gave a result that the precipitation will decrease during summer and there will be drier summers than at present.

The lake model predictions show that the ice cover length will decrease by several weeks, especially in spring time. The lake surface water temperature increase will follow the air temperature increase. The catchment model predictions show that peak flows of rivers will increase towards the end of the century and a clear shift towards winter floods will be seen. The change is stronger in Yläneenjoki as in Pyhäjoki. Similarly the time distribution of suspended solids and phosphorus load are markedly shifted towards winter time, especially in Yläneenjoki.

The main ecological response to increased phosphorus loading is the excessive abundance of phytoplankton. No ecological threshold has been detected.

The management options relevant for Lake Pyhäjärvi are

1. minimizing the external loading by
    - implementing buffer zones as in master plan on 4152 fields with total surface area of 17500 ha
    - increasing the winter time vegetation cover by 30, 50 and 70% from present.
    - changes in animal husbandry and reduction of manure usage on fields.
2. biomanipulation by harvesting pelagial fish

The decrease of phytoplankton due to modified practises of agriculture and fisheries has been revealed by observational and modelling studies.

Guidlines for model usage

Recommendations for model usage were formulated based on the feedback from stakeholders:

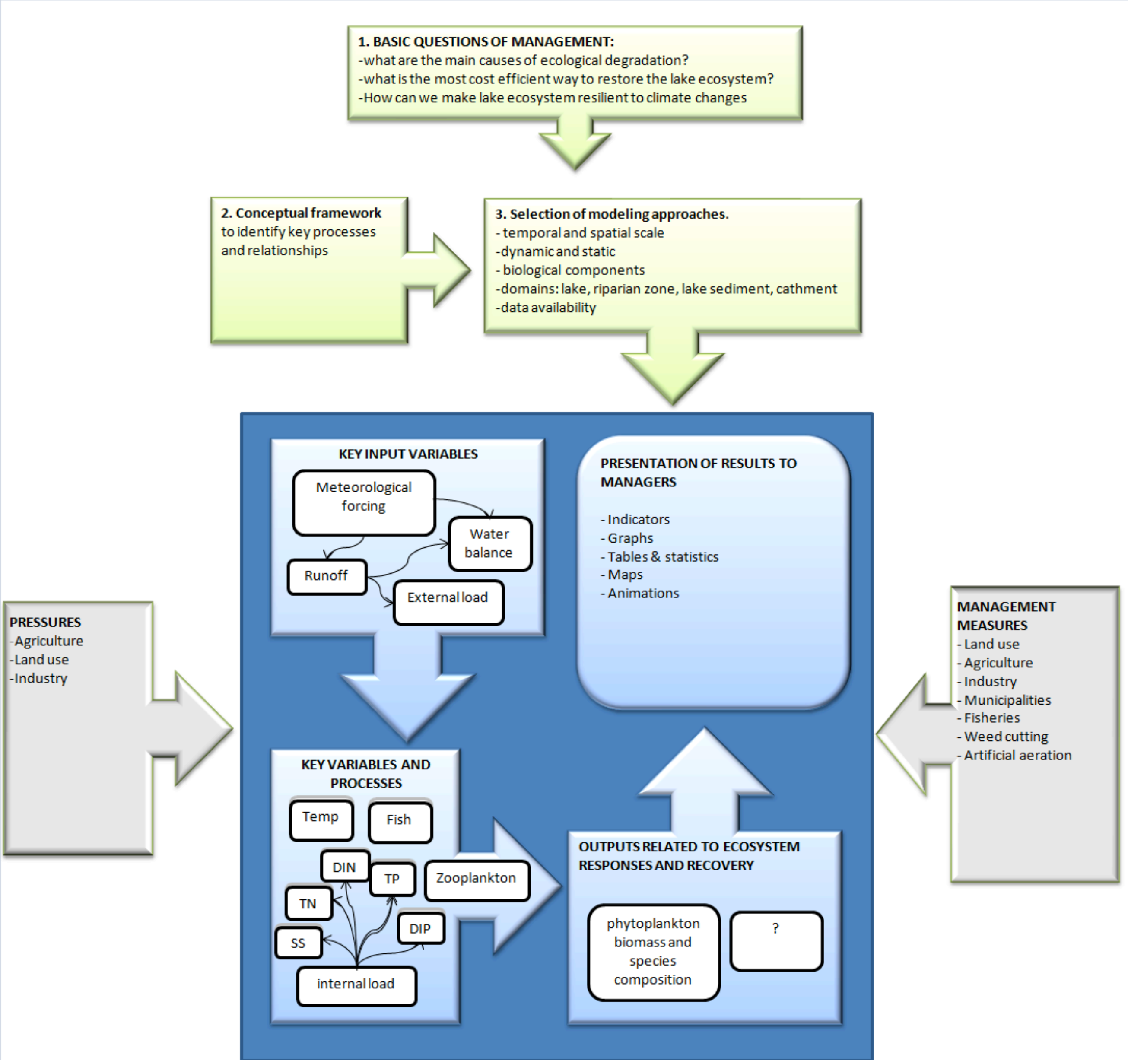
1. Close co-operation with end users in the formulation of management questions and the conceptual framework.
2. Designing conceptual models together with stakeholders to have a common understanding and to create a clear picture about the issues that should be addressed
3. Non scientific reporting of impacts of management measures directly to the end users
4. More time and resources to collateral data collation and analysis together with end user
5. Flexible selection of temporal and spatial scales
6. Use of bayesian networks throughout the modeling process in the analysis and dissemination of ecological and social impacts.
7. The storing of input and output data into a generic data base.

Selected model selection criteria and modelling guidelines revealed different aspects in model selection and use: a. TMDL criteria (Reckhow 2001) - policy-relevant functioning of simple probabilistic lake models b. BMW criteria (Saloranta et al 2003) - technical standard of sophisticated hydrodynamic models c. HarmoniQuA guidelines (Refsgaard et al. 2005) - communication between modeller and user during the modelling process.

Usability of case study models in design and decision making was evaluated with the good modeling practice (comparison of criteria of modelers and stakeholders) developed in HARMONICA-project as described in our deliverable D 5.2-1: Analysis of applied modelling approaches in the case studies. The feedback from the stakeholders of Lake Pyhjärvi about models and their usage were gathered in a workshop. Lake and catchment models applied to the region were presented to the participants and resulting feedback was analyzed to formulate recommendations for model usage. The criterias are listed below:

- The integration of models and simulations
- Communication of uncertainty
- Data demands
- Public participation
- Communication with model users
- Delivery of knowledge rules
- Usability in design and decision making
- Climate change scenarios
- Management and restoration scenarios

A good modelling practise include following steps as indicated in Figure 12. It is guided by indentified pressures and management measures which lead to the basic questions of management, definition on framework, selection of modelling approach and to identification of key input and output variables and processes.



gure 21: Steps of modelling according to the good modelling practise.

Discussion and summary

Evidence for climate change effects

Concerns about the implications of climate change on the WFD are reflected in a guidance document published by the European Commission which considers how river basin management can accommodate changing climate conditions (European Commission, 2009). The report highlights several cases where evidence suggests that anthropogenic climate change is already having a significant impact on European water bodies (e.g. Durance & Ormerod, 2007) but argues that, generally, within the timescale of the implementation of the WFD (i.e. up to 2027) and for the metrics used for assessment, it will not be possible to disentangle climate effects from those caused by other pressures. Only those systems close to key temperature or hydrological thresholds are likely to be vulnerable to climatic shifts in the early WFD river basin management cycles. The report concludes that reference conditions and targets should not be revised in the light of climate change scenarios over the timescale of WFD implementation unless there is overwhelming evidence to justify this.

However, there is increasing evidence that aquatic ecosystems are already responding to changes due to global warming, particularly vulnerable high altitude and high latitude regions (Catalan et al., 2009, Jeppesen et al, 2009,2012). Analyses of long-term datasets show clear warming effects on lake fish assemblages across Europe with major changes in fish assemblages composition, size and age structure, loss of cold stenothermic Arctic charr, reduced trout harvesting and increases in eurythermal species. (Jeppesen et al 2012). Slight increases in the temperature of French rivers are predicted to reduce the abundance of salmon irrespective of how other catchment pressures are managed (Pont et al., 2006) with the implication that the currently definition of reference conditions for trout in these systems will need to be revised as well as the boundaries between status classes. Water temperature drives many processes operating in aquatic ecosystems such as fish migration and the phenology of food-web interactions (e.g. Davidson and Hazelwood, 2005; Durance & Ormerod, 2007). Meeting WFD objectives relative to reference conditions will be much more difficult where increased temperature effects begin to override existing restoration measures.

Synchronous change - climate change and other stresses
In addition to the direct impacts of climate change through, for example, increased water temperature or changes in stream discharge regime climate change interacts with existing pressures and evidence suggests that where there are climate change effects occurring currently, these are more often than not operating in tandem with pressures from other human activities and this is likely to be the case for the foreseeable future.

With regard to the interaction between climate and eutrophication, for example, one of the key questions is whether nutrients structure ecosystems in different ways under current and anticipated future climatic conditions? There is a clear change in trophic structure in lakes along climatic gradients from simple, often elongated food-webs in cold climates to truncated webs in warmer systems with a higher degree of omnivory. With increasing temperature, European lakes are likely witness significant impacts on phytoplankton, zooplankton, fish and macrophyte populations with implications for ecological status and biological assessments, particularly where there are barriers to migration. As well as offsetting or modulating the potential recovery following restoration efforts, there will certainly be a deviation from notional reference conditions. Recovery from acidification may also be compromised through for example an increase in acid pulses as a result of more extreme precipitation events or remobilisation of catchment nitrogen. Climate change can also influence the distribution patterns and mobility of organic pollutants and toxic metals and lead to changes in the uptake and accumulation of these substances in food chains as well as the remobilisation of legacy pollutants stored in catchment soils.

The interaction between these different drivers means that it is often difficult to disentangle the effects of climate change from those of existing pressures. Improving our understanding of these synergistic effects will be crucial in terms of developing management strategies in future, particularly with regard to identifying restoration targets to achieve WFD compliance.
Whether the effects are direct or indirect, climate change may result in those water bodies positioned near the type boundaries to change their type and water quality parameters may be particularly vulnerable in this respect (Nõges et al., 2007). This deterioration of water quality within the time frame relevant for WFD implementation may be sufficient to threaten compliance with

water quality objectives. Additionally, where type switches occur the good status criteria for the altered type would have to be applied. In most cases, however, climate change impacts will not necessarily result in changes in water body type, rather changes in water temperatures or discharge although as noted, these may interact with existing stresses within the catchment or from atmospheric sources and exacerbate the impacts of such pressures.

Climate change induces shifting targets?

Although there is little direct evidence that climate change has altered baselines so that new system equilibria have resulted, independent of the effects of existing pressures, increasing numbers of studies have identified cases where, despite measures taken to combat the effects of stresses such as acidification or acidification, the recovery trajectory is not indicating a return to pre-impact reference conditions. This may be due to lag or hysteresis effects, ecological constraints or the influence of other confounding pressures such as nitrogen deposition. But there is increasing evidence to demonstrate that climate is a key factor influencing the success of restoration efforts and therefore the value of the reference condition concept in determining management targets. Studies of the recovery of boreal lake ecosystems from acidification (Stendera & Johnson 2008) showed decreasing acidity with changes in phytoplankton and littoral invertebrate assemblages indicating signs of recovery from acidification. However, other changes unrelated to changes in acid deposition occurred including changes in phytoplankton and littoral invertebrates in the reference lakes, probably caused by climatically-related changes in water colour and temperature. Decreases in the richness of sublittoral and profundal invertebrate assemblages may have resulted from climate-related change acting on habitat quality, such as ambient oxygen concentrations and temperature, rather than changes in lake acidity. Palaeolimnological analysis of the sediment record of a large shallow lake concluded that on a decadal-centennial scale, the nutrient enrichment signal overshadows the potential impact of climate as a control on the diatom community. However, at an inter-annual scale, changes in species composition that may be attributed to climatic controls (Bennion et al., 2011). Numerous studies now illustrate how certain climate change impacts on biological quality elements cannot be mitigated by river basin management measures. For example, it has been noted that climate change may have a significant effects on hypolimnetic dissolved oxygen concentration in lakes (Jeppesen et al, 2012), having deleterious effects on habitats for benthic macroinvertebrates (Quinlan et al., 2002) and fish (Nurnberg, 2004).

These and other studies illustrate the complexity of ecosystem response to synchronous changes in different drivers, and the difficulty of disentangling the effects of multiple, interacting pressures. For lakes in general future climate change is likely to change the physical (through stratification and duration of ice cover) structure, food web structure, biogeochemical processes and land water interactions dictating ecosystem response preventing reestablishment of communities found over the past centuries (Battarbee et al., 2005) If systems are changing so that species assemblages during the recovery phase contain different species mixes or relative abundances in comparison with the compositional change during the equivalent degradation stage, then the water managers may be faced with water quality targets that are impossible to achieve.

Modelling of climate change and of management effects

A wide variety of models were used and demonstrated in this work package including statistical, probabilistic and mechanistic models and the different approaches all have their merits.. Simple statistical and probabilistic models are designed for data based inference, uncertainty management and decision making whereas complicated mechanistic models can store and synthesize knowledge of a whole spectrum of pressures, variables, processes and scales available for the analysis of management and climate change scenarios.

According to our mixed linear chlorophyll *a* model fitted to data from 351 European lakes, there are clear climate change effects on eutrophication although the effect was masked by the variability of other pressures such as nutrient load and runoff. It was concluded that under warmer climatic conditions, a greater reduction in nutrient loading may be needed to attain good ecological status in a lake. This is in accordance with the model simulations s from Lake Pyhäjärvi, results from fish and zooplankton studies and the aforementioned conclusions.

The LLR (LakeLoadResponse) internet tool (http://lakestate.vyh.fi/cgi-bin/frontpage.cgi?kieli=ENG) demonstrated in WP5.2 produces water quality predictions with statistical confidence intervals to inform management strategies. While uncertainty and risk management and ease of model use have drawn growing attention in lake and river basin management, this will be useful in terms of estimating reduction requirements for nutrient load at present and in changed climate conditions.

Management-oriented Bayes networks demonstrated in this work package have many benefits, especially in relation to environmental risk assessment and management. The approach can combine data or other information from different sources, quantify model uncertainties (as probability distributions) and predict the probability of different outcomes of interest (such as different status classes). Predicted impacts of climate change and restoration measures on ecological status class showed that a climate change of +2 °C would almost counteract the benefit from 20% P loading reduction, while +4 °C would more than outweigh the benefits from 40% P loading reduction. The Bayesian network modelling approach proved is very general, and can easily be extended to include more lake types and other biological quality elements, as well as different scenarios.

The modelling of management options in case study lakes revealed the most successful management options for eutrophication but gave support to the need for additional measures to counter climate change effects. In addition, the demonstration to stake holders of management under climate change provided useful feedback and guidelines for model use in management.

Scenarios from Lake Veluwe and Lake Pyhäjärvi were well in accordance with conclusion drawn above in this section. The recovery of shallow lake ecosystems from eutrophication - even when eutrophic conditions have lasted for decades- requires both internal and external nutrient loadings to be reduced significantly. Recovery processes may need a long time, even when ecosystem shifts are facilitated by human interventions (e.g. biomanipulation, eco-engineering, building with nature, eco-innovation). Macrophytes play an important role within this recovery process. In deeper lakes, recovery is more complex. Future increases in water temperature due to changing climate, will have a significant impact on both ecosystem processes and species diversity and abundance.

Recommendations for the management of Lake Pyhäjärvi, Finland were also well in accordance with the conclusions above. Good ecological status in terms of phytoplankton biomass can be achieved most efficiently by reduction of external nutrient load and fisheries management. A warming climate will have an adverse effect which can be compensated with additional load reduction and fisheries management.

Recommendations for future research

A major challenge is to relate observed changes to climate as there have been many other changes to freshwater systems that have occurred during the period of climate change. These other influences, principally from land-use change and from pollution, are currently far stronger and more explicit than climate change and may mask its effect. Research is needed to understand how climate change will affect the ecological thresholds currently used to set the good/moderate status class boundary. For systems close to the boundary a relatively small shift in climatic conditions may be sufficient to alter the classification of a lake from good to moderate status. There is also acknowledgement that the reference condition is not static or oscillating around long-term means, but showing monotonic climate-driven trends. Hence approaches that explicitly include inter annual variability in weather and/or long-term trends in climate should be given greater consideration. Research is needed to disentangle the effects of short-term climatic events such as the influence of inter annual NAO oscillations from long-term climate-driven trends would be of interest. This should be underpinned by long-term monitoring programmes to continue (or be implemented) at reference sites across Europe to increase our understanding of the response of these systems to climate change. Analyses of such long-term datasets will be needed to support arguments for adopting revised reference conditions and restoration targets,

There is a need to adapt baselines, or more accurately, target status objectives, to accommodate those effects of climate change which cannot be mitigated by programmes of measures that can be carried out by water managers. For this to succeed there is a need for increased understanding of how climate change, through its interaction with other stressors such as eutrophication and independently as a pressure in its own right, affects the ecological status of water bodies and may impact on restoration efforts in future. It may not be possible to achieve restoration targets simply by reducing existing pressures as these pressures may be exacerbated though interactions with climate change.

The concept of a flexible reference condition could be underpinned by periodic monitoring of reference sites allowing adjustments of the values of reference conditions to be proposed which account for long-term natural processes. The WFD requires reviews of river basin characterization every six year. These could incorporate re-evaluation of reference conditions depending on changes observed at pristine reference sites. If required, the restoration targets (i.e., good ecological status) could also be evaluated periodically although there will be difficulties in disentangling climate change effects from existing pressures in the catchment (Nõges et al., 2007).

Fish, invertebrate and macrophyte communities will change in response to climate change. Metrics based on indicator groups and species are thus likely to become redundant. Ecosystem structure and process-based metrics are likely to become more relevant. Therefore it may be possible to include climate change effects by assessing the impact of climate change on existing WFD metrics and then adjusting the existing assessment systems accordingly. Another approach is to add 'climate specific components' to assessment systems, (i.e. metrics which account for the temperature sensitivity of species, Hering et al., 2010). Natural variation and climatic trends can be incorporated into the reference condition using a multivariate analysis, yielding a table or a nomogramm showing the relationship of the specific quality parameter to the most important natural factors (JRC, 2005).

The extent to which recovery can result in attaining a reference condition is dependant both on the reduction in pressures and also whether the pre-defined reference condition remains relevant as a target. Gradual, sustained changes in lake systems resulting from natural processes (e.g. lake infilling) may mean that recovery targets based on a fixed baseline are impossible or extremely costly to achieve. A similar outcome may result from anthropogenic pressures such as the chronic base cation depletion of catchment soils as a result of acid deposition or eutrophication driven changes to ecosystem structure and function. In future, climate change may potentially modify the fundamental behaviour of freshwater ecosystems to a point for some ecosystems where an historical reference state can no longer be a realistic target for recovery (Battarbee et al. 2005). It will not be possible, for example to restore winter ice-cover or freshwater conditions after inundation following a rise in sea-level. Nevertheless, the reference concept remains valuable in terms of understanding the current status of a lake and in providing a benchmark, if not a target, for recovery (Bennion et al., 2011). Figure 1 shows not only the need to reduce all pressures to reach a reference-based recovery target, but also the need to re-define the reference as boundary conditions change. The diagram illustrates in this case the use of the historically defined reference, not as a target for recovery, but as the starting point for modelling the probable future impacts of climate change on the reference condition (cf Battarbee et al. 2008).

The need to understand whether, and how, climate is affecting aquatic ecosystems independently or through its interaction with other stressors is of critical importance. In particular, policy makers and managers need to know whether climate change is exacerbating existing pressures or creating additional problems. This is a key factor in determining how management strategies can be adapted to accommodate climate change, in particular the reference condition concept. Although there is a need to embrace the concept of the dynamic reference condition, this should not be seen as a 'get out clause' or a means of seeking exemption from WFD compliance. Climate change should be considered an anthropogenic stress within the context of the WFD even if those tasked with implementing the Directive are unable to mitigate some of the direct effects. Where climate change results in increasing pressure from existing stresses, more action is needed to reduce the impact of the existing stress. In the case of nutrient enrichment, where increasing temperatures are likely to result in further eutrophication the response should be to further reduce catchment nutrient loading rather than revise the reference condition. Where the direct effects of temperature dominate it will be important to minimise the impacts of other stresses to increase system resilience. Only when climate changes induces ecological thresholds to be crossed, recovery trajectories to deflect away from the reference condition or novel ecosystems be created following climate driven species migration should achievable targets be reviewed and the reference condition be revised accordingly.

With climate modelling studies indicating that even if greenhouse gases were stabilised at present levels, future climate change is inevitable as the climate system adjusts to emissions that have already taken place. As emissions are expected to rise until at least the middle of the century, stabilisation will occur, if at all, at significantly higher levels than present. As a consequence there will inevitably be adverse effects that cannot be avoided, even with coordinated action at a European level and environmental legislation needs to be adapted taking this into account. Although climate may play a comparatively minor role in the early WFD management cycles (in comparison with other factors) in the longer term climate change impacts may influence proportionality of costs of measures necessary to reach good status, and may thus play a role with respect to exemptions (extended deadlines or less stringent objectives).

Guidelines for model usage

Climate change impacts should be taken into account in modelling and analytical assessments underpinning management . Statistical and probabilistic modelling for risk assessment and decision making should be further promoted in lake and river basin management. General recommendations for model usage were formulated based on the feedback from stakeholders:

1. Close co-operation with end users in the formulation of management questions and the conceptual framework.
2. Designing conceptual models together with stakeholders to facilitate a common understanding and to create a clear picture about the issues that should be addressed

3. Non scientific reporting of impacts of management measures directly to the end users
4. More time and resources for joint data collation and analysis with end user
5. Flexible selection of temporal and spatial scales
6. Use of bayesian networks throughout the modeling process in the analysis and dissemination of ecological and social impacts.
7. Securing input and output data in a generic data base.
8. Good modelling practise should be applied carefully

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