SEVENTH FRAMEWORK PROGRAMME
THEME 6: Environment (including Climate Change)

Adaptive strategies to Mitigate the Impacts of Climate Change on European Freshwater Ecosystems

Collaborative Project (large-scale integrating project)
Grant Agreement 244121
Duration: February 1st, 2010 – January 31st, 2014

Deliverable 3.21: Guidelines for the development of adaptation, mitigation and restoration strategies (for lakes)

Lead contractor: **Estonian University of Life Sciences (EMU)**
Other contractors involved: **NIVA**

Due date of deliverable: **Month 48**
Actual submission date: **Month 48**

Work package: 3
Contributors: **Peeter Nõges, Tiina Nõges, Jannicke Moe and Anne Lyche Solheim**
Estimated person months: 6

Project co-funded by the European Commission within the Seventh Framework Programme (2007-2013) Dissemination Level (add X to PU, PP. RE or CO)		
PU	Public	X
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)	

Abstract

The present Guidelines summarize the recent achievements in lake research including the key results of the Project REFRESH and based on those formulate some robust guiding principles for lake managers which

- recall some of the basic rules in limnology regarding nutrient limitation, cascading effects in foodwebs, and type-specific differences of lakes,
- put a clear emphasis on needs and cost-effective methods to combat eutrophication as the dominating pressure on lakes in Europe,
- if followed, will lead to win-win solutions.

Similarly to the CIS guidance (CIS, 2009), the guiding principles are followed by suggested actions and case studies or examples for more details. The four guiding principles relate to various aspects of the river basin management - the risk assessment, monitoring and assessment of the status of surface water, objective setting, and the programme of measures. The implications of the principles for lake restoration as well as for CC mitigation and adaptation are highlighted.

Contents

1. Introduction

1.1. Basic scheme of CC policy making

Policy making in lake management aims at sustainable maintenance and, if needed, improvement of ecological status and services offered by lakes by cost-effectively minimising the adverse effects by local and CC hazards on lake water quality and quantity (Fig. 1).

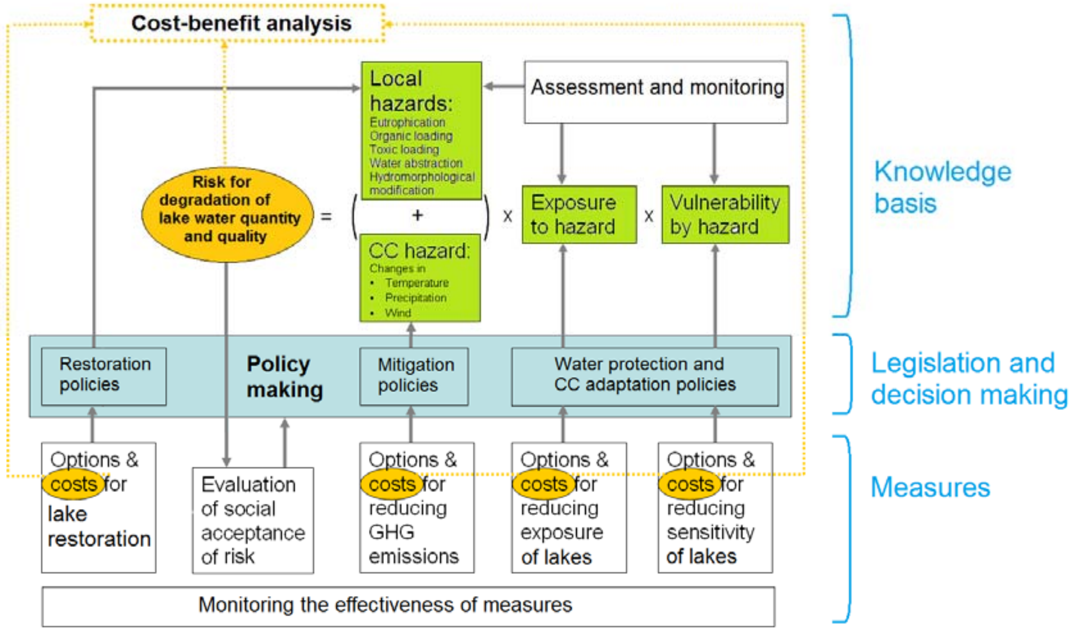

Fig. 1 Policy framework for lake management in changing climate

Mapping of the hazards and collecting information on lakes and their catchments builds up the knowledge basis for lake management. In this scheme **local hazards** are defined as anthropogenic impacts at a water body or river basin scale, such as excessive nutrient loading causing eutrophication, pollution of waters by organic and toxic substances, water abstraction, and a complex of changes commonly referred to as hydromorphological modifications. As opposed to local hazards, **CC hazards** act at regional, continental or global scale and are induced by changes in air temperature, precipitation and wind regimes, including both changes in long-term mean values and extremes. To keep the scheme simple, CC hazard is shown as a box that simply ads to local hazards. In fact, CC can impact on all three steps: occurrence, exposure and vulnerability to local hazards. Local and CC hazards may interact (e.g. changes in precipitation alter the mobility and loading of substances from the catchment) creating multiple pressures to lake ecosystems.

The reaction of lakes to these hazards depends on the **exposure** of lakes to hazards that is mostly a function of location, morphometry and connectivity, and on **vulnerability** of lake by hazards that is a more type specific feature. Different sensitivity of lake types to local hazards has been considered in the type specific approach of the WFD and forms one of its conceptual pillars.

Policy making in lake management refers to the decision making, legislative measures and practical actions taken by governments and water managers with the intention to guarantee sustainable use of services offered by lakes to the citizens and minimise the risk for degradation of their ecosystems. The main risk abatement measures can be divided into **CC mitigation and adaptation** measures if CC hazards are considered, whereas **conservation and restoration** measures are intended mostly for local risk abatement.

CC mitigation policy aims at reducing greenhouse gas (GHG) emissions while adaptation measures should reduce the vulnerability of societies and ecosystems to adverse effects of CC. In respect of water resources and ecological status of water bodies the two approaches are often disconnected that, instead of synergies, can create trade-offs between them.

Conservation policy attempts to preserve and maintain existing lake habitats and biodiversity. Restoration ecology assumes that environmental degradation and population declines are somewhat reversible processes. Therefore, targeted human intervention is used in lake restoration to promote habitat and biodiversity recovery and associated gains. Lake restoration measures are applied mostly to remediate the harm caused by local hazards, such as eutrophication, toxic pollution or hydromorphological modification but potentially they can be useful for CC impact remediation as well.

As all these measures are applied at the expenses of local taxpayers, informing of the society about the hazards and public consultation of all management actions belong as integral parts to environmental policy making. The cost-benefit analysis should compare the cost of the measures with the potential losses caused by the degradation of lake water quantity and quality in a long perspective. Article 9 of the WFD requires implementation of pricing policies that provide an incentive to use water efficiently. Pricing is a powerful awareness-raising tool for consumers and combines environmental with economic benefits. Key research-policy challenges regarding economic analysis include addressing the operationalization of the ecosystem service approach and principles for cost-effectiveness and disproportionality (Martin-Ortega et al., 2011). While continuing enforcement action to ensure compliance with Article 9, the Commission will try to facilitate implementation by developing a guidance document, in the framework of the CIS. The guidance document will focus on the methodology to assess the costs and benefits of water measures supporting cost-effectiveness and further implementation of the concept of payment for ecosystem services. This will help identify water efficiency measures and also implement the polluter pays principle (EC, 2012).

1.2. The state of the art in climate change policy making in Europe regarding lakes

The anticipated effects of the climate change (CC) will add to the already existing cumulative anthropogenic pressures on water bodies (Schindler, 2001) and may multiply the stress by synergistic effects (Root & Schneider, 2006; Baron et al., 2012; Lürling & Senerpont Domis, 2013). Most of the physical, chemical and biological parameters of water bodies used for ecological quality assessment are likely affected (Nõges et al., 2007) although, it may be often hard to disentangle the CC effects on surface waters from the local direct human impact (Ormerod et al., 2010; Battarbee et al., 2012).

As disregarding of CC effects by the Water Framework Directive (WFD; EC, 2000) potentially endangered the designed assessment scheme of ecological status, and achieving of the WFD targets in general, a Strategic Steering Group on Climate Change and Water was created in 2007 under the Common Implementation Strategy (CIS), to produce guidance on how Member States should incorporate consideration of climate variability and change into implementation of the Water Framework Directive. This **guidance document „River basin management in**

a changing climate" (CIS, 2009) describes the policy framework around climate-water issues and is a helpful manual giving advice on

- handling scientific knowledge and uncertainties about CC;
- developing strategies that build adaptive capacity for managing climate risks;
- integrating adaptation within key steps of River Basin Management Planning;
- specific issues relating to flood risk and water scarcity.

In the beginning of each chapter, the CIS Guidance describes guiding principles for adaptation, and relates each to steps in RBMP, flood risk management or drought management. The principles are intentionally broad to be applicable across all Member States regardless of regional variations in potential impacts. These principles are intended to help river basin managers to take well informed decisions that are proportionate and robust, given acknowledged uncertainties in regional CC impacts. Each of the guiding principles are followed by suggested actions to be taken in the coming years in order to apply the principles, and case studies or examples for more details.

With the statement that *"…it is unlikely that within the timeframe of the Water Framework Directive implementation (i.e. up to 2027), the effects of a climate change signal will be adequately distinguishable from other human pressures and natural variability to the extent that extensive changes in status become necessary"*, the CIS Guidance created a balanced, temperate atmosphere around the question how to integrate CC in the framework of water management. As a result, the consideration of CC has been introduced to the 1st River Basin Management Plans (RBMP) of Member States in a largely qualitative way, if at all (Nõges et al., 2010). The decision whether or not to include CC issues to the plan was obviously depending on the availability of information but also on the urgency of the CC related problems involved for each country.

In the 2nd RBMPs, Member States are obliged to incorporate considerations of the impact of climate variability and change on water management.

The recently published **Blueprint to Safeguard Europe's Water Resources** (EC, 2012) recognises the interlinkage of the main causes of negative impacts on water status including CC, land use, economic activities such as energy production, industry, agriculture and tourism; urban development and demographic change. Some of the actions proposed by the Blueprint such as the

⇒ application of the ecological flow concept,

⇒ developing natural water retention measures and buffer strips,

⇒ extending nitrate vulnerable zones,

⇒ improving compliance rates on waste water treatment, and

⇒ reducing water pollution from the use of pesticides

undoubtedly contribute to the achievment of lake management goals, however, the document is not specifically addressing lake issues. **Eutrophication of lakes caused by excessive nutrient levels, which has been a major environmental concern for some decades, is largely ignored by the Blueprint**, although it remains the main risk of failure to meet the WFD requirements for ecological status, as with a few regional exceptions, no clear improvement in nutrient water quality is evident in most agricultural catchments across Europe (EEA, 2010).

Instead the Blueprint identifies the following as the two most widespread pressures on ecological status in the EU:

1. Pressures that originate from **hydromorphological modifications** to water bodies due, for example, to dams for hydropower and navigation or draining land for agriculture; embankments for flood protection (reported by 19 Member States).

2. Pressure that stems from **over-abstraction of water** (reported by 16 Member States).

The strong emphasis of the Blueprint on water quantity issues is obviously reflecting the general concern of many Member States about water scarcity and stress, which, is expected to affect in 2030 about half of EU river basins, but this statement in the Blueprint may mislead or perplex water managers giving **a wrong signal as if the eutrophication problems in the EU were almost solved.**

Within the **REFRESH** project threats on current adaptive management and lake restoration efforts due to climate change were reviewed (**Deliverable 3.12**).

CC adaptation and mitigation measures included in the first river basin management plans were reviewed in REFRESH **deliverable 1.1** and a comprehensive review including more than 450 CC adaptation measures was given in **deliverable 1.2.**

1.3. Purpose of the present guidelines

The main purpose of the present guidelines is to summarize the recent achievements in lake research including the results of the REFRESH Project and based on those formulate some robust guiding principles for lake managers which

- recall some of the basic rules in limnology regarding nutrient limitation, cascading effects in foodwebs, and type-specific differences of lakes,
- put a clear emphasis on needs and cost-effective methods to combat eutrophication as the dominating pressure on lakes in Europe,
- if followed, will unexceptionally lead to win-win solutions.

Similarly to the CIS guidance (CIS, 2009), the guiding principles are followed by suggested actions and case studies or examples for more details. We will highlight the implications of the principles on lake restoration as well as on CC mitigation and adaptation.

The guidelines <u>will not</u>

- repeat the description of the policy framework given in the CIS guidance;
- repeat the CC projections and scenarios for Europe described in the IPCC fourth Assessment Report (IPCC, 2007)
- repeat the guiding principles described in the CIS guidance although some of them have strong linkages with those proposed in the present guidelines;
- describe the specific CC adaptation strategies addressed in the REFRESH Project and the related measures, as a profound overview of this has been given in REFRESH deliverable 1.2.

2. Overview of the guiding principles

This chapter describes guiding principles for adaptation to climate change and relates them to each of the steps in River Basin Management (RBM) under the WFD.

Step of the WFD RBM	**Guiding principles**
Risk assessment	Reducing external phosphorus loading to lakes remains the key for successful lake restoration and meeting water quality targets also in CC conditions
Monitoring and assessment of the status of surface water	Be aware of the dominant cascading effects in your lakes
Objective setting, programme of measures	Consider geographic, and type-specific differences of lakes for selecting appropriate conservation, adaptation and restoration measures
Programme of measures	Avoid tradeoffs between measures

2.1. Combating eutrophication of lakes

Guiding principle 1: **Reducing external nutrient loading to lakes remains the key for successful lake restoration and meeting water quality targets also in CC conditions**

Actions:

⇒ Critical loading for good ecological status (WFD) in lakes has to be lowered in a future warmer climate as natural mechanisms that support zooplankton grazing weaken.

⇒ Increase the sustainability of agriculture by improving nutrient and soil management to reduce loss of nutrients to surface waters,

⇒ reduce loading from point sources,

⇒ where appropriate, re-establish lost wetlands, riparian buffer zones and re-meander channelized streams.

2.1.1 Explanation of the principle: *Reducing N or P or both?*

According to common textbook knowledge, nitrogen is the key mineral nutrient controlling primary production in the ocean, whereas excessive concentrations of phosphorus is the most common cause of eutrophication in freshwater lakes, reservoirs, streams, and headwaters of estuarine systems (Correll, 1998; Schindler & Hecky, 2009). Phosphorus enters lakes mostly as dissolved load from the catchment and a large part of it is retained in lake sediments which may become a persistent P source after the external loads are cut down (Søndergaard et al., 2012).

In addition to riverine loading, nitrogen (N) can enter lake foodchains also from the atmosphere. This process of nitrogen fixation is mediated by N-fixing cyanobacteria and some species of aquatic macrophytes. In this way, the ecosystem can overcome N-limitation. N can control primary production in the ocean, because the lack of nitrogen fixers.

Although in most studies, P is considered the most important limiting nutrient in lakes and the one responsible for eutrophication, several authors advocate for consideration of both N and P (e.g. Conley et al., 2009; Lewis et al., 2011; see also the discussion in Jeppesen et al., 2011a).

Recently a hot discussion started following a statement by Conley et al. (2009) in Science that improvements in the water quality of many freshwater and most coastal marine ecosystems requires reductions in both nitrogen and phosphorus inputs. The authors showed that reducing phosphorus inputs has not reduced eutrophication in some lakes and many estuaries.

In their response Schindler & Hecky (2009) showed that many of the arguments put forward by Conley et al. were based on physiological or short-term indices of nitrogen limitation, which may give spurious results as they do not consider long-term community change likely leading to development of N-fixers in N-depleted systems over time. The fact that reducing phosphorus inputs has not reduced eutrophication in some lakes and many estuaries is caused by high "internal loading" of phosphorus from anoxic sediments (Schelske, 2009; Søndergaard et al., 2012). High concentrations of phosphorus and anoxia in surface sediments are the result of decades of high phosphorus loading causing increased settling and decomposition of organic matter.

The protection of waters against nitrogen (nitrate) pollution from agricultural sources has been one of the prioritised EU policies covered by a special directive since 1991 (EU, 1991). Despite the key importance in controlling water quality and ecological status of inland waters, **phosphorus has got disproportionately less attention** in water policy documents compared to nitrogen. The Blueprint (EC, 2012) acknowledges that eutrophication due to excessive nutrient load remains a major threat to the good status of waters, but to counter these threats, it suggests again extending nitrate vulnerable zones as the first measure.

A strong argument to support P rather than N reduction is its **cost-effectiveness**. According to Schindler & Hecky (2008, 2009a), controlling nitrogen is a costly process. The costs of removing both nutrients may even discourage any treatment in developing countries, particularly in the current economic depression.

Still there are compelling reasons for **controlling agricultural and industrial sources of nitrogen:**

1. In many areas, nitrate and ammonium are now the main pollutants causing damage by acidification and base cation depletion in forests and freshwaters.
2. With climate warming, lakes with a low buffering capacity become increasingly susceptible to ammonia toxicity, which causes summer fish-kills, as both factors (pH and temperature) will increase.
3. As two papers stemming from the REFRESH project (Jeppesen et al., 2011a; Kosten et al., 2012) have shown, in warmer climate, the importance of nitrogen in supporting cyanobacteria blooms will increase also in lakes (see example below).

4. N loading from land to streams is expected to increase in North European temperate lakes due to higher winter rainfall and changes in cropping patterns (Jeppesen et al., 2011a).
5. In southern lakes, N loading from diffuse sources may decrease with lower runoff, however, concentrations in inlets may increase due to enhanced evaporation and water loss (Beklioglu et al., 2007).
6. High N may negatively affect the species richness and abundance of submerged macrophytes in lakes (Moss et al., 2013; James et al., 2005).
7. In some areas, nitrate concentrations in drinking water have increased enough to exceed health standards.

Developing buffer strips along river banks and restoration of riparian areas, wetlands and floodplains to retain nutrients are valuable measures which will decrease loadings of both nutrient to rivers and ultimately to lakes where the most adverse effects of eutrophication usually become evident.

As an answer to the question raised in the title of this chapter, Moss et al. (2013) concluded that effects of N on macrophyte communities provide justification for **control of both nutrients, at least in shallow lakes and estuaries**. Increased N loading reduces plant biodiversity, changes the structure, and is associated with eventual loss, of macrophyte communities. **P control alone may suffice in many deep lakes** where denitrification is low and stratified conditions favour cyanobacterial development.

2.1.2 Implications for CC adaptation

Changes in air temperature, precipitation and wind can affect the retention of phosphorus in the catchment, its transport into the lakes, ant the internal recycling of phosphorus within lakes. Warming exacerbate some symptoms of eutrophication in both cold and warm climates (Beklioglu et al., 2011). An analysis by Jeppesen et al. show that **critical loading of both nutrients has to be lowered in a future warmer climate** for maintaining good ecological status in lakes as natural mechanisms that control phytoplankton biomass (e.g. zooplankton grazing) weaken.

2.1.3 Implications for restoration

In temperate lakes, submerged plants stabilize the clear water state achieved by lake restotration through several physico-chemical buffer mechanisms. As shown by Beklioglu et al. (2011), some of these mechanisms become weak in warm lakes and, as a result, an abrupt shift to a turbid state may occur after surpassing a lake-specific nutrient threshold. Hence, if for cold lakes, restoration methods encompass both bottom-up and top-down controls, then in warmer future bottom-up or nutrient control methods remain the only means for eutrophication control.

Søndergaard et al. (2007) analysing the long-term successes and failures in lake restoration using fish manipulation, found that long-term positive effects (8–10 years) were rare and manipulated lakes often returned to turbid conditions unless fish removal was repeated. They identified insufficient external loading reduction, internal phosphorus loading and absence of stable submerged macrophyte communities to stabilize the clear-water state as the most probable causes for this relapse to earlier conditions.

2.1.4 Examples

Warmer climates boost the dominance of cyanobacteria in shallow lakes

Multiple regression analyses of data from 250 Danish lakes sampled in August showed higher dominance of bloom-forming cyanobacteria and dinophytes **at higher temperatures** (Jeppesen et al., 2011a). The analyses also indicate **high sensitivity to changes in the TN** concentration in lakes when TP is moderately high, as is the case in areas with intensive agriculture.

Similarly, the analysis, by Kosten et al. (2012) based on a study of 143 lakes along a latitudinal transect ranging from subarctic Europe to southern South America, shows that although warmer climates do not result in higher overall phytoplankton biomass, the percentage of the total phytoplankton biovolume attributable to cyanobacteria increases steeply with temperature. The results indicate **a synergistic effect of nutrients and climate** (Fig. 2). The implications are that in a future warmer climate, nutrient concentrations may have to be reduced substantially from present values in many lakes if cyanobacterial dominance is to be controlled.

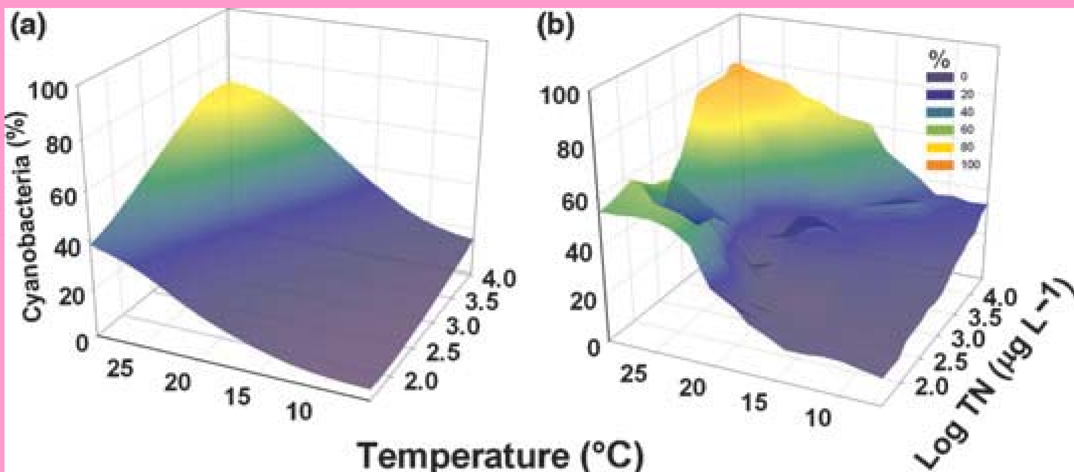

Fig. 2. Percentage of cyanobacterial biovolume in phytoplankton communities as a function of water temperature and total nitrogen concentration in 143 lakes along a climatic gradient in Europe and South America. (a) Combined effects of temperature and nutrients as captured by a logistic regression model, (b) Response surface obtained from interpolation of the raw data using inverse distance weighting (from Kosten et al., 2012).

Warming increases eutrophication symptoms

As expressively shown by a scheme published by Moss et al. (2011), rising nutrient inputs and increasing temperatures tend mutually to intensify eutrophication symptoms (Fig. 3). Cyanobacterial dominance, predominance of floating plants, and perhaps even complete loss of underwater vegetation, occurs at lower nutrient concentrations as temperatures increase. The deoxygenation that may kill fish on still summer nights becomes worse as both nutrients and temperature increase. Moreover, rising temperature increases the nutrient loading by increasing the rate of mineralization in catchment soils and causing greater deoxygenation at the surfaces of lake sediments, so that more nutrients are released in summer. Also often associated with increasing temperature are short, intense storms that increase soil erosion and delivery of nutrients and decreased rainfall in summer or dry seasons. Consequent falling lake levels may concentrate the nutrients already present, expose marginal sediment to mineralization and nutrient release, and increase residence times, favouring bigger crops of slow-growing but persistent phytoplankters like cyanobacteria.

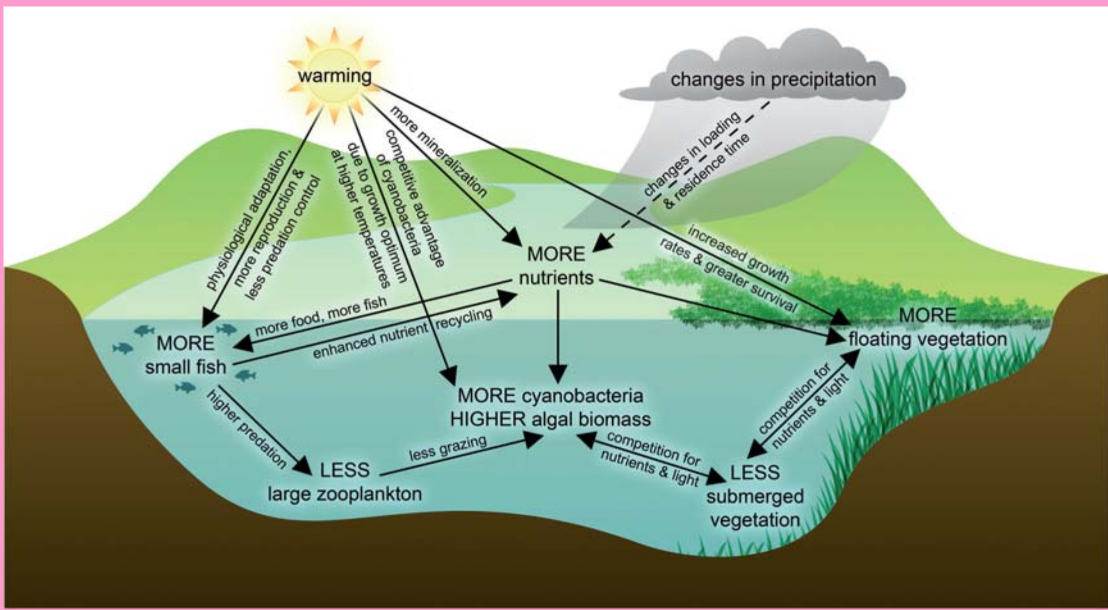

Fig. 3. Some relationships that link climate change and eutrophication symptoms in lakes (from Moss et al., 2011).

2.2 Lake status reflects the functioning of the food-chain

Guiding principle 2: **Be aware of the dominant cascading effects in your lakes**

- ⇒ Rehabilitate zooplankton in lake monitoring schemes
- ⇒ Besides age structure follow changes in fish traits reflecting trophic ecology (feeding types) and size structure.

2.2.1 Explanation of the principle: *Foodweb structure is the key to lake ecosystem*

A simplified aquatic food chain is commonly viewed as composed of four major levels: phytoplankton - zooplankton - zooplanktivore - piscivore, with a strict upward flow of energy, but with simultaneous producer-to-consumer and consumer-to-producer control (Hessen, 1997). Promoted by the trophic cascade theory (Carpenter et al. 1985; Carpenter, 1988; Carpenter & Kitchell, 1988, McQueen et al. 1986) food web manipulation as a tool for eutrophication control has been extensively explored in lake ecosystems (Gulati et al., 1990). Food web regulation in terms of "top-down" and "bottom-up" interactions has also been extensively examined in a vast number of lake ecosystems.

Large majority of observations for lakes confirm the strong effect of planktivorous fish on zooplankton size and species composition. The link from fish to phytoplankton biomass and composition is less clear, although there is evidence that at least large species of daphnids may exert efficient control on most phytoplankton species when released from predation by planktivores. As highlighted by Hessen (1997), one of the things that really stands up after years of study on food web effects related to eutrophication, is in fact the key role of *Daphnia* in lake ecosystems.

A number of enclosure and whole-lake experiments have demonstrated how reduced stocks of planktivorous fish may cause rapid and dramatic decrease in phytoplankton including cyanobacteria.

Understanding the trophic cascade of a lake is the key for interpreting changes in variables characterizing the WFD ecological status, making future predictions and selecting effective restoration measures.

2.2.1.1 ***Rehabilitate zooplankton in lake monitoring schemes***

Following the discovery that changes at the top of the food web can have cascading effects through zooplankton to phytoplankton (Carpenter et al., 2001), and even to nutrient concentrations (Jeppesen et al., 1998) and carbon emission (Cole et al., 2000), zooplankton and sometimes fish were added to several national monitoring programmes and have had a prominent position in lake status assessment, for example, in Austria, Denmark, Finland, the Netherlands and Norway, and several East European countries. As a matter of some surprise to lake ecologists, zooplankton were not included as a biological quality element (BQE) in the WFD ecological status assessment scheme despite their being considered to be an important and integrated component of the pelagic food web. The decision of omitting zooplankton has resulted in the withdrawal of zooplankton from many so-far-solid monitoring programmes. An analysis carried out within the REFRESH Project (Jeppesen et al., 2011b) and using examples from particularly Danish, Estonian, and the UK lakes, showed that zooplankton have a strong indicator value, which cannot be covered by sampling fish and phytoplankton without a very comprehensive and costly effort. When selecting the right metrics, zooplankton are cost-efficient indicators of the trophic state and ecological quality of lakes. Moreover, they are important indicators of the success/failure of measures taken to bring the lakes to at least good ecological status. The authors strongly recommend the EU to include zooplankton as a central BQE in the WFD assessments, and undertake similar regional calibration exercises to obtain relevant and robust metrics also for zooplankton as is being done at present in the cases of fish, phytoplankton, macrophytes and benthic invertebrates.

Case studies provided in the review by Jeppesen et al. (2011b) exemplify

- ⇒ Zooplankton community as indicator of eutrophication
- ⇒ Zooplankton size as indicator of eutrophication and predation
- ⇒ Zooplankton as indicator of change in climate
- ⇒ Zooplankton:phytoplankton ratio as indicator of zooplankton grazing
- ⇒ Cladoceran remains as indicators of structure and function.

The authors conclude that the absence of zooplankton from monitoring recommended under the European Water Framework Directive seems a curious omission for which there is no well argued scientific explanation.

Although the WFD acknowledges the possibility to include zooplankton into operational and investigative monitoring schemes if it can be proven to be the quality element most sensitive to the pressures to which the waterbodies are subjected. However, there is a very high risk that the policy makers and managers follow the minimum requirement policy. Therefore, the authors strongly appeal to the relevant EU authorities to **include zooplankton as a BQE during the first revision of the WFD.**

2.2.1.2 *Besides age structure follow changes in fish traits reflecting trophic ecology (feeding types) and size structure*

The structure and functioning of cold temperate shallow lakes are expected to become more similar to those of (sub)tropical shallow lakes, as the temperature increase will enhance the top-down controls of omnivorous and benthivorous fish as well as the nutrient cycling.

Jeppesen et al. (2012a) found profound changes in fish assemblage composition, size and age structure during recent decades and a shift towards higher dominance of eurythermal species and small individuals occurring in high abundance. Studies into winter ecology of shallow lakes (Sørensen et al., 2011) showed that climate warming, supposedly leading to reduced winter mortality and dominance of small fish, may enhance the risk of turbid state conditions in nutrient-enriched shallow lakes, not only during the summer season, but also during winter.

As shown by Jeppesen et al. (2012a), the shift towards dominance of eurythermal species has occurred despite an overall reduction in nutrient loading that should have benefited the fish species typically inhabiting cold-water low-nutrient lakes and larger-sized individuals. The response of fish to the warming in recent decades has been surprisingly strong, making them ideal sentinels for detecting and documenting climate-induced modifications of freshwater ecosystems.

Mesocosm experiments carried out in Turkey (Bucak et al., 2012) indicated that despite a strong negative effect of fish predation on water clarity, a reduction in water level during summer may help maintain the growth of macrophytes in Mediterranean eutrophic shallow lakes and, in this way, at least partly counteract the effect of enhanced top-down control of fish.

2.2.2 Implications for CC adaptation

A recent development of the food-chain theory by Hansson et al. (2012) emphasized the cascading effect and showed that food-chain length alters community responses to global change in aquatic systems. On the basis of modelling, monitoring and experimental data, they demonstrated that the top trophic level, and every second level below, will benefit from climate change, whereas the levels in between will suffer. In contrast to other taxa, cyanobacteria benefit from a higher temperature and humic content irrespective of the food-chain composition. By mechanistically merging present food-chain theory with large-scale environmental and climate changes, the authors provided a powerful framework for predicting and understanding future aquatic ecosystems and their provision of ecosystem services and water resources.

Jeppesen et al., 2011a warn that temperature increase will enhance the top-down controls of omnivorous and benthivorous fish on zooplankton as well as the nutrient cycling that through cascading effect lead to higher abundance of phytoplankton. Cyanobacteria are further stimulated by higher temperatures and more stable and longer stratification. In addition, macrophytes will be a less efficient refuge for zooplankton further reducing the grazing capacity of zooplankton on phytoplankton.

2.2.3 Implications for restoration

Lake restoration, and in particular fish removal in shallow eutrophic lakes, has been widely used in Denmark and the Netherlands, where it has had marked effects on lake water quality in many lakes. Søndergaard et al. (2007) analysing the long-term successes and failures in lake restoration using fish manipulation, found that in more than half of the biomanipulation projects, Secchi depth increased and chlorophyll *a* decreased to less than 50% within the first few years. In some of the shallow lakes, total phosphorus and total nitrogen levels decreased considerably, indicating an increased retention or loss by denitrification. The strongest effects seemed to be obtained 4–6 years after the start of fish removal. The often seen relapse to earlier conditions was in most cases caused by insufficient external loading reduction, internal phosphorus loading and absence of stable submerged macrophyte communities to stabilize the clear-water state.

2.2.4 Examples

Zooplankton size as indicator of eutrophication and predation

Zooplankton size changes markedly with eutrophication and the related increase in fish predation. In a study of Danish lakes (Jeppesen et al., 2000), the mean individual body weight of cladocerans decreased substantially with increasing TP (Fig. 4). This reduction reflected not only the low relative abundance of large *Daphnia* spp., but also a reduction in the body weight of the remaining *Daphnia* spp. and of small cladocerans.

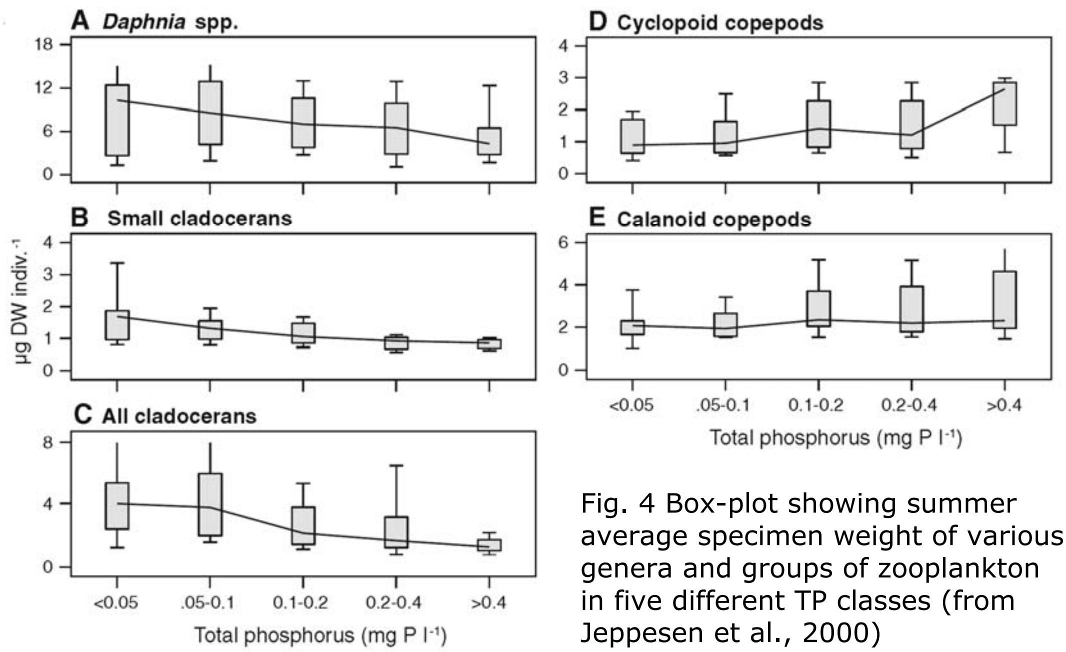

Fig. 4 Box-plot showing summer average specimen weight of various genera and groups of zooplankton in five different TP classes (from Jeppesen et al., 2000)

The U.S. fish traits database

In the USA the need for integrated and widely accessible sources of fish species traits data to facilitate studies of ecology, conservation, and management has motivated development of traits databases for various taxa (http://www.fishtraits.info/FishTraits/). Funding to develop FishTraits and the website came from the U.S. Geological Survey (USGS) Aquatic Gap Analysis Program and the Virginia Tech Department of Fisheries and Wildlife Sciences.

The database contains information on four major categories of traits: (1) trophic ecology, (2) body size and reproductive ecology (life history), (3) habitat associations, and (4) salinity and temperature tolerances. Information on geographic distribution and conservation status is also included. Developing a similar database for Europe would facilitate large-scale analysis involving many fish species and/or traits that is especially valuable for tracking long term trends.

Impacts of climate warming on lake fish assemblages in Europe

Fish play a key role in the trophic dynamics of lakes. With climate warming, complex changes in fish assemblage structure may be expected owing to direct effects on temperature and indirect effects operating through eutrophication, water level changes, stratification and salinisation. A review of published and new long-term fish data series from 24 European lakes carried out within the REFRESH project (Jeppesen et al., 2012a) found profound changes in fish assemblage composition, size and age structure during recent decades and a shift towards higher dominance of eurythermal species. The shift has occurred despite an overall reduction in nutrient loading that should have benefited the fish species typically inhabiting cold-water low-nutrient lakes and larger-sized individuals. The cold-stenothermic Arctic charr (*Salvelinus alpinus*) 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. Vendace (*Coregonus albula*), whitefish (*Coregonus* spp.) and smelt (*Osmerus eperlanus*) show a different response depending on lake depth and latitude. Perch (*Perca fluviatilis*) was apparently stimulated in the north, with stronger year classes in warm years, but its abundance has declined in the southern Lake Maggiore. Where introduced, roach (*Rutilus rutilus*) now seems to take advantage of the higher temperature after years of low populations. Eurythermal species such as common bream (*Abramis brama*), pike-perch (*Sander lucioperca*) and shad (*Alosa agone*) are on the increase. The response of fish to the warming in recent decades has been surprisingly strong, making them ideal sentinels for detecting and documenting climate-induced modifications of freshwater ecosystems.

Changes in shallow lake food chains

According to a conceptual model (Jeppesen et al., 2012b) the trophic structure in temperate meso-eutrophic will shift towards higher degree of small fish dominance that will decrease zooplankton size and abundance (Fig. 5). Weaker control of phytoplankton growth by zooplankton implies that 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.

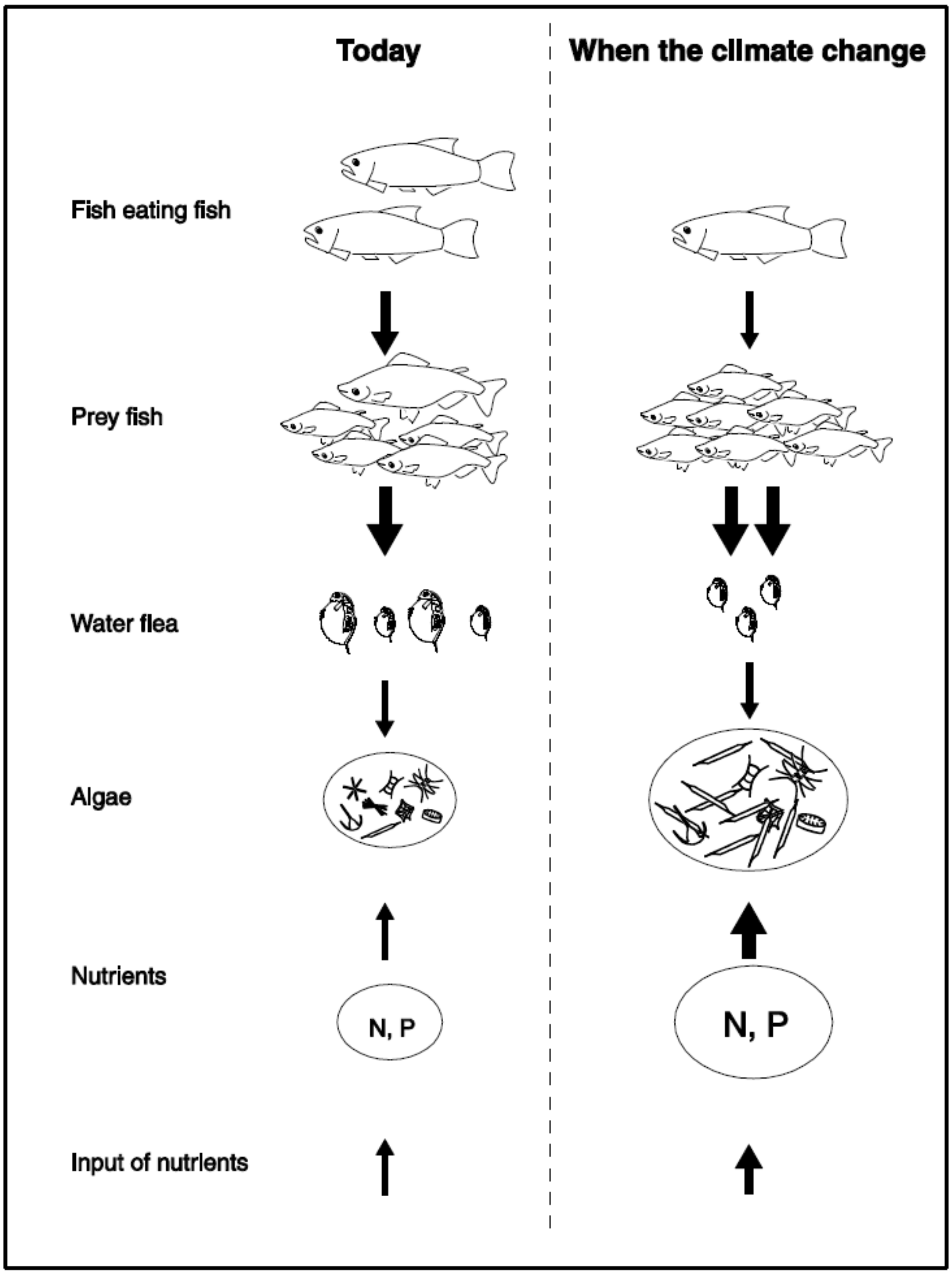

Fig. 5. Conceptual model showing trophic structure in meso-eutrophic Danish lakes and the suggested changes in a climate change perspective (from Jeppesen et al. 2012b)

2.3 Sensitivity and vulnerability of lakes

Guiding principle 3: **Consider geographic, and type-specific differences of lakes for selecting appropriate conservation, adaptation and restoration measures**

Actions:

⇒ Consider latitudinal and altitudinal differences in pressures and lake reactions;

⇒ Consider differences in pressures and reactions of lakes depending on their morphometry and retention time;

⇒ Consider different buffering capacity of lakes depending on alkalinity and humic content.

2.3.2 Explanation of the principle

The terms 'sensitivity of lakes' to anthropogenic pressures and 'vulnerability of lakes' are often used as synonyms although their exact meaning is vaguely defined. Obviously because of this vagueness, little effort has been made to operationalize the terms within an ecosystem context that would make their measurement and empirical testing straightforward (Angeler & Johnson, 2012). There are three types of reactions occurring in lakes in response to climatic or other pressures for which the term 'sensitivity' is commonly used:

1. **Fast and sensitive reaction** of some ecosystem parameters to the pressure levels, e.g. climatic forcing, expressed in strong correlation with the signal. In this sense, the thermal conditions of shallow lakes where the water temperature rapidly follows changes in air temperature can be considered more sensitive to climatic forcing compared to deep and stratified lakes.

2. **"Memorizing" of the climate signal by lakes**. Lakes with a longer "memory" can be considered more sensitive as the longer exposure time to the pressure due to the memory may have cumulative effects on lake ecosystems. In this respect, deep and stratified lakes in which winter conditions determine the completeness of mixing and the impact can remain traceable in the hypolimnion temperature for years (Livingstone, 1997; Gerten & Adrian, 2001) during which it affects chemical reactions and biological activity, can be considered more sensitive than shallow lakes.

3. **Regime switching in lakes resulting from pressure**. According to the concept of alternative stable states (Scheffer et al., 2001), lake ecosystems exposed to gradual changes in climate, nutrient loading, habitat fragmentation etc. may change abruptly to a contrasting alternative stable state when its resilience to the pressure is exceeded. Return to the initial conditions is questionable and often involves hysteresis meaning that the forward and backward switching points are located at different levels of the forcing factor. In this sense, lakes which are located closer to the threshold levels for breaking their resilience can be considered more sensitive compared to lakes which have enough tolerance to withstand the actual pressure level. Based on paleolimnological analyses Fritz (1996) postulated that lakes located in extreme habitats or near an ecotone or climatic boundary will respond most sensitively to climate change. In this respect, lakes located in the Alpine/Perialpine region, on the Atlantic coast, and at high latitudes are likely to respond most sensitively to the climatic changes (Arvola et al., 2010).

In whatever context the sensitivity of lakes is considered, for management issues it is fundamental to recognize the individuality of lakes and differences occurring in their resistance both to pressure factors and to restoration efforts.

Over recent years, abundant evidence has been accumulated on differences in lake sensitivity to environmental change depending on geographic location, lake morphometry and catchment chemistry, which could be useful for selecting appropriate adaptation and restoration measures.

Latitudinal and altitudinal differences in pressures and lake reactions

The time-series analysed within the EU FP-5 project CLIME (George, 2010) showed that lakes located in different parts of Europe responded in different ways to climatic forcing. In Northern Europe, the most important effects were those associated with the change in the freeze-thaw dates of the lakes and the resulting seasonal shift in their hydrological cycles. In Britain and Ireland, the lakes were more sensitive to the flushing effects of heavy rain and the stabilizing effects of calm, anticyclonic weather. The responses observed in Central Europe were more diverse since this region included some shallow as well as a number of very deep lakes. Type-specific and ecoregional approach adopted in the Water Framework Directive provides ideal means of addressing these issues, but more needs to be done to quantify the climatic sensitivity of lakes at a regional as well as European level.

In a REFRESH paper (D 3.19) dedicated to lake vulnerability, Angeler & Johnson, basing on a literature review, showed that ecosystems at high latitudes, especially freshwaters including subarctic and arctic lakes, streams, ponds and wetlands, are particularly sensitive to change due to the combined impacts of modified ice cover regimes, increasing water temperature, thawing permafrost, and changes to hydrological processes and water balance. The impact of global change on northern freshwaters is likely twice as much as the global average and may be having dramatic effects on the abiotic template and biodiversity of freshwater ecosystems.

The lakes in the perialpine region are known to be very sensitive to short-term changes in the weather (Psenner, 2003; Thompson et al., 2005). Here, the topography and the steep orography enhance the water cycle, and result in flooding, debris flows, avalanches, vertical plant migration etc. The Alps also form a barrier to the mass movement of air and are responsible for the sharp climatic divide between Atlantic, Continental and Mediterranean influences (Dokulil et al., 2010).

Lakes in different ecoregions vary by the complexity of the ecosystem structure. For example, the number of fish species was significantly higher in lakes in the Alpine than in the Mediterranean ecoregion (Volta et al., 2011). Among Alpine lakes, the number of fish species increased significantly with lake volume whilst decreased with altitude. In the Mediterranean lakes, none of the selected parameters was significant.

In ice-covered lakes the most sensitive climatic indicators are usually the timing and duration of ice cover. A study of 196 Swedish lakes by Weyhenmeyer et al. (2004) demonstrated the high degree of coherence in the timing of ice-off among lakes in northern Europe but showed also that the timing of ice-off responds much more sensitively to air temperature in warmer regions than colder regions. In southern Sweden the sensitivity of the timing of ice-off to air temperature was ~14 days per 1◦C, but only ~4 days per 1◦C in northern Sweden. That contradicts to some extent to the common view of increasing climate sensitivity of lakes with increasing latitude. Although the physical factors influencing ice cover and its sensitivity to climate change are by now quite well understood, the impact of these changes on the ecological status of lakes is less clear (Leppäranta et al. 2010).

Nordic catchments can be highly sensitive to small variations in precipitation and temperature. Long-term lake water temperature records could be very useful and provide a costeffective way of detecting subtle patterns of change, quantifying the sensitivity of lakes to the changing climate, and providing the information required for the future management of this important resource (Arvola et al., 2010). Nevertheless, this kind of information is still surprisingly rare.

Geographic differences affect strongly also the mobility of nutrients. In northern catchments the increase in total nitrogen loss from the catchment was primarily governed by the climatic factors that had the most pronounced effect on hydrological processes (Moore et al. 2010) whereas the effects of storm runoff on the export of phosphorus can be very sensitive to levels of soil saturation and soil moisture (Pierson et al. 2010).

Differences in pressures and reactions of lakes depending on morphometry and retention time

Analyses based on the data of 1,337 lakes included in the European Environment Agency (EEA) database (Nõges 2009) revealed strong relationships between lake and catchment morphometry, water chemistry, and water quality indices over a large scale of European lakes. The study showed that in Europe, the lakes towards North are larger but shallower and have smaller catchment areas than the southern lakes; lakes at higher altitudes are deeper and smaller and have smaller catchment areas than the lowland lakes. Larger lakes have generally larger catchment areas and bigger volumes, and they are deeper than smaller lakes, but the relative depth decreases with increasing surface area. The lakes at higher latitudes have lower alkalinity, pH and conductivity, and also lower concentrations of nitrogen and phosphorus while the concentration of organic matter is higher. In the lakes at higher altitudes, the concentration of organic matter and nutrient contents are lower and water is more transparent than in lowland lakes. In larger lakes with larger catchment area, the alkalinity, pH, conductivity and the concentrations of nutrients and organic matter are generally higher than in smaller lakes with smaller catchments. If the lake is deep and/or its residence time is long, the water is more transparent and the concentrations of chlorophyll a, organic matter and nutrients are lower than in shallower lakes with shorter residence times. The larger the catchment area is with respect to lake depth, area and volume, the lower is the water transparency and the higher are the concentrations of the nutrients, organic matter and chlorophyll as well as pH, alkalinity and conductivity. The links between lake water quality and morphometry become stronger towards large and shallow lakes. Along the decreasing gradients of latitude, altitude and relative depth, the present phosphorus concentration and its deviation from the reference concentration increases. All these differences should be considered in the assessment schemes and strongly diversify the reaction of lakes to forcing factors.

The response of any lake to a change in the weather is critically dependent on its size, its heat storage capacity and its exposure to wind mixing. Large lakes integrate the variations in the weather on longer time-scales whilst small lakes are more sensitive to short-lived events. The sensitivity of large lake ecosystems to changes in water level increases with decreasing mean depth. A similar change in a shallow lake’s water level compared to a deep lake affects much stronger lake volume related parameters such as water retention time, heat storage or dilution capacity. Moreover, several environmental key factors (e.g. light intensity and wind-induced bottom shear stress) attenuate exponentially with increasing depth and so does their impact on benthic habitats which in shallow lakes may be by orders of magnitude stronger than in deep lakes following a similar absolute change in water levels.

Livingstone (2010) reviewing the effect of morphometric differences of lakes on their sensitivity showed that deep lakes tend to exhibit a more persistent physical response to climatic forcing than shallow lakes. The deep waters of lakes which are chemically stratified also tend to respond less sensitively to climatic forcing than the deep waters of lakes which are not. Large lakes – especially those with a convoluted coastline and multiple basins, such as the Swedish lakes Mälaren, Vättern and Vänern – can be exposed simultaneously to several different local weather regimes, to which they exhibit an internally heterogeneous response.

Lake morphometry influences the distribution of heat with larger wind exposed surface contributing to the dominance of convective wind mixing among the mechanisms of heat transfer (Nõges et al. 2011a). Analysis carried out in the UK (George et al. 2010a) showed that the shallower lakes were more sensitive to the observed variations in the number of northerly days since they lose proportionately more of the heat stored during the summer. In temperate regions, the highest surface water temperatures in winter are recorded in deep lakes that retain heat and the lowest in shallower lakes that loose more heat to the atmosphere (Dokulil et al. 2010).

The effect of the projected increases in the winter rainfall on the residence time of lakes depends on lake size, catchment area and geographic location. George et al. (2010a) showed that the most sensitive sites in Britain and Ireland will be small lakes located in the 'wet' west and the least sensitive sites large lakes situated in the dry'east.

Pettersson et al. (2010) demonstrated the importance of physical characteristics of individual lakes for controlling P cycling in lakes. Lakes with a short retention time were typically more sensitive than lakes with a long retention time whilst the internal dynamics of phosphorus was very different in isothermal and thermally stratified lakes. The authors concluded that the anticipated impacts of changes in the climate on the supply of phosphorus and the growth of phytoplankton should be thus of growing concern to water managers.

Different buffering capacity of lakes depending on alkalinity and humic content

The alkalinity of water refers to its capacity to neutralize acids or to resist changes in pH. Alkalinity is a measure of the concentrations of three basic ions: carbonates (CO_3^{2-}), bicarbonates (HCO_3^-), and hydroxides (OH^-) in water. Rain water is naturally slightly acidic (pH=5.6), and the acidity increases when pollutants (SO_2, NO_x) dissolve in water causing acid rain. Lakes that have a limestone bed have a natural buffering ability to neutralize acid rain. Liming is one of the most cost-effective methods of slowing the effects of acidification, restoring acidic waters, and enhancing the abundance and diversity of aquatic life. It also reduces the toxic effects of metals, especially aluminum, copper, cadmium, lead, nickel, and zinc, which can threaten fish, other aquatic life, and human health.

Hard water lakes with high carbonate alkalinity have a lower sensitivity not only to acidification but also to eutrophication and its consequences. In hard water lakes phosphorus can be bound with calcium and removed from nutrient cycle. Utilization of bicarbonate by charophytes, for example, is accompanied by precipitation of calcite during periods of intensive photosynthesis, favoring immobilization of P by binding in the crystal structure or sorption on sedimenting mineral particles (Kufel & Kufel, 2002). Smaller pH dynamics in hard water lakes avoids ammonia formation. Ammonia toxicity is one of the main causes of summer fish-kills in lakes (Farnsworth-Lee & Baker, 2000).

Dissolved humic substances, with high concentrations of organic acids contribute to the naturally low pH observed in humic waters. In contrast, they also markedly contribute to buffering capacity, which can mitigate the effects of acidification (Kortelainen, 1999).

Jennings et al. (2010) showed that high concentrations of humic substances can also impact on nutrient availability. In the boreal region, highly coloured lakes may stratify in the spring immediately after icemelt, preventing the supply of hypolimnetic nutrients to the epilimnion. Metal binding properties of humic substances and the interaction of humus-iron complexes with phosphate can further reduce the concentration of dissolved nutrients, but also act as nutrient reservoirs during periods of low availability.

2.3.3 Implications for CC adaptation

With climate change the sensitivity of lakes to anthropogenic pressures and, hence, their vulnerability will likely increase. Climate change will further diversify the reactions of different lake types.

The percentage of unionized ammonia (NH_3) in water is a function pH and temperature. Peaks of pH in summer are mainly caused by phytoplankton blooms the extent and intensity of which is expected to increase. With climate warming, lakes with a low buffering capacity become increasingly susceptible to ammonia toxicity as both factors (pH and temperature) will increase (Table 1).

Table 1. Percentage of total ammonia that is unionized at various temperatures and pH (Swann, 1997)

pH	12°C	17°C	20°C	24°C	28°C	32°C
7	0.2	0.3	0.4	0.5	0.7	1
7.4	0.5	0.7	1	1.3	1.7	2.4
7.8	1.4	1.8	2.5	3.2	4.2	5.7
8.2	3.3	4.5	5.9	7.7	11	13.2
8.6	7.9	10.6	13.7	17.3	21.8	27.7
9	17.8	22.9	28.5	34.4	41.2	49
9.2	35.2	42.7	50	56.9	63.8	70.8
9.6	57.7	65.2	71.5	76.8	81.6	85.9
10	68.4	74.8	79.9	84	87.5	90.6

2.3.4 Implications for restoration

Although the problems pointing at the need for lake restoration may be common for many lake types (algal blooms, low transparency, siltation, fish-kills), the restoration procedure starting from setting objectives and targeting changes in lake water quality to managing the watershed and selecting restoration techniques, should be type specific and firmly based on knowledge of the individual systems. Restoration targets (i.e., the good ecological status) would also need to be evaluated periodically, accepting that some changes (as loss or immigration of some species, or permanently elevated biomass levels of phytoplankton being fed by internal sources of nutrients) have become permanent (Nõges et al. 2009).

Examples

Zooplankton behaviour defines the bio-manipulation success

Different behavioural responses of planktonic animals to their main predators, fish, have been reported from shallow lakes. In north temperate lakes, large-bodied zooplankton may seek refuge from predation among macrophytes, whereas in subtropical lakes, avoidance of macrophytes has been observed (Nihan Tavsanoglu et al. 2012). The prevalent behaviour probably depends on the characteristics of the fish community, which in Mediterranean lakes is typically dispersed in both the open water zone and in the littoral, as in temperate lakes, and is dominated by small size classes, as in subtropical lakes. Different behavioural responses of zooplankton to fish implies that fish manipulation and/or re-establishment of submerged macrophyte stands will, as lake restoration measures, have completely different effect in north temperate and Mediterranean lakes.

Assessing vulnerability of lakes to environmental change:
A case study using subarctic and hemiboreal lakes (D 3.19, SLU) assessed the responses of littoral invertebrate communities to changing abiotic conditions in Sweden. The study focused on temporal scale effects. Two patterns of temporal change within the invertebrate communities were identified that were consistent across the lakes. The first pattern was one of monotonic change associated with changing abiotic lake conditions. The second was one of showing fluctuation patterns largely unrelated to gradual environmental change. An understanding of scale-specific processes provides managers with a realistic assessment of vulnerabilities and the relative resilience of ecosystems to environmental change.

Opposite reactions of oligo- and eutrophic lakes to increased amount of precipitation

A study (Nõges et al. 2011b) carried out in two adjacent stratified lakes in North Italy with different trophic status (oligo-mesotrophic and eutrophic) observed increased nutrient (N, P, Si) loading during a rainy winter and spring and increased nutrient concentrations, whereas the N/P ratio decreased in both lakes. The weakened Si limitation enabled an increase of diatom biovolumes in spring in both lakes leading to P depletion in the epilimnion. Summer chlorophyll a concentration increased in the oligo-mesotrophic lake, but dropped markedly in the eutrophic lake where the series of commonly occurring cyanobacteria blooms was interrupted. The projected increase of winter precipitation in southern Europe is likely to increase the nutrient loadings to lakes and will generally contribute to their eutrophication. The authors suggested that the impact is proportional to the runoff/in-lake concentration ratio of nutrients rather than to the retention time, and is more pronounced in lakes with lower trophy.

Different residence time of lakes diversifies the climatic impact on the internal recycling of phosphorus

A modelling approach to simulate the effects of changes in the weather on the internal cycling of phosphorus (Blenckner et al. 2010) was carried out in three lakes in central Sweden. The lakes were morphometrically very different and had residence times that ranged from one month to seven years. The monthly discharge and phosphorus loads used in the simulations were the averages calculated for the last ten years. In all cases, it was assumed that the external loading remained the same since the main aim of the study was to compare the climatic sensitivity of the individual lakes. The results showed that the projected changes in the climate have a very different effect on the internal recycling of phosphorus in the three lakes. In the two lakes with the shorter residence time (Galten and Ekoln), there was no significant change in the internal flux of phosphorus under 'warm world' conditions. The average annual concentration of total phosphorus projected for the lakes for 2071–2100 was also very similar to that recorded between 1961 and 1990. In contrast, in the lake with the longest residence time (Erken), there was a marked intensification of the phosphorus cycle as more phosphate was released from the sediment during the extended period of thermal stratification. Here, the average concentration of total phosphorus in the epilimnion almost doubled under the conditions projected by the 'warm world' A2 emission scenario. Short-lived increases in the amount of phosphorus released by the sediment were also predicted for Ekoln during periods of strong stratification but the relatively short residence time of that lake ensured that these episodes had little effect on the mean phosphorus concentration. Taken together, these results imply that the internal recycling of phosphorus will increase as the world becomes warmer. The magnitude of this response will, however, vary from lake to lake and may be masked in lakes with very long hydrological residence times. In addition to the residence time, the projected change in the stability of Lake Erken also lead to a higher mineralization rate, elevated bacterial activity, a decreased concentration of oxygen in and higher rates of phosphate diffusion from the sediment into the water column. As a consequence, the biomass of phytoplankton in the lake increases and leads to a further intensification of the phosphorus cycle.

2.4 Composing the programme of measures

Guiding principle 4: **Avoid tradeoffs between measures**

⇒ Avoid CC adaptation and mitigation measures which endanger water resources and the ecological status of lakes

⇒ Avoid energy-intensive adaptation measures based on continuous use of fossil fuels and thus contradict to CC mitigation

⇒ Avoid energy-intensive restoration measures if supporting natural processes to sustain the achieved results are not developed.

2.4.1 Explanation of the principle

Adaptation planning is constrained by uncertainty about evolving climatic and non-climatic pressures, by difficulties in predicting species- and ecosystem-level responses to these forces, and by the plasticity of management goals (Wilby et al., 2010). Adaptation measures will have greatest acceptance when they are robust and deliver multiple benefits, including, but not limited to, the amelioration of climate impacts.

Dessai & Hulme define robust such decisions that work well (that achieve their goals) even with the inclusion of various uncertainties. In other words, robust decisions are decisions that are insensitive to uncertainties known at the time. The authors emphasize urgent need for clearly robust adaptation decisions but show also that robust adaptation to CC is not easy because

1. robustness to climate change uncertainties usually means higher costs (and therefore lower aspirations);
2. different impact sectors will be sensitive to different uncertainties in climate change assessments;
3. the robustness of adaptation strategies to climate change uncertainties will likely depend on the pressure exerted on the decision process by drivers other than climate. This implies that local context is crucial and that it is difficult to generalise any lessons from a single case study to wider adaptation planning.

In contrast to large uncertainty in decision making regarding CC adaptation and water, one thing has become pretty clear and this is **the issue of potential conflicts and trade-offs between management measures.** Uncoordinated sectoral responses can be ineffective or even counterproductive, because a response in one sector can increase the vulnerability of another sector and/or reduce the effectiveness of its adaptation responses. Abundant evidence of measures becoming contraproductive or antagonistic enables their identification and **avoiding such situations becomes a robust measure of adaptive management** on its own.

The United Nations Guidance on Water and Adaptation to Climate Change (U.N. 2009) emphasizes that climate change adaptation should be integrated into existing policy development, in planning, programmes and budgeting, across a broad range of economic sectors – a process generally called "mainstreaming". And mitigation measures should be considered in the light of their consequences for adaptation options, and vice versa.

The IPCC defines **CC mitigation** as: "An anthropogenic intervention to reduce the sources or enhance the sinks of greenhouse gases". Several measures in water management can contribute to CC mitigation. Reduction of CO_2 atmospheric loading can be achieved by biological, chemical and technological options through either reducing or sequestering emissions. Replacement of fossil fuels with renewable energy sources is one of CC mitigation measures. Hydropower continues to serve as an important alternative energy source to fossil fuel and nuclear power in many parts of the world and is the cheapest way to generate electricity today. In addition to clean electricity, impoundment hydropower creates reservoirs that offer a variety of recreational opportunities, notably fishing, swimming, and boating. Most hydropower installations are required to provide some public access to the reservoir to allow the public to take advantage of these opportunities.

In the long term, mitigation measures can reduce the magnitude of the impacts of global warming on water resources, in turn reducing adaptation needs. However, mitigation measures can also have considerable negative side effects for adaptation, such as increased water requirements for bio-energy crops, if projects are not sustainably located, designed and managed (U.N. 2009). Dams and water reservoirs can emit additional GHGs undermining their performance as CC mitigation options. The rise of public awareness of environmental issues of the early 1970s narrowed public acceptance of hydropower as energy source and reduced significantly its role in the energy matrix in numerous countries (Sternberg, 2008).

Climate adaptation refers to the ability of a system

- to adjust to CC (including climate variability and extremes)
- to moderate potential damage,
- to take advantage of opportunities, or
- to cope with the consequences.

The IPCC defines adaptation as the "adjustment in natural or human systems to a new or changing environment". Adaptation to CC refers to adjustment in natural or human systems in response to actual or expected climatic stimuli or their effects, which moderates harm or exploits beneficial opportunities. Various types of adaptation can be distinguished, including anticipatory and reactive adaptation, private and public adaptation, and autonomous and planned adaptation."

Adaptation measures can also have negative effects for mitigation. This is often because many adaptation measures increase energy use, which, if no renewable sources are used, will increase greenhouse gas (GHG) emissions and hence mitigation requirements. Examples are desalinization, irrigation, and energy-intensive construction of flood protection infrastructure (U.N. 2009).

Using lakes for hydro-electric generation, for water storage for irrigation schemes and for flood control has altered lakeshores through artificially manipulating lake levels. The littoral area of lakes is a region of dynamic physical processes, and high biodiversity and productivity, serving as a crucial habitat for terrestrial and aquatic plants and many invertebrates, fish and birds during all or part of their life cycles. This is often the first zone of a lake to suffer the effects of human activity.

Also the CIS guidance (CIS, 2009) admits that some adaptation measures may actually be counterproductive for the water environment, and these should be avoided as much as possible. In case there is no chance to avoid those measures, they need to meet the conditions set in Article 4.7 of the WFD on new modifications which may still lead to the conclusion that better environmental options exist for the planned measures.

Lake restoration is a broad term used for different techniques aiming to bring a lake back to or closer to anthropogenically undisturbed conditions. Usually, lake restoration refers to methods used inside the lake, but sometimes it also refers to measures taken outside the lake such as reduction of the external nutrient loading by improved waste water treatment. Eutrophication of lakes due to high nutrient loading (especially phosphorus) has been the paramount environmental problem for lakes worldwide for the past approximately 50 years (Carpenter et al., 1999) and thus the vast majority of lake restoration projects have been conducted to combat eutrophication. Many approaches have been used during the past 20–30 years to improve lake water quality but the results from the various lake restoration initiatives are diverse and the long-term effects are not well described. (Søndergaard et al., 2007). The removal of zooplanktivorous and benthivorous fish has been by far the most common internal lake measure.

Lake restoration methods can be divided into preventive and ameliorative methods with the latter being generally more expensive and less successful. The endurance of effects of some ameliorative methods, such as aeration of water or biomass harvesting does not exceed much the period of their application requiring that the treatments be constantly running or periodically repeated. In the long run, such methods cannot be considered sustainable. Also sediment removal, if not properly designed, may become a counterproductive measure as large amounts of phosphorus may release from the disturbed sediments to the water or return with surface runoff from the sediment deposition areas.

2.4.2 Examples

Eutrophication of lakes contributes to carbon sequestration

Lakes not only metabolise carbon that is received from the catchment but also take up considerable amounts of CO_2 from the atmosphere in the course of photosynthesis of phytoplankton and macrophytes. The difference between CO_2 photosynthetic uptake (gross primary production, GPP) and CO_2 respiratory release (community respiration, CR), can be used to define whether a lake is predominantly **autotrophic (= a carbon sink)** or **heterotrophic (= a carbon source to the atmosphere)**. Lakes become net heterotrophic only because they receive large allochthonous inputs of organic C from the catchment which is respired in the lake. A review by Andersson & Sobek (2006) showed that switching from net autotrophy to net heterotrophy occurred at dissolved organic carbon (DOC) concentrations higher than 4–6 mg l^{-1}. Brown-coloured lakes with increased humic matter content, often associated with heavily forested or peaty catchments, display stronger net heterotrophy than open, clear-water lakes (del Giorgio et al., 1999; Sobek et al., 2005). Oligo- and mesotrophic lakes with low total phosphorus (TP) content, mean seasonal chlorophyll *a* (Chl *a*) concentration below 20 mg m^{-2} and GPP less than 140 mmol C m^{-2} day^{-1}, as a rule, have a net heterotrophic C balance (Cole et al., 2000; Hanson et al., 2003). Net autotrophic systems produce significantly more organic material than they degrade. The excess organic material may either be exported to adjacent systems and/or accumulated within the sediment (Staehr et al., 2010). **Eutrophic lakes with high TP and Chl *a* and low DOC concentrations tend to be autotrophic, whereas lakes with low TP and high DOC tend to be heterotrophic** (Hanson et al., 2003). Still it would be short-sighted to promote eutrophication of lakes as a CC mitigation measure.

The CC mitigating effect of eutrophication that intensifies photosynthetic CO_2 uptake from the atmosphere is likely outweighed by the enhanced methane (CH_4) release from eutrophic lake sediments. A recent study (Bastviken et al., 2011) indicated that the release of methane from freshwater areas changes the net absorption of greenhouse gases by natural continental environments, such as forests, by at least 25 percent. In the global change scenarios (IPCC, 2007), methane has a high global warming potential: 72 times that of carbon dioxide over 20 years, and 25 times over 100 years. Past analyses of carbon and greenhouse gas exchanges on continents failed to account for the methane gas that is naturally released from lakes and running water. According to the study, small methane emissions from the surfaces of water bodies occur continuously but greater emissions occur suddenly and with irregular timing. Open water methane emission can be predicted from variables such as lake area, water depth, concentrations of TP, DOC, and methane, and the anoxic lake volume fraction (Bastviken et al., 2004). Shallow, epilimnetic sediments were major contributors of CH_4 to the atmosphere while 51–80% of the CH_4 produced in deep sediments was oxidized in the water column (Bastviken et al., 2008).

Valuing hydrological alteration in multi-objective water resources management

The management of water through the impoundment of rivers by dams and reservoirs is necessary to support key human activities such as hydropower production, agriculture and flood risk mitigation. Advances in multi-objective optimization techniques and ever growing computing power make it possible to design reservoir operating policies that represent Pareto-optimal tradeoffs between multiple interests. On the one hand, such optimization methods can enhance performances of commonly targeted objectives (such as hydropower production or water supply), on the other hand they risk strongly penalizing all the interests not directly (i.e. mathematically) included in the optimization algorithm. The alteration of the downstream hydrological regime is a well established cause of ecological degradation and its evaluation and rehabilitation is commonly required by recent legislation (as the Water Framework Directive in Europe). However, it is rarely embedded in reservoir optimization routines and, even when explicitly considered, the criteria adopted for its evaluation are doubted and not commonly trusted, undermining the possibility of real implementation of environmentally friendly policies. The main challenges in defining and assessing hydrological alterations are: how to define a reference state (referencing); how to define criteria upon which to build mathematical indicators of alteration (measuring); and finally how to aggregate the indicators in a single evaluation index (valuing) that can serve as objective function in the optimization problem. A paper by Bizzi et al. (2012) propose a method to define **an aggregate index of hydrological alteration**. Aggregation by weighting must be based on a-posteriori analysis of indicator values. The index can be used to design minimum environmental flow (MEF) and reservoir operation. Time-varying MEF can reduce hydrological alteration at almost no cost for irrigation.

References

Andersson, E. & S. Sobek, 2006. Comparison of a mass balance and an ecosystem model approach when evaluating the carbon cycling in a lake ecosystem. Ambio 35: 476–483.

Angeler, D.G. & R.K. Johnson, 2012. Assessing vulnerability of lakes to environmental change: A case study using subarctic and hemiboreal lakes. REFRESH Deliverable 3.19: Report on methodology for vulnerability assessment.

Arvola, L., D. G. George, D. M. Livingstone, M. Järvinen, T. Blenckner, M. T. Dokulil, E. Jennings, C. N. Aonghusa, P. Nõges, T. Nõges, G. A. Weyhenmeyer, 2010. The impact of changing climate on the thermal characteristics of lakes. Ch. 6 in D. G. George (ed.), The Impact of Climate Change on European Lakes, Aquatic Ecology Series 4, Springer, p. 85-101.

Baron, J.S., E. K. Hall, B. T. Nolan, J. C. Finlay, E. S. Bernhardt, J. A. Harrison, F. Chan & E. W. Boyer, 2012. The interactive effects of excess reactive nitrogen and climate change on aquatic ecosystems and water resources of the United States. Biogeochemistry. DOI 10.1007/s10533-012-9788-y

Bastviken D, Cole J, Pace M, Tranvik L (2004) Methane emissions from lakes. Dependence of lake characteristics, two regional assessments, and a global estimate. Global Biogeochemical Cycles, 18, GB4009, DOI:10,1029/2004GB002238

Bastviken, D., L. J. Tranvik, J. A. Downing, P. M. Crill, & A. Enrich-Prast. 2011. Freshwater Methane Emissions Offset the Continental Carbon Sink. Science 331 (6013): 50 DOI: 10.1126/science.1196808

Battarbee, R.W., N.J. Anderson, H. Bennion & G.L. Simpson, 2012.Combining limnological and palaeolimnological data to disentangle the effects of nutrient pollution and climate change on lake ecosystems: problems and potential. Freshwater Biology 57(10): 2091–2106.

Beklioglu M., M. Meerfhoff , M. Søndergaard & E. Jeppesen, 2011. Eutrophication and Restoration of Shallow Lakes from a cold Temperate to a warm Mediterranean and a (Sub)Tropical climate. in AA Ansari-S Singh-GR Lanza-W Rast (Editors). Eutrophication: Causes, Consequences and Control. Springer. pp 91-108.

Beklioglu, M., S. Romo, I. Kagalou, X. Quintana & E. Becares, 2007. State of the art in the functioning of shallow Mediterranean lakes: workshop conclusions. Hydrobiologia 584: 317–326.

Blenckner, T., A. Elliott, H. Markensten, C. Pers & S. Thackeray, 2010. Modeling the Effects of Climate Change on the Seasonal Dynamics of Phytoplankton. In D. G. George (ed.), The Impact of Climate Change on European Lakes, Aquatic Ecology Series 4, Springer: 275-291.

Bizzi, S., F. Pianosi & R. Soncini-Sessa 2012. Valuing hydrological alteration in multi-objective water resources management. Journal of Hydrology 11: 472–473.

Bucak, T., Saraoglu, E., Levi, E.E., Nihan Tavasanoglu, Ü., İdİl Çakiroglu, A., Jeppesen, E. & M. Beklioglu, 2012. The influence of water level on macrophyte growth and trophic interactions in eutrophic Mediterranean shallow lakes: a mesocosm experiment with and without fish. Freshwater Biology 57: 1631–1642.

Carpenter, S. R. 1988. Complex interactions in lake communities. - Springer-Verlag, New York.

Carpenter, S. R. & J. F. Kitchell, 1988. Consumer control of lake productivity. - BioScience 38: 764-769.

Carpenter, S. R., J. F. Kitchell & J.R. Hodgson, 1985. Cascading trophic interactions and lake productivity. - BioScience 35: 634-639.

Carpenter S., D. Ludwig & W. Brock, 1999. Management of eutrophication for lakes subject to potentially irreversible change. Ecological Applications 9(3), 751-771.

Carpenter, S. R., J. J. Cole, J. R. Hodgson, J. F. Kitchell, M. L. Pace, D. Bade, K. L. Cottingham, T. E. Essington, J. N. Houser & D. E. Schindler, 2001. Trophic cascades, nutrients, and lake productivity: whole-lake experiments. Ecological Monographs 71: 163–186.

CIS, 2009. River basin management in a changing climate. Common Implementation Strategy for the Water Framework Directive (2000/60/EC), Guidance document 24. European Commission, 141 pp.

Cole, J. J., M. L. Pace, S. R. Carpenter & J. F. Kitchell, 2000. Persistence of net heterotrophy in lakes during nutrient addition and food web manipulations. Limnology and Oceanography 45: 1718–1730.

Conley, D.J., H.W. Paerl, R.W. Howarth, D.F. Boesch, S.P. Seitzinger, K.E. Havens, C. Lancelot & G.E. Likens, 2009. Controlling eutrophication: Nitrogen and phosphorus. Science 323: 1014-1015.

Correll, D.L., 1998. The Role of Phosphorus in the Eutrophication of Receiving Waters: A Review. Journal of Environmental Quality 27 No. 2, p. 261-266.

del Giorgio, Paul A.; Cole, Jonathan J.; Caraco, Nina F.; Peters, Robert H., 1999: Linking planktonic biomass and metabolism to net gas fluxes in northern temperate lakes. Ecology (Washington D C). June; 804: 1422-1431

Dokulil, M.T., K. Teubner, A. Jagsch, U. Nickus, R. Adrian, D. Straile, T. Jankowski, A. Herzig & J. Padisak, 2010. The Impact of Climate Change on Lakes in Central Europe. In D. G. George (ed.), The Impact of Climate Change on European Lakes, Aquatic Ecology Series 4, Springer, p. 387-409.

EEA 2010. Freshwater eutrophication assessment. Background Report for EEA European Environment State and Outlook Report 2010. ETC Water Technical Report 2/2010.

EEA, 2012. European waters — assessment of status and pressures. EEA Report No 8/2012. Office for Official Publications of the European Union. Luxembourg: 100 pp.

EC, 1991. Council Directive 91/676/EEC concerning the protection of waters against pollution caused by nitrates from agricultural sources. OJ L375, 31.12.91

EC. 2000. Directive 2000/60/ EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for Community action in the field of water policy. OJ L 327, 22.12.2000

EC, 2012. COMMUNICATION FROM THE COMMISSION TO THE EUROPEAN PARLIAMENT, THE COUNCIL, THE EUROPEAN ECONOMIC AND SOCIAL

COMMITTEE AND THE COMMITTEE OF THE REGIONS. A Blueprint to Safeguard Europe's Water Resources. Brussels, 14.11.2012. COM(2012) 673 final.

Farnsworth-Lee, L. & L. Baker, 2000. Conceptual Model of Aquatic Plant Decay and Ammonia Toxicity for Shallow Lakes. Journal of Environmental Engineering 126: 199–207.

Fritz, S. C., 1996. Paleolimnological records of climate change in North America, Limnology and Oceanography 41: 882–889.

George, G., 2010. The impact of climate change on European lakes. In D. G. George (ed.), The Impact of Climate Change on European Lakes, Aquatic Ecology Series 4, Springer: 1-13.

George, G., E. Jennings & N. Allott, 2010a. The Impact of Climate Change on Lakes in Britain and Ireland. In D. G. George (ed.), The Impact of Climate Change on European Lakes, Aquatic Ecology Series 4, Springer: 359-386.

George, G. U. Nickus, M. T. Dokulil & T. Blenckner 2010b. The Influence of Changes in the Atmospheric Circulation on the Surface Temperature of Lakes. In D. G. George (ed.), The Impact of Climate Change on European Lakes, Aquatic Ecology Series 4, Springer: 293-310.

Gerten, D. & R. Adrian, 2001. Differences in the persistency of the North Atlantic Oscillation signal among lakes, Limnology and Oceanography 46, 448–455.

Gulati, R. D., Lammens, E. H. R. R., Meijer, M.-L. and Van Donk, E. 1990. Biomanipulation tool for water management. Kluwer, Dordrecht.

Hanson, P. C., D. L. Bade. & S. R. Carpenter, 2003. Lake metabolism: relationships with dissolved organic carbon and phosphorus. Limnology and Oceanography 48: 1112–1119.

Hansson, L.-A., A. Nicolle, W. Granéli, P. Hallgren, E. Kritzberg, A. Persson, J. Björk, A. Nilsson & C. Brönmark. 2012. Food-chain length alters community responses to global change in aquatic systems. Nature Climate Change. DOI: 10.1038/NCLIMATE1689.

Hessen, D.O., 1997. Stoichiometry in Food Webs: Lotka Revisited. Oikos 79: 195-200.

IPCC. 2007. Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change (Solomon, S. et al., eds). Cambridge Univ. Press, Cambridge, U. K.

James, C., J. Fisher, V. Russell, S. Collings & B. Moss, 2005. Nitrate availability and hydrophyte species richness in shallow lakes. Freshwater Biology 50: 1049–1063.

Jennings, E., M. Järvinen, N. Allott, L. Arvola, K. Moore, P. Naden, C.N. Aonghusa, T. Nõges & G.A.Weyhenmeyer, 2010. Impacts of Climate on the Flux of Dissolved Organic Carbon from Catchments. In D. G. George (ed.), The Impact of Climate Change on European Lakes, Aquatic Ecology Series 4, Springer: 199-220.

Jeppesen, E., J. P. Jensen, J. Windolf, T. Lauridsen, M. Søndergaard, K. Sandby & P. Hald Møller, 1998. Changes in nitrogen retention in shallow eutrophic lakes following a decline in density of cyprinids. Archiv für Hydrobiologie 142: 129–152.

Jeppesen, E., J. P. Jensen, M. Søndergaard, T. L. Lauridsen & F. Landkildehus, 2000. Trophic structure, species richness and biodiversity in Danish Lakes: changes along a phosphorus gradient. Freshwater Biology 45: 201–218.

Jeppesen E, B.Kronvang, J. E. Olesen, M. Søndergaard, C.C. Hoffmann, J., H.E. Andersen, T.L. Lauridsen , L.Liboriussen, M. Meerhoff, M.Beklioglu, A.Özen, 2011a. Climate change effect on nitrogen loading from catchment in Europe: implications for nitrogen retention and ecological state of lakes and adaptations. Hydrobiologia 663:1-21.

Jeppesen, E., Nõges, P., Davidson ,T..A., Haberman, J., Nõges, T., Blank, K., Lauridsen, T.L., Søndergaard, M., Sayer, C., Laugaste, R., Johansson, L.S., Bjerring, R., Amsinck, S.L. 2011b. Zooplankton as indicators in lakes: a scientific-based plea for including zooplankton in the ecological quality assessment of lakes according to the European Water Framework Directive (WFD). Hydrobiologia 676: 279–297.

Jeppesen E., T. Mehner, I. J. Winfield, K. Kangur, J. Sarvala, D. Gerdeaux, M. Rask, H. J. Malmquist, K. Holmgren, P. Volta, S. Romo, R. Eckmann, A. Sandström, S. Blanco, A. Kangur, H. R. Stabo, M. Meerhoff, A.-M. Ventelä, M. Søndergaard, T. L. Lauridsen, 2012a. Impacts of climate warming on lake fish assemblages: evidence from 24 European long-term data series. Hydrobiologia 694: 1-39.

Jeppesen E., M. Søndergaard, T.L. Lauridsen, L. Liboriussen, R. Bjerring, L.S. Johanssen, F.Landkildehus, B. Kronvang;H.E. Andersen, D. Trolle, T. Jørgensen, 2012b. Recent climate induced changes in freshwaters in Denmark. In Goldman CR, M. Kumagari & R.D. Robarts Global impacts of climate change on inland waters. Wiley pub. (in press).

Kortelainen, P., 1999. Acidity and buffer capacity. In Keskitalo and P. Eloranta (eds.), Limnology of humic waters, Backhuys Publishers, Leiden, The Netherlands, pp. 95–115.

Kosten, S.; Huszar, V.L.M.; Bécares, E.; Costa, L.S.; van Donk, E.; Hansson, L.; Jeppesen, E.; Kruk, C.; Lacerot, G.; Mazzeo, N.; De Meester, L.; Moss, B.; Lürling, M.; Nõges, T.; Romo, S.; Scheffer, M., 2012. Warmer climates boost cyanobacterial dominance in shallow lakes. Global Change Biology, Volume 18, Issue 1, p.118–126

Kufel, L. & I. Kufel, 2002. Chara beds acting as nutrient sinks in shallow lakes—a review. Aquatic Botany 72: 249–260.

Leppäranta, M., 2010. Modelling the Formation and Decay of Lake Ice. In D. G. George (ed.), The Impact of Climate Change on European Lakes, Aquatic Ecology Series 4, Springer: 39-59.

Lewis, W.M. Jr., W.A. Wurtsbaugh & H.W. Paerl, 2011.Rationale for Control of Anthropogenic Nitrogen and Phosphorus to Reduce Eutrophication of Inland Waters. Environmental Science and Technology 45, 10300–10305.

Lindeman, R. L. 1942. The trophic-dynamic aspect of ecology. Ecology 23:399-418.

Livingstone, D. M., 1997. An example of the simultaneous occurrence of climate-driven "sawtooth" deep-water warming/cooling episodes in several Swiss lakes, Verhandlungen der Internationalen Vereinigung für Theoretische und Angewandte Limnologie 26(2), 822–826.

Livingstone, D.M., R. Adrian, T. Blenckner, G. George & G. A.Weyhenmeyer, 2010. Lake Ice Phenology. In D. G. George (ed.), The Impact of Climate Change on European Lakes, Aquatic Ecology Series 4, Springer: 51-61.

Lürling, M. & L.N. de Senerpont Domis, 2013. Predictability of plankton communities in an unpredictable world. Freshwater Biology. DOI: 10.1111/fwb.12092

Martin-Ortega, J., G. Giannoccaro & J. Berbel, 2011. Environmental and resource costs under water scarcity conditions: An estimation in the context of the European Water Framework Directive. Water Resources Management 25:1615–1633.

McQueen, D. J., J. R. Post, and E. L. Mills. 1986. Trophic relationships in freshwater pelagic ecosystems. Canadian Journal of Fisheries and Aquatic Sciences 43: 1571-81

Moore, K., E. Jennings, N. Allott, L. May, M. Järvinen, L. Arvola, T. Tamm, A. Järvet, T. Nõges, D. Pierson & E. Schneiderman, 2010. Modelling the Effects of Climate Change on the Supply of Inorganic Nitrogen. In D. G. George (ed.), The Impact of Climate Change on European Lakes, Aquatic Ecology Series 4, Springer: 179-197.

Moss, B., S. Kosten, M. Meerhoff, R. W. Battarbee, E. Jeppesen, N. Mazzeo, K. Havens, G. Lacerot, Z. Liu, L. De Meester, H. Paerl and M. Scheffer, 2011 Allied attack: climate change and eutrophication. Inland Waters: 101-105.

Moss,B., E. Jeppesen, M. Søndergaard, T. L. Lauridsen & Z. Liu, 2013. Nitrogen, macrophytes, shallow lakes and nutrient limitation: resolution of a current controversy? Hydrobiologia 710:3–21

Nihan Tavsanoglu, Ü., A. Idil Çakiroglu, S. Erdogan, M. Meerhoff, E. Jeppesen & M. Beklioglu, 2012. Sediments, not plants, offer the preferred refuge for Daphnia against fish predation in Mediterranean shallow lakes: an experimental demonstration. Freshwater Biology 57: 795–802.

Nõges P., W. Van de Bund, A. C. Cardoso, & A. S. Heiskanen. 2007. Impact of climatic variability on parameters used in typology and ecological quality assessment of surface waters - implications on the Water Framework Directive. Hydrobiologia 584: 373–379.

Nõges, P. and A. De Jager, 2009. Clustering large lakes and reservoirs in Europe on the basis of their morphometry, catchment, and climatic conditions. Verh. Internat. Verein. Limnol. 30

Nõges, P., W. van de Bund, A. C. Cardoso, A. G. Solimini and A.-S. Heiskanen, 2009. Assessment of the ecological status of European surface waters: a work in progress. Hydrobiologia 633: 197–211.

Nõges, T., 2009. Relationships between morphometry, geographic location and water quality parameters of European lakes. Hydrobiologia 633: 33–43.

Nõges, T., P. Nõges & A. C. Cardoso, 2010. Climate change adaptation and mitigation strategies already in practice based on the 1st River Basin Management Plans of the EU Member States. Scientific and Technical Research Series EUR 24663 EN, Joint Research Centre. Publications Office of the European Union. Luxembourg, 127 pp. [REFRESH deliverable 1.1]

Nõges P., Nõges T., Ghiani M., Paracchini B., Pinto Grande, J., Sena, F. 2011a. Morphometry and trophic state modify the thermal response of lakes to meteorological forcing. Hydrobiologia 667:241-254.

Nõges P., Nõges T., Ghiani M., Sena, F., Fresner R., Friedl M., Mildner J. 2011b. Increased nutrient loading and rapid changes in phytoplankton expected with climate change in stratified South European lakes: sensitivity of lakes with different trophic state and catchment properties. Hydrobiologia. 667:255-270. DOI 10.1007/s10750-011-0649-9.

Ormerod, S. J., M. Dobson, A. G. Hildrew & C. R. Townsend Multiple stressors in freshwater ecosystems. Freshwater Biology 55:1-4.

Pettersson, K., D. G. George, P. Nõges, T. Nõges and T. Blenckner, 2010. The impact of the changing climate on the supply and re-cycling of phosphorus. Ch. 8 in D. G. George (ed.), The Impact of Climate Change on European Lakes, Aquatic Ecology Series 4, Springer, p. 121-137.

Pierson, D., L. Arvola, N. Allott, M. Järvinen, E. Jennings, L. May, K. Moore & E. Schneiderman, 2010. Modeling the Effects of Climate Change on the Supply of Phosphate-Phosphorus. In D. G. George (ed.), The Impact of Climate Change on European Lakes, Aquatic Ecology Series 4, Springer: 139-159.

Psenner, R., 2003. Alpine lakes: extreme ecosystems under the pressure of global change, EAWAG news 55, 12–14.

Root, T.L. & S.H. Schneider, 2006. Conservation and Climate Change: the Challenges Ahead. Conservation Biology 20(3): 706–708.

Scheffer, M., S. Carpenter, J. A. Foley, C. Folke & B. Walkerk, 2001. Catastrophic shifts in ecosystems. Nature 413: 591-596.

Schelske, C.L., 2009. Eutrophication: Focus on phosphorus. Science 324: 722

Schindler, D.W., 2001. The cumulative effects of climate warming and other human stresses on Canadian freshwaters in the new millennium. Canadian Journal of Fisheries and Aquatic Sciences, 2001, 58(1): 18-29.

Schindler, D.W. & R.E. Hecky, 2008. Reply to Howarth and Paerl: Is control of both nitrogen and phosphorus necessary? PNAS 105(49): E104.

Schindler, D.W. & R.E. Hecky, 2009. Eutrophication: More nitrogen data needed. Science 324: 721-722.

Sobek, S., L. J. Tranvik & J. J. Cole, 2005. Temperature independence of carbon dioxide supersaturation in global lakes. Global Biogeochemical Cycles 19: 1–10.

Søndergaard, M., E. Jeppesen, T.L. Lauridsen, C. Skov, E.H. van Nes, R. Roijackers, E. Lammens and R. Pprtielje, 2007. Lake restoration: successes, failures and long-term effects. Journal of Applied Ecology 44: 1095–1105.

Søndergaard, M.; Larsen, S.; Jørgensen, T.; Jeppesen, E., 2012. Using chlorophyll a and cyanobacteria in the ecological classification of lakes. Ecological Indicators 11: 1403 – 1412.

Sørensen, T., G. Mulderij, M. Søndergaard, T. L. Lauridsen, L. Liboriussen, S. Brucet & E. Jeppesen, 2011. Winter ecology of shallow lakes: strongest effect of fish on water clarity at high nutrient levels. Hydrobiologia 664:147–162.

Staehr, P. A., K. Sand-Jensen, A. L. Raun, B. Nielsson & J. Kidmose, 2010. Drivers of metabolism and net heterotrophy in contrasting lakes. Limnology and Oceanography 55: 817-830.

Sternberg, R., 2008. Hydropower: Dimensions of social and environmental coexistence. Renewable and Sustainable Energy Reviews 12: 1588–1621.

Swann, L. 1997. A Fish Farmer's Guide to Understanding Water Quality. Fact Sheet AS-503. Aquaculture Extension. Illinois-Indiana Sea Grant Program. https://www.extension.purdue.edu/extmedia/AS/AS-503.pdf

Thompson, R., C. Kamenik & R. Schmidt, 2005. Ultra-sensitive Alpine lakes and climate change, Journal Limnology 64: 139–152.

U.N. 2009. Guidance on Water and Adaptation to Climate Change. Economic Commission for Europe Convention on the Protection and Use of Transboundary Watercourses and International Lakes. United Nations, New York and Geneva, 144 p.

Volta, P., A. Oggioni, R. Bettinetti & E. Jeppesen, 2011. Assessing lake typologies and indicator fish species for Italian natural lakes using past fish richness and assemblages. Hydrobiologia 671: 227-240.

Weyhenmeyer, G. A., M. Meili & D.M. Livingstone, 2004. Nonlinear temperature response of lake ice breakup, Geophysical Research Letters 31(7): L07203.

Wilby, R.L., H. Orr, G. Watts, R.W. Battarbee, P.M. Berry, R. Chadd, S.J. Dugdale, M.J. Dunbar, J.A. Elliott, C. Extence, D.M. Hannah, N. Holmes, A.C. Johnson, B. Knights, N.J. Milner, S.J. Ormerod, D. Solomon, R. Timlett, P.J. Whitehead, P.J. Wood, 2010. Evidence needed to manage freshwater ecosystems in a changing climate: Turning adaptation principles into practice. Science of the Total Environment 408(19):4150-64.