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REVIEW PAPER

Climate change effects on nitrogen loading from cultivated catchments in Europe: implications for nitrogen retention, ecological state of lakes and adaptation

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Abstract Climate change might have profound effects on the nitrogen (N) dynamics in the cultivated landscape as well as on N transport in streams and the eutrophication of lakes. 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. Scenario (IPCC, A2) analyses using a number of models of various complexity for Danish streams and lakes suggest an increase in runoff and N transport on an annual basis (higher during winter and typically lower during summer) in streams, a slight increase in N concentrations in streams despite higher losses in riparian wetlands, higher absolute retention of N in lakes (but not as percentage of loading), but only minor changes in lake water concentrations. However, when taking into account also a predicted higher temperature there is a risk of higher frequency and abundance of potentially toxic cyanobacteria in lakes and they may stay longer during the season. Somewhat higher risk of loss of submerged macrophytes at increased N and phosphorus (P) loading and a shift to dominance of small-sized fish preying upon the key grazers on phytoplankton may also enhance the risk of lake shifts from clear to turbid in a warmer North European temperate climate. However, it must be emphasised that the prediction of N transport and thus effects is uncertain as the prediction of regional precipitation and changes in land-use is uncertain. By contrast, N loading is expected to decline in warm temperate and arid climates. However, in warm arid lakes much higher N concentrations are currently observed despite reduced external loading. This is

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due to increased evapotranspiration leading to higher nutrient concentrations in the remaining water, but may also reflect a low-oxygen induced reduction of nitrification. Therefore, the critical N as well as P loading for good ecological state in lakes likely has to be lower in a future warmer climate in both north temperate and Mediterranean lakes. To obtain this objective, adaptation measures are required. In both climate zones the obvious methods are to change agricultural practices for reducing the loss of nutrients to surface waters, to improve sewage treatment and to reduce the storm-water nutrient runoff. In north temperate zones adaptations may also include re-establishment of artificial and natural wetlands, introduction of riparian buffer zones and re-meandering of channelised streams, which may all have a large impact on, not least, the N loading of lakes. In the arid zone, also restrictions on human use of water are urgently needed, not least on the quantity of water used for irrigation purposes.

Keywords Climate change · Nitrogen loading · Lakes · Ecological state · Modelling · Streams · Subtropics · Temperate · Warm arid

Introduction

Intensive farming and livestock production characterise large parts of Western Europe including Denmark (European Environment Agency, 1999). This has led to significant nitrogen (N) losses from agriculture resulting in eutrophication of surface water bodies—with detrimental environmental effects (European Environment Agency, 1995; Kronvang et al., 2005). To reduce the N loading to groundwater and marine areas, this has led to a number of national and international initiatives. In Europe, the European Union has introduced two major directives to mitigate the effects of N emissions to the environment. First, The Nitrates Directive (1991/696/EC) was introduced in 1991, aiming to reduce nitrate water pollution in the Nitrate Vulnerable Zones (NVZs) of Europe. Secondly, The Water Framework Directive (2000/60/EC) was introduced in 2000 with the aim to preserve groundwater resources and all surface waters in a state with only minor anthropogenic interference. As a result, N transport has declined in many streams and rivers in Europe, although an upward trend has been traced in some rivers, primarily in Eastern Europe, due to the recent intensification of agriculture (Table 1).

In Denmark, since 1985, seven National Action Plans have been implemented to reduce the N discharge from point sources and N losses from agriculture (Kronvang et al., 2008). The efforts include regulations on point source discharges from waste water treatment plants, area-related measures (e.g. re-establishment of wetlands, afforestation and use of catch crops) and nutrient-related measures (e.g. mandatory fertilizer plans and improved utilization of N in manure). Since 1985, the loading of N from point sources has been reduced by 74% (1989–2003), the field N surplus by 31% (1990–2003), the model-calculated N leaching from the root zone on agricultural land by 33% (1989–2002), and the total nitrogen (TN) concentrations and loads in 86 streams draining smaller agricultural catchments by, respectively, 29 and 32% (1989–2004) (Kronvang et al., 2008) (Fig. 1).

Major changes in the loading of N to freshwaters can be expected due to global warming, with consequent impacts on the ecological state and water

Table 1 Percentage of rivers in different European countries showing a downward trend, no trend or an upward trend in nitrate-nitrogen concentrations during the period 1992–2002 (Kronvang et al., 2008)

Country (number of rivers analysed)	Trend in nitrate-N concentrations in rivers		
	Downward trend (%)	No trend (%)	Upward trend (%)
Slovenia (10)	10	60	30
Finland (42)	14	69	17
Lithuania (42)	17	55	28
Spain (115)	21	58	21
France (225)	24	52	25
UK (173)	31	55	14
Poland (68)	31	35	34
Bulgaria (43)	40	44	16
Norway (153)	40	54	6
Hungary (60)	40	35	25
Sweden (55)	44	40	16
Austria (239)	44	51	5
Slovakia (26)	46	35	19
Latvia (29)	48	41	10
Germany (148)	73	26	1
Czech R. (65)	74	26	0

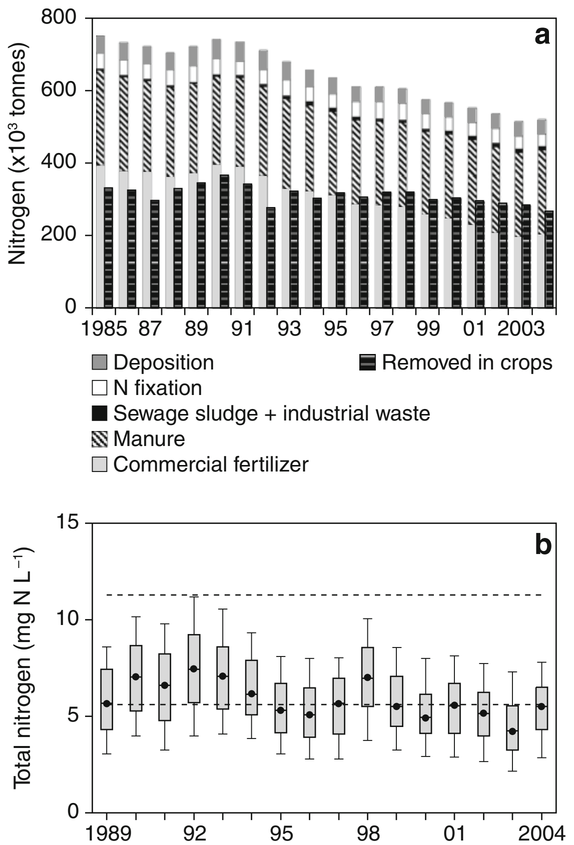

Fig. 1 Nitrogen balances in Danish agriculture during the period 1985–2003 (**a**) (Andersen et al., 2006a), Box-Cox plot for the annual total nitrogen concentrations in the 86 arable catchments studied during the period 1989–2004 (**b**). The 50 mg NO_3 l^{-1} and 25 mg NO_3 l^{-1} thresholds stipulated by the EC Nitrates Directive are depicted (from Kronvang et al., 2008)

quality of surface waters and a potentially negative effect on the measures taken to improve the ecological state of freshwaters. Warming itself affects N dynamics in terrestrial and aquatic ecosystems (Rustad et al., 2001; Weyhenmeyer et al., 2007; Weyhenmeyer & Jeppesen, 2010), and GCM model scenarios have shown a spatial variability in predictions of future precipitation, with reduced rainfall in the arid zones and increased rainfall at higher latitudes and in parts of the tropics (IPCC, 2007). The changes in temperature and precipitation would eventually entail changes in agricultural land use and management, including changes in soil cultivation and in the rates and timing of fertilization (Howden et al., 2007). These changes will, in turn, have both direct and indirect cascading influences on the N cycling, affecting the aquatic environment. Here, we discuss the potential impact of global warming on runoff and transport of N to lakes in Europe, focusing on cultivated catchments, on how such changes may affect the ecological state of lakes and on how we can find ways to counteract the likely negative effects of climate change. We first focus on the cultivated catchment, then on N runoff in rivers and N retention in wetlands, followed by retention of N (loss) and N effects on lakes. Finally, we discuss potential adaptations and mitigations. Examples included are mainly from north temperate coastal Denmark and southern arid Turkey. In a recent paper we did a similar exercise for phosphorus (Jeppesen et al., 2009).

Agriculture and nitrogen in a future warmer climate

Biophysical processes of agroecosystems are strongly affected by environmental conditions. The projected increase in greenhouse gases will affect agroecosystems either directly (primarily by increasing photosynthesis at higher CO_2 (Kimball et al., 2002)) or indirectly via effects on climate (e.g. temperature and rainfall affecting several aspects of ecosystem functioning (Olesen & Bindi, 2002)). The exact responses depend on the sensitivity of the particular ecosystem and on the relative changes in the controlling factors. These responses have consequences for growth and development of agricultural crops and therefore also for their management. All this leads to changes in carbon and N flows and transformations in the agroecosystems, and thus also to changes in the N loss pathways from agroecosystems, and to either lower or higher runoff and leaching of N from agricultural systems to aquatic environments.

Several factors related to climate change affect the N cycling in agroecosystems. An increasing CO_2 concentration leads to higher CO_2 assimilation with concomitant increases in plant C/N ratios (Soussana & Lüscher, 2007), higher rhizodeposition (Bazot et al., 2008) and reduced crop transpiration (Kruijt et al., 2008), influencing plant N uptake and soil C and N dynamics (Sowerby et al., 2005). In many environments this leads to more N efficient crops, which reduces the risk of N leaching. However, elevated CO_2 concentrations also enhance biological

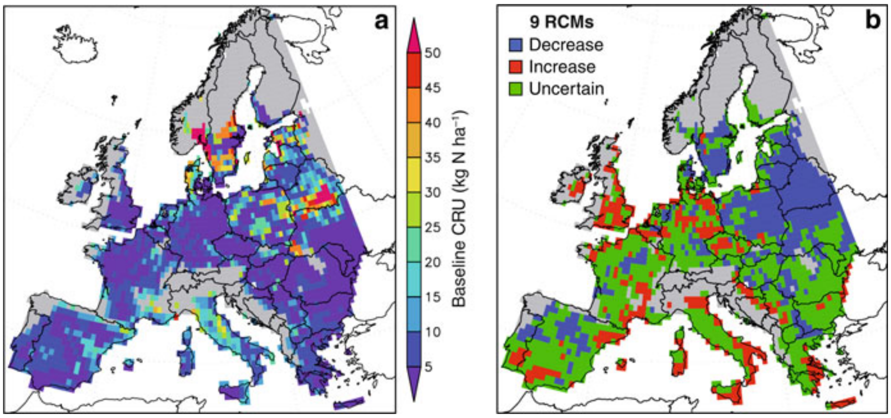

Fig. 2 Estimated nitrate leaching at optimal N fertilizer rates (CRU) from winter wheat cultivation for the baseline 1961–1990 period (**a**), and qualitative changes by 2071–2100 for nine Regional Climate Models (RCM) with HadAM3H as bounding Global Climate Model for the A2 emissions scenario (**b**) with decreasing (*blue*/*dark grey*), increasing (*red*/*dark grey*) and uncertain (*green*/*light grey*) leaching. *Grey* areas are estimated to be unsuitable for winter wheat (from Olesen et al., 2007)

N fixation in legumes (Miyagi et al., 2007), with a consequently larger soil N input and therefore a higher risk of N leaching.

Increasing temperature affects crops primarily via plant development. With warming, the start of active growth is advanced, plants develop faster, and the potential growing season is extended. This may have the greatest effect in cooler regions and may be beneficial for perennial crops or crops remaining in their vegetative phase, such as sugar beets. However, increased temperature reduces crop duration for many annual crops. In wheat, an increase of 1°C during grain filling reduces the length of this phase by 5% and yield declines by a similar amount (Olesen et al., 2000). Compared to temperate crops, sensitivity to warming may be even greater in tropical crops, in particular when they are grown at the borders of their natural range. This leads to an expansion of warm-season crops (e.g. maize and sunflower) into areas currently dominated by small-grain cereals and oilseed crops (Olesen & Bindi, 2002).

Higher temperatures and elevated CO_2 will also entail changes in planting and harvesting times (Olesen, 2005) as well as in fertilization rates and strategies (Olesen et al., 2007). Crop rotation must be adapted to changes in crop choices as well as in crop timing and the need to control weeds, pests and diseases, and this will affect the amount of N released to freshwaters and its seasonal pattern. Moreover, an earlier harvest of crops and later planting of winter crops may result in a prolonged period of bare soil in autumn, which will increase the risk of N loss (Olesen et al., 2004), particularly in connection with heavy precipitation in temperate regions (Eckersten et al., 2001).

Warming is expected to increase soil organic matter turnover provided that sufficient water is available, and experiments have shown that increases in net N mineralisation rates may be considerably higher than the increases in soil respiration (Rustad et al., 2001). A longer period in autumn with bare soil under a warmer climate (Olesen et al., 2004)—in combination with increased soil organic matter turnover due to higher temperatures—may increase the risk of N loss, in particular through leaching (Patil et al., 2010). The increases may be greater than predicted by many models, as more recalcitrant organic matter pools are mineralised under higher temperatures (Bol et al., 2003).

Projections made at the European level for winter wheat for the 2071–2100 time-slice showed large spatial variations in N-leaching due to the many interactions between changes in crop uptake, fertilization, nitrogen mineralisation and soil water balance (Fig. 2; Olesen et al., 2007).

Runoff and nitrogen responses to climate change

Several authors have investigated the response of climate change projections with General Circulation Models (GCMs) on runoff conditions in streams and rivers, N losses from the catchment under different land use and N retention in soils and surface waters (Wright et al., 1998; Drogue et al., 2004; Graham, 2004; Arheimer et al., 2005; Andersen et al., 2006; Kaste et al., 2006; Whitehead et al., 2006; Nõges et al., 2007). Hydrological model predictions of climate change impacts on stream runoff in the North-Western region of Europe seem to be very dependent on the GCM applied as an input to the hydrological model, as shown by Whitehead et al. (2006) in a study using INCA-N for a lowland chalk stream in England. The predictions of annual runoff changes for an IPCC A2 scenario varied from large increases in annual runoff (2080s with CGCM2 and CSIRO: 34–54%) to nearly no change on an annual basis (2080s with HadCM3: −3%) (Whitehead et al., 2006). All three GCMs, however, experience an increase in the nitrate-N and ammonium-N concentrations in the river for the A2 scenario (Whitehead et al., 2006).

Differences in modelled runoff using different GCMs were also found by Graham (2004) when simulating climate change effects on river runoff to the Baltic Sea. Utilising ECHAM4/OPYC3 (Max Planck Institute, Germany) and HadAM3H (Hadley Centre, UK) for an A2 emission scenario he found that total runoff to the Baltic Sea decreased for HadAM3H and increased for the ECHAM4/OPYC3 GCM's, but with large variations for the different parts of the Baltic Sea. Arheimer et al. (2005) modelled N leaching, water discharge and N fluxes in a Swedish river using different climate change scenarios and found an average increase in N concentrations of 50% in the root zone on arable land and a 13% increase in the river N concentration. Similarly, a modelling of N-fluxes in a Norwegian catchment revealed significant increases in nitrate-N fluxes (40–50%) applying a Hadley A2 scenario (Kaste et al., 2006).

Climate-driven changes in runoff and nitrogen loading and concentration: an example from Denmark

To illustrate the effect of climate change on runoff we coupled a series of models established for Danish streams and catchments with contrasting soil types and hydrological regimes. Climate change projections by the ECHAM4/OPYC GCM (IPCC A2 scenario) were dynamically downscaled by the Danish HIRHAM RCM (25 km grid) for two time periods: 1961–1990 (control) and 2071–2100 (scenario) and used as input to the hydrological NAM model, to model water discharge (DHI, 2003).

Scenario A2 foresees a world with a continuously increasing population, regional economic development, and slow technological change and is a high emission scenario, although the recent observations show even higher CO_2 emissions from fossil fuel use than predicted by this scenario. NAM is a deterministic, conceptual, non-distributed hydrological model describing the hydrological system in a catchment by routing water with a daily time step through four linear reservoirs: snow storage, surface storage, root-zone storage and groundwater storage. The NAM model was calibrated to ten 1st and 2nd order streams and five 5th order streams within the major geo-regions of Denmark for the purpose of conducting a nationwide analysis of climate change impacts on stream hydrology (for details, see Jeppesen et al., 2009).

Average annual runoff increased when comparing the control (1961–1990) and scenario period (2071–2100) for all nine main coastal catchments in Denmark. The highest increase was found in catchments draining to the Great Belt (20%) and the Southern Belt Sea (35%). The lowest increase in average annual runoff was predicted for the Kattegat catchment (7%). Generally, the projected seasonal changes in runoff with the HIRHAM and NAM models showed more pronounced changes than those found on an annual basis (Fig. 3). The model simulations showed increases in the winter period January to March (20–50%) and a decrease in September (−20 to −40%) for all nine major coastal catchments (Fig. 3). In the other months the simulated runoff differed widely between catchments; some showing increasing and others decreasing runoff; the differences being most pronounced during April, May, August and November (Fig. 3). The most substantial monthly changes in runoff occurred in June and July in eastern coastal catchments in the HIRHAM projections of increased summer precipitation (Fig. 3).

It is difficult to predict the impact of climate change on diffuse N loadings and concentrations in rivers, as N is influenced by multiple biogeochemical

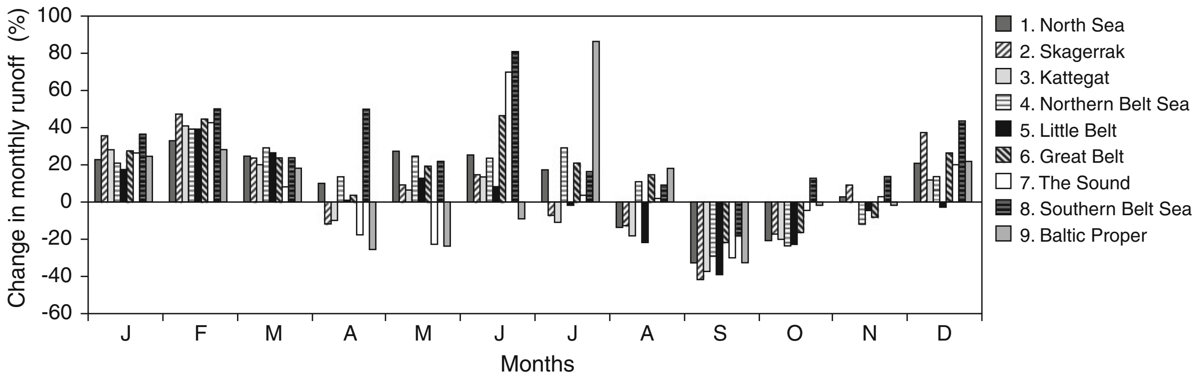

Fig. 3 Simulated monthly changes in runoff (A2 scenario) from the control period (1961–1990) to the scenario period (2071–2100) for nine major coastal catchments in Denmark

processes in soils, groundwater, rivers, wetlands and lakes (e.g. Hatfield & Follett, 2008). We therefore used input from three different models (a statistical N-model, MIKE11-TRANS model and INCA-N model) with different complexities to analyse the impact of climate change on the N load and N concentrations in surface waters at conditions of land use and agricultural practices similar to those prevailing in Denmark during the 1990s.

The statistical model being developed and deployed in this study for the climate change impact on TN losses at catchment scale is based on an extensive dataset from 93 monitored Danish catchments covering a range in land use from predominantly agricultural to predominantly forested catchments (Andersen et al., 2006). The statistical models mimic all biogeochemical processes in the catchment influencing TN loss in a lumped way. The MIKE11-TRANS model simulates N transport through the river system and the processes governing N during transport such as denitrification, uptake and sedimentation in river channels, lakes and wetlands (Kronvang et al., 1999; Andersen et al., 2006). The INCA-N model is designed to investigate the fate and distribution of N in terrestrial and aquatic environments in catchments (Wade et al., 2002a, b). We calibrated the INCA-N model on the approx. 480 km^2 Odense river basin draining to the Northern Belt Sea in Denmark for the period 1990–2001 (B. Kronvang, unpublished data). We utilised the calibrated INCA-N model to evaluate the impact of increasing temperature on the N cycling in the catchments as temperature was not included in the statistical model developed for TN loss.

The climate change impact on monthly losses of TN in the nine major Danish coastal catchments was estimated applying the following statistical model for diffuse TN losses from Danish catchments to surface waters (Eq. 1):

$$\log(Y_{ij}) = \mu + \mu_i + \alpha_j + \beta_1 \cdot \log(Q_{ij}) + \beta_2 \cdot \mathrm{AL} + \beta_3 \cdot \mathrm{SS} + \beta_4 \cdot \mathrm{OS} + \beta_5 \cdot \mathrm{WA} \tag{1}$$

where Y is TN in kg N ha^{-1}, μ is the intercept of the regression equation modified on an annual and monthly basis by μ_i *and* α_j, respectively, where i and j are the annual and monthly indexes; the β parameters are slopes of the regression line relating to runoff (Q) in mm, β_1; the percentage of arable land (AL), β_2; the percentage of sandy soils (SS), β_3; the percentage of organic soils (OS), β_4; and the percentage of wetlands in a catchment (WA), β_5. The TN model was developed based on monitoring data from 93 streams, each draining a catchment area less than 50 km^2 (Andersen et al., 2006). The TN model explained 80% of the variance for the observed diffuse TN losses from the catchments. The uncertainties of the statistical model in different seasons are shown in Fig. 4. A detailed description of the TN model can be found in Andersen et al. (2006).

The simulated changes in average annual riverine runoff, diffuse TN loading and discharge weighted TN concentrations from Denmark during the 30-year control period and the 30-year scenario period are shown in Fig. 5. The climate change simulated enhanced diffuse TN loading during winter and

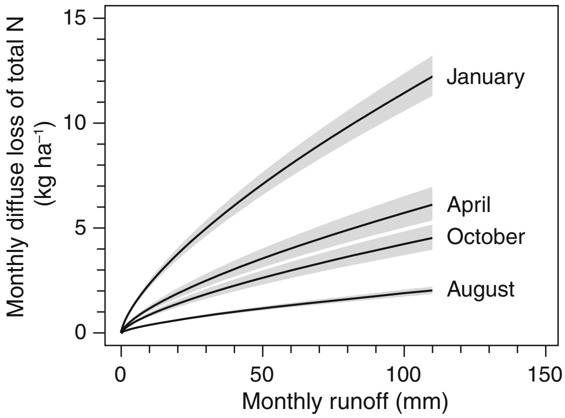

Fig. 4 Calculations of catchment loss of diffuse TN (with 95% confidence bands) for increasing runoff in selected months using the statistical model (Eq. 1)

spring with increases between 2.7 and 25%, the highest increase occurring in February. The simulated diffuse TN loading decreased during the late summer and autumn period (August–November), the most marked decrease in TN loading occurring in September (−37%) (Fig. 5).

The discharge weighted concentration of TN decreased in nine of the months as a combined result of the model simulated changes in monthly runoff and diffuse TN loadings (Fig. 5). The decrease in TN concentrations is most pronounced during the winter period (January–March) (6.2–9.7%). The discharge weighted concentration of TN is predicted to increase during the months of April, August and October. As the increase occurs during the productive period, it would influence primary production in standing waters such as wetlands and lakes.

The statistical model does not directly consider the impacts of a monthly increasing air temperature of 2.5–4.4°C, which is the mean projected change for Denmark between the control period (1961–1990) and the scenario period (2071–2100) (Andersen et al., 2006). The increase in air temperature is expected to be highest during summer and autumn (August–October) and lowest during spring (February–June) (Andersen et al., 2006). Evidently, such changes in air temperature will influence N processes such as mineralisation, nitrification and denitrification in soil and water compartments. We analysed the response of an isolated change in the monthly air temperature for the River Odense catchment with the INCA-N model by simulating the HIRHAM predicted changes in air

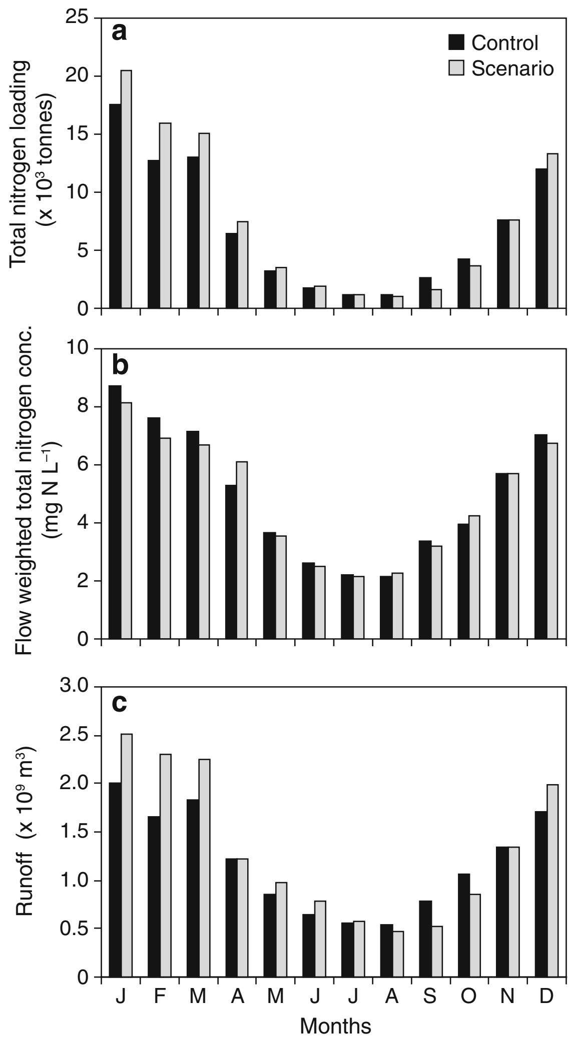

Fig. 5 Modelled average monthly changes in total nitrogen loading from diffuse sources to surface waters (**a**), flow-weighted total nitrogen concentration (**b**), and runoff (**c**) in two major river basins in Denmark from the control period (1961–1990) to the scenario period (2071–2100)

temperature for the catchment. We found that the average annual nitrate-N and ammonium-N concentration increased by 6.1% and decreased by 16.6%, respectively, for a 12-year model simulation period (1991–2002). The average annual sum of nitrate-N and ammonium-N, however, depicted a 5.9% increase during the simulation period. Considered as monthly changes, the simulation period showed an increase in nitrate-N loadings during autumn and winter and a decline in spring and summer, while the load of ammonium-N fell throughout the year as expected from the concentration data on River Odense (Fig. 6).

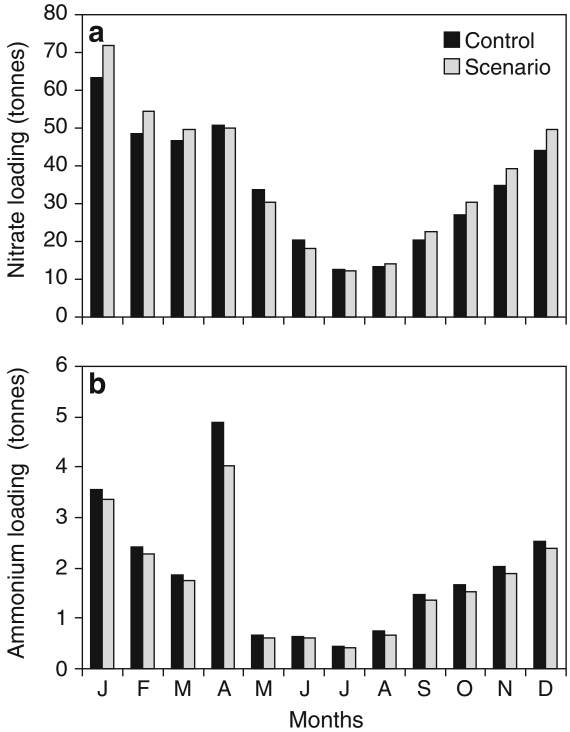

Fig. 6 Modelled average monthly changes in nitrate (**a**) and ammonium (**b**) loading to surface waters from the control period (1961–1990) to the scenario period (2071–2100) (A2 Scenario) in the river Odense catchment using the INCA-model and the HIRHAM predicted changes in air temperature in the catchment. A predicted change in the precipitation will further enhance the loading (see text)

In summary, the two independent models applied both simulated an increase in average annual N loadings. The statistical model simulated an average annual increase of 7.0% for total N loading in the A2 scenario using only precipitation and runoff as drivers, whereas the INCA-N model simulated an additional increase of 5.1% due to the projected temperature increase. However, it must be emphasised that changes in agricultural practices, crops and land as a response to climate change may easily override such model predictions of changes in climate. Also the projections of precipitation changes are uncertain (Kendon et al., 2010).

Riparian zones

Riparian wetlands often function as discharge areas for groundwater to nearby streams. Increased runoff and increasing nitrate concentrations from upland agricultural areas, according to the above mentioned scenarios, will thus result in an increased N load and wetter conditions in riparian wetlands. As temperature is one of the parameters controlling denitrification, the process rate will increase provided that one or more of the other controlling parameters (i.e. pH, nitrate concentration, carbon as energy source, redox conditions) are not rate limiting. Most European studies of N removal in riparian wetlands have shown high removal rates and high efficiency (i.e. up to 100% removal) due to nitrate reduction by denitrification (Haycock & Pinay, 1993; Vought et al., 1994; Blackwell & Maltby, 1998; Sabater et al., 2003; Hoffmann et al., 2000, 2006a, b). Two French studies have shown that N removal in riparian wetlands plays a significant role at catchment level (Sánchez-Pérez et al., 1999; Montreuil & Merot, 2006). In one area covering 40 km^2 N removal was estimated to 1,398 kg N ha^{-1} $year^{-1}$ (Sánchez-Pérez et al., 1999), while in another study including 18 selected catchments (total 400 km^2) a 30% reduction in nitrate concentrations was estimated due to wetland discharge (Montreuil & Merot, 2006). However, while an increased N loading in a future wetter climate in the North Temperate Zone may lead to higher loss in wetlands, the efficiency of N removal may abate due to shorter hydraulic retention time. Moreover, a few studies have revealed that the effective nitrate removal was exceeded or counterbalanced by leaching of dissolved organic N (Davidson et al., 1997; Kieckbusch & Schrautzer, 2007). So clearly, it is still uncertain how the capacity of wetlands to remove N will be affected by climate change and which effects will occur on, for instance, organic N and P mobilisation.

A higher nitrate load may also result in higher nitrous oxide emission (Groffman et al., 2000), especially in overloaded (N-stressed) riparian areas because the potent greenhouse gas N_2O as an end product of denitrification may increase (Hefting et al., 2003). The emission of nitrous oxide from riparian areas may take place as a result of nitrate-rich and/or nitrous oxide-rich groundwater coming from upland agricultural areas which recharge riparian wetlands, thus being hot spots for N_2O emission and/or zones of enhanced denitrification (Fleischer et al., 1998; Kroeze & Mosier, 2002; McClain et al., 2003). Denitrification of both nitrous oxide and nitrate in groundwater recharging a permanently water-covered riparian fen has been documented (Blicher-Mathiesen & Hoffmann, 1999).

Loss of the riparian zones: examples from Denmark

The current intensive agricultural production of cereals in most Danish riparian areas is becoming increasingly difficult to sustain due to increased autumn and winter rainfall and thus wet and often inundated conditions during periods of soil cultivation in spring and autumn. The problems will increase considerably under the projected climate change, and it is predicted that in many riparian areas intensive agricultural production must cease in the future (Andersen et al., 2006). Back in 1998 the Danish government adopted the Second Action Plan on the Aquatic Environment to reduce N losses from agriculture. The plan included, for the first time, large scale restoration of former wetlands as a measure to reduce N loading to coastal marine waters (Kronvang et al., 2008). A total of 3,844 ha of former wetlands were restored or approved for restoration in 57 projects in 2005 (Hoffmann et al., 2006b; Hoffmann & Baattrup-Pedersen, 2007). A 1-year mass-balance monitoring of some of the wetlands revealed that they removed between 14 and 337 kg N ha^{-1} year^{-1} (Table 2) with an efficiency varying between 28 and 100%. The amount of N removed depended on climatic conditions, especially precipitation (Hoffmann et al., 2006b; Hoffmann & Baattrup-Pedersen, 2007) and, thus, actual N removal also varied compared to prior estimated N removal rates (Table 2).

Model projections under climate change documented that an increase in stream flow during winter will impact the interactions between streams and their riparian areas. Estimations with the hydrodynamic MIKE11 model show a 50% increase in the average number of days per year with overbank flooding from the control period (1961–1990) to the scenario period (2071–2100) for the River Gjern in Central Jutland, Denmark (Andersen et al., 2006). The predicted increase in overbank flooding will worsen the problems for intensive farming of riparian areas and may therefore promote the willingness to restore former drained wetlands. Restoration of wetlands will be a very efficient

Table 2 Measured and estimated nitrogen removal rated in 12 restored wetlands

Name of project	Area (ha)	Wetland type	Measured N-removal (kg N ha^{-1} year^{-1})	Changed land use—CLU (kg N ha^{-1} year^{-1})	Measured N-removal + CLU (kg N ha^{-1} year^{-1})	Estimated N-removal (kg N ha^{-1} year^{-1})
Egebjerg Meadows	34	Irrigation + inundation	53	–	53	200
Kappel	28	Irrigation	14	25	39	140
Geddebækken	39	Irrigation + stream water	90	35	125	215
Horne Mølleå	14	Irrigation	220	35	255	200
Karlsmosen	63	Irrigation + inundation	337	35	372	270
Lindkær	84	Irrigation	191	35	226	235
Snaremose	34	Irrigation	256	35	291	200
Frisvad Mill Brook[a]	39		(95)	–		279
Ulleruplund	13	Irrigation	133	37	170	210
Gammelby Brook	27	Irrigation + stream water	83	22	105	343
Nagbøl River	64	Irrigation + inundation	163	24	187	300
Hjarup Brook	31	Irrigation + inundation	170	30	200	475

Irrigation type is wetlands receiving tile drainage water from disconnected upland drains, thereby allowing water to infiltrate or flow across the wetland area. CLU is calculated reduction in land use input from the restored area. Crop rotation and permanent grass have been given standard leaching rates of 50 and 10 kg N ha^{-1} year^{-1}, respectively, and CLU was then calculated from their percentage share of the area (adapted from Hoffmann et al., 2006b; Hoffmann & Baattrup-Pedersen, 2007)

[a] Uncertain due to extremely dry conditions

adaptation strategy to store water in the river systems and retain N from higher-lying intensively farmed areas. They will, at least in the short-term, also be able to capture phosphorus and function as a carbon sink.

North European temperate lakes

How changes in N loading due to climate change will affect lake ecosystems in a warmer climate is debatable. In most studies, phosphorus (P) is considered the most important limiting nutrient in lakes and the one responsible for eutrophication (Schindler, 1977; Schindler et al., 2008). However, several authors advocate for consideration of both N and P (e.g. Conley et al., 2009, Abell et al., 2010). In a recent meta-analysis including experimental investigations of limitation in various ecosystems, Elser et al. (2007) found as many instances of limitation by N as by P in freshwaters. The importance of both N and P was also highlighted in a review by Lewis & Wurtsbaugh (2008) and in studies of Chinese Lake Taihu (Xu et al., 2010; Paerl et al., 2010) and New Zealand lakes (Abell et al., 2010). Studies of macrophytes dominated shallow lakes further emphasise the role of N as high N may negatively affect the species richness and abundance of submerged macrophytes in lakes (Moss, 2001; González Sagrario et al., 2005; James et al., 2005) (Fig. 7), likely due to shading by phytoplankton and epiphytes. Submerged macrophytes play a key role for maintaining a clear water state in eutrophic shallow lakes (Moss, 1990; Scheffer et al., 1993) and loss of these plants may shift a lake from clear to turbid. Enclosure experiments run at various combinations of P and N in a shallow Danish lake indicated that the risk of loss of submerged macrophytes was high when N was above 1.2–2 mg l^{-1} and P higher than 0.1–0.2 mg l^{-1} (González Sagrario et al., 2005). Accordingly, submerged plants typically disappear in Danish lakes at concentrations of 1–2 mg N l^{-1} when total phosphorus (TP) is moderately high (González Sagrario et al., 2005; Jeppesen et al., 2007a, b) (Fig. 7). A recent experiment conducted by Barker et al. (2008) supports these results, suggesting a critical TN threshold of 1.5 mg N l^{-1} to maintain a stable high species richness. However, tank experiments have shown that if P is low the role of N is of minor

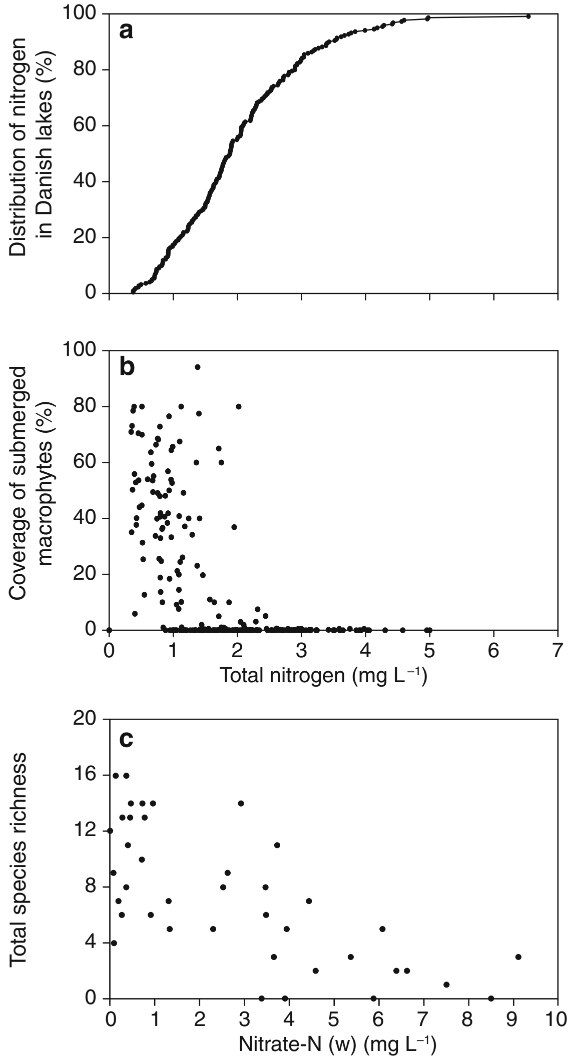

Fig. 7 The distribution of summer averages of nitrogen in 204 Danish lakes (**a**) and the coverage of submerged macrophytes in late summer in 44 Danish lakes (**b**) along a lake nitrogen gradient (summer mean). All lakes have a mean depth <5 m and a lake area >5 ha (from Gonzales Sagrario et al., 2005). Also shown is the number of species in English and Polish lakes along a gradient in winter (w) nitrate concentrations (**c**) (from James et al., 2005)

importance for nutrient tolerant species such as *Vallesneria* (Li et al., 2008).

The role of N for dominance of potentially toxic cyanobacteria is also debatable. Some authors argue that low N:P ratios in the inlet of lakes and within the lake as well will facilitate cyanobacteria growth and dominance in eutrophic lakes (e.g. Smith, 1982; Vrede et al., 2009) due to the capacity of some species to fixate N_2 dissolved in the water, but this pattern is far from general (Jensen et al., 1994; Jeppesen et al., 2005). Several studies indicate that warming will stimulate cyanobacteria and likely prolong the period with cyanobacteria blooming (Romo et al., 2005; Blenckner et al., 2007; Paerl & Huisman, 2008). Multiple regression analyses of data from 250 Danish lakes sampled in August showed higher dominance of cyanobacteria in terms of biovolume in eutrophic lakes, most notably of potentially N-fixing forms, but also of dinophytes at higher temperatures. There was a tendency to an increase in chlorophyll *a* and total phytoplankton biovolumes, whereas not least diatoms became less important (Jeppesen et al., 2009). 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 (Kronvang et al., 2005), and thus to the N loading, as cyanobacteria dominance with increasing temperature apparently is strongly exacerbated with increasing TN concentrations at moderately high to high TP concentrations (Fig. 8). The relationship between N loading and cyanobacteria dominance is, however, ambiguous, as some studies, for example in Lake Peipsi, Estonia, show an increase in cyanobacteria

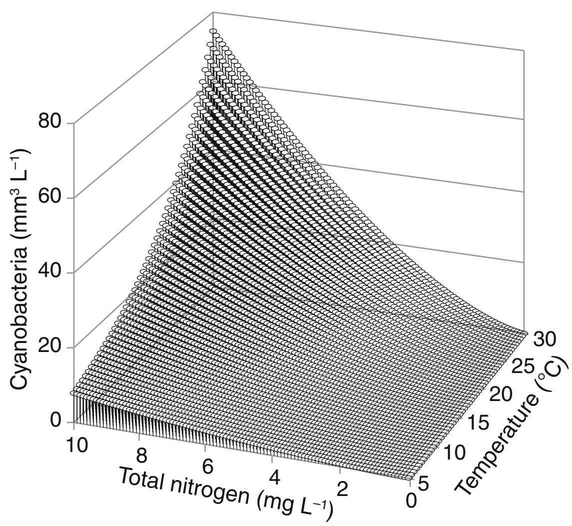

Fig. 8 Calculated effects of changes in biovolume of cyanobacteria (blue-green algae) in August in lakes at fixed depth (mean depth 3 m) and fixed total phosphorus concentration (0.1 mg P l^{-1}) but with contrasting temperatures and total nitrogen concentrations. The analysis is based on regression analyses of data from 355 lakes including log (mean depth), log (TP), log (TN) and temperature as independent variables (see further in Jeppesen et al., 2009)

dominance at low N loading, which was attributed to the capacity of N-fixing cyanobacteria to fuel the lake with N at low N:P (Nõges et al., 2004, 2005). Part of the N-cyanobacteria relationship in Fig. 8 might therefore be attributed to N fixation by cyanobacteria, though it must be emphasised that in many lakes N fixation has been shown to be of minor or modest importance (Lewis & Wurtsbaugh, 2008). Warm dry years with low water level in summer have also resulted in high phytoplankton and cyanobacteria biomass in Estonian Lake Vortsjärv (Nõges et al., 2003).

The question is then how the in-lake N concentration will change in a future warmer climate. Riverine N is retained or denitrified when passing through lakes and the loss is particularly affected by the hydraulic retention time (OECD, 1982; Lijklema et al., 1989), lake depth, trophic structure (Jeppesen et al., 1998) and temperature (Windolf et al., 1996). Several simple empirical models have been developed to describe the loss percentage on an annual (OECD, 1982; Saunders & Kalff, 2001) and a seasonal scale (Windolf et al., 1996) as a function of the external N loading. A recently updated annual model for Danish lakes (Jeppesen et al., unpublished data) is:

$$\begin{aligned} \mathrm{TN_{lake}} = \exp\big(&-0.80 \pm 0.08(\mathrm{SE}) \\ &+ 0.82 \pm 0.03\,\log(\mathrm{TN_{in}}) + 0.13 \pm 0.03\,\log(Z) \\ &- 0.20 \pm 0.02\log(\mathrm{TW})\big) \quad N = 295,\ r^2 = 0.76 \qquad (2) \end{aligned}$$

where $\mathrm{TN_{in}}$ is the discharge weighted annual mean inlet concentration, $\mathrm{TN_{lake}}$ the annual mean lake concentration, Z the mean depth (m) and TW the hydraulic retention time (years). Using annual mean input data from the studied Danish stream catchments (Fig. 3) and Eq. 2, the annual mean lake concentration is predicted to change only insignificantly, from −1 to 4% (average 1%) in the A2 scenario of climate change, while the absolute N retention (including loss by denitrification) is predicted to increase in lakes with long hydraulic retention time and to decrease when retention time is very short (Fig. 9). However, according to Eq. 2 the percentage N retention is, however, decreasing as hydraulic retention time is decreasing on an annual basis. In these model predictions, though, the potential effect of changes in temperature and seasonal variation in N dynamics have not been considered.

To illustrate the role of lakes in catchments we also applied the more complex MIKE11-TRANS model to a river basin with a 1st order stream without lakes, a sub-catchment with a 40 ha lake and a total catchment of 11,350 ha (for more information, see Andersen et al., 2006). N retention in 1st order sub-catchments without wetlands and lakes seems to decrease from the control period to the scenario period (Table 3). This is possibly due to lower N concentrations during most of the year combined with lower macrophyte biomass caused by increased plant loss as a result of enhanced runoff during winter and spring periods. In contrast, N removal increases in sub-catchments with lakes (such as the Blaamose sub-catchment), possibly due to a combination of higher N load and enhanced water temperature-induced denitrification in the lake during most of the year, counteracting the influence of lower hydraulic residence time during winter and spring (Table 3). As a result, N retention in the entire river Gjern catchment increases slightly, as the influence of wetlands and lakes on N retention is more significant than that of decreased N retention in watercourses (Table 3). This concurs with Harrison et al. (2009) who, based on sensitivity analysis of a new global model of N loss in lentic waters, concluded that, on average, N removal within lentic systems will respond more strongly to changes in land use and N load than to changes in temperature at the global scale.

The lake modelling results must be interpreted with caution, however, as other factors such as lake physics and trophic dynamics are also affected by climate (Mooij et al., 2005; Blenckner et al., 2007; Meerhoff et al., 2007; Livingstone, 2008; Jeppesen et al., 2009, 2010a), potentially impacting lake N dynamics. Higher temperatures influence the mixing regime in lakes (Livingstone, 2008). In deep summer-stratified Danish lakes the thermocline will often occur at shallower depth (Closter, 2007). Furthermore, the duration of stratification will increase, enhancing the risk of oxygen depletion below the thermocline (Blenckner et al., 2007) and, consequently, the N dynamics in the hypolimnion, whereas polymictic lakes may become temporarily stratified. An analysis of data from five summer stratified Danish lakes, monitored biweekly during summer 1989–2003 (Kronvang et al., 2005) when the lakes were subjected to nutrient loading reduction, revealed

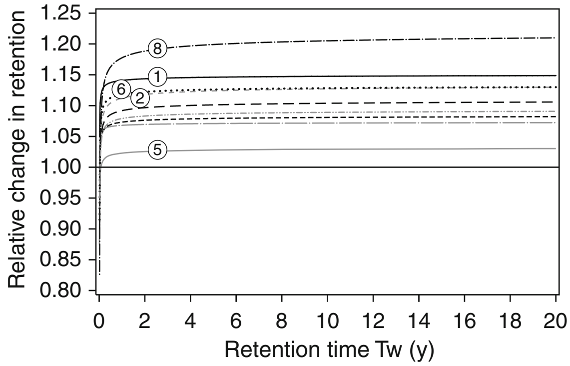

Fig. 9 Proportional changes in annual mean lake nitrogen retention from the control period (1961–1990) to the A2 scenario period (2071–2100) calculated using Eq. 2 for lakes with different hydraulic retention times. The simulated inlet TN concentrations were obtained from Eq. 1. These calculations are all based on annual means and do not consider the seasonal variations in retention. *Numbers* refer to those given in Fig. 3

a response to the loading reduction in the surface waters, but also strong interannual variations in oxygen and N concentrations in the hypolimnion, depending on, for instance, summer temperatures (Fig. 10). In years with high summer temperatures, the depth, where the minimum oxygen concentrations in the hypolimnion passed below 3 mg l^{-1} during the stratification period, was shallower than the average for the study period, but overall much deeper in colder years. Maximum ammonium concentrations were often much higher in warm years and minimum nitrate concentrations higher in cold years as a result of higher mineralisation and nitrate loss likely by denitrification during warm years (longer stratification period). However, total inorganic N and TN in the hypolimnion varied only negligibly between years and no clear effects of these changes were seen in surface water TN in the following year. TN responded more strongly to changes in external loading and was particularly low in the dry years 1996–1997 (data not shown). Chlorophyll *a* showed a declining tendency following nutrient loading reduction, but no clear effect of temperature was observed. Warming may lead to more frequent and prolonged stratification periods in temporarily stratified lakes, and some polymictic lakes may become temporarily stratified. This may enhance the sediment release of nutrients due to oxygen depletion in the bottom

Table 3 Simulated changes in total nitrogen removal (kg N ha^{-1} $year^{-1}$ in two sub-catchments and the entire River Gjern catchment for a control (1961–1990) and a scenario period (2071–2100)

Nitrogen retention in surface waters (streams, wetlands and lakes)	1st order sub-catchment (Gelbæk; 1,110 ha catchment area)	2nd order lake sub-catchment (Blaamose: 2,000 ha catchment area)	Gjern catchment (11,350 ha catchment area)
Nitrogen retention during control period	3.8	19.1	7.2
Nitrogen retention during scenario period	3.7	19.8	7.5
Change in nitrogen retention (scenario minus control period)	−0.1	0.7	0.3

water, and these nutrients may eventually reach the photic zone during overturn and stimulate phytoplankton growth, as seen in large shallow Lake Müggelsee, Germany (Wilhelm & Adrian, 2008).

Also the biological communities are altered markedly by climate change with implications for the function and eventually the loss of N (Fig. 11 and Jeppesen et al., 2010a, b). In north temperate mesotrophic lakes plankti-benthivorous fish are controlled by piscivorous fish, releasing the predation on large-bodied zooplankton (*Daphnia* spp.). Therefore, grazing on phytoplankton is relatively high and the lakes are only moderately sensitive to small increases in nutrient loading, and, when clear, the retention of nutrients is high (Jeppesen et al., 1998; Søndergaard et al., 2003, 2008) (e.g. less nitrogen in the algae that leave the lake via the outlet, and higher chances of denitrification of nitrogen entering the lake). When such lakes become eutrophic the control of piscivores is reduced and cyprinids feeding on large-bodied zooplankton abound. Accordingly, there are few *Daphnia* and little grazing on phytoplankton. Such lakes are extremely sensitive to additional nutrient loading which is channelised into phytoplankton growth. With warming we can expect major changes in the fish communities, with less piscivores, more omnivores, higher dominance of cyprinids, reproduction at a smaller size, higher proportions of small-sized fish and, accordingly, higher fish predation on zooplankton (Gyllström et al., 2005; Blanck & Lamouroux, 2007; Meerhoff et al., 2007; Jeppesen et al., 2007a, b, 2009, Jeppesen et al. 2010b). This, in turn, reduces the zooplankton grazing on phytoplankton, resulting in higher eutrophication in nutrient enriched lakes. Thus, subtropical lakes are dominated by numerous small omnivorous fish and zooplankton is scarce in both mesotrophic and eutrophic lakes (Meerhoff et al., 2007). Accordingly, these lakes are all highly sensitive to addition of nutrients and nutrient retention is likely relatively low (Fig. 11). Moreover, a change from a clear eutrophic state with macrophyte dominance to a turbid phytoplankton dominated state with high abundance of cyprinid fish reduces the capacity of lakes to function as an N filter (Jeppesen et al., 1998). This is further exacerbated by a shorter hydraulic retention time (Eq. 2), leading to higher N loading of downstream-situated N sensitive coastal areas in North Europe.

With increasing warming it will therefore become increasingly difficult to meet the present-day targets set for the ecological state of Northern European lakes. Additional efforts must be initiated to reduce the external nutrient loading to levels lower than the present-day recommendations.

Warmer lakes in the Mediterranean: examples from Turkey

In warm southern Europe, the expected reduction in precipitation and the higher evaporation in a warmer climate will lead to an even stronger reduction of runoff, which is projected to be as large as 30–40% in Mediterranean Europe and Asia. Spain and Turkey are projected to be among the worst affected in the world, from the last century to 2040, and are thus particularly threatened. In the closed Konya basin of Central Anatolia where irrigated crop farming is particularly intensive, periods of drought occurred in 37 years of the 40-year study period (since 1970). Moreover, extensive water use for irrigated crop farming led to an average 15 m drop in the ground water level, most significantly during the last 10 years (Göçmen et al., 2008). Thus, many wetlands, lakes and dams have

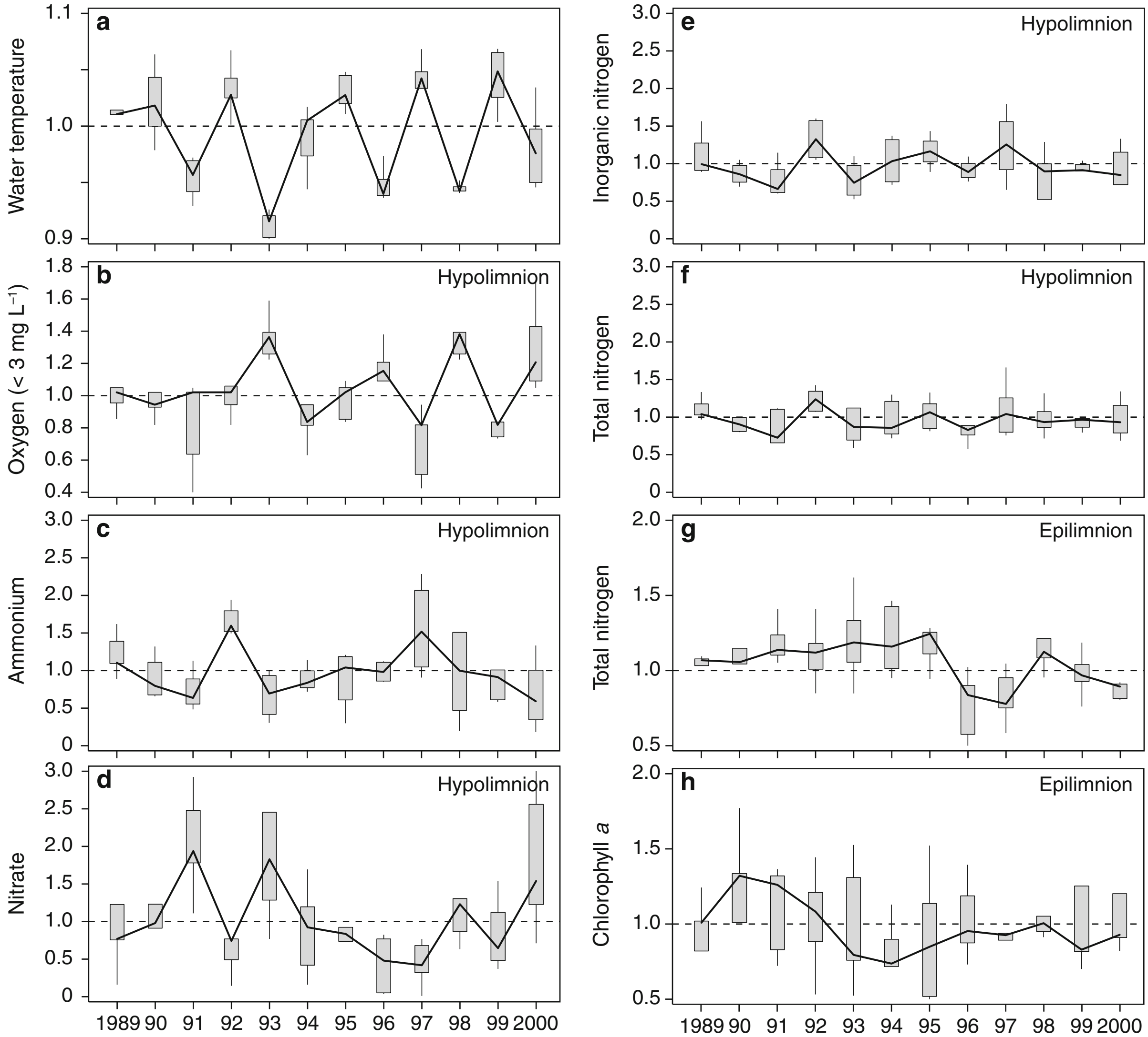

Fig. 10 Boxplot of various variables in each year normalised to the means for the period 1989–2003 in five stratified, eutrophic Danish lakes. The variables are interpolated summer mean (May 1–September 30) temperatures in the epilimnion (**a**), the minimum depth where oxygen passes below 3 mg O_2 l^{-1} (**b**), maximum ammonium-nitrogen (**c**), minimum nitrate-nitrogen (**d**), maximum inorganic nitrogen (**e**), and total nitrogen (**f**), respectively, at any sampling date during stratification, the summer mean epilimnion concentration of total nitrogen (**g**) and chlorophyll *a* (**h**). Median, 25, 75, 10 and 90% percentiles are shown

already dried out (>1,000 km^2) and the remaining water bodies face the risk of salinisation due to reduced hydraulic loading. With lower runoff, N loading from diffuse sources may decrease. However, concentrations in inlets may increase due to enhanced evaporation and water loss (Beklioglu et al., 2007). A comprehensive mass balance study of Lake Mogan, a shallow Turkish lake, shows that inorganic N increases in dry years when the water table is low despite a major reduction in external loading due to reduced runoff during 2001 and 2005–2007 (Fig. 12). The same holds true for TP and dissolved inorganic P (Jeppesen et al., 2009; Özen et al., 2010). A likely explanation is the lower volume of water in the lake and lower dissolved oxygen concentrations in the water due to higher consumption and a lower saturation level of oxygen at higher temperatures (Beklioglu & Tan, 2008; Beklioglu & Özen, 2008). This, in turn, reduces nitrification

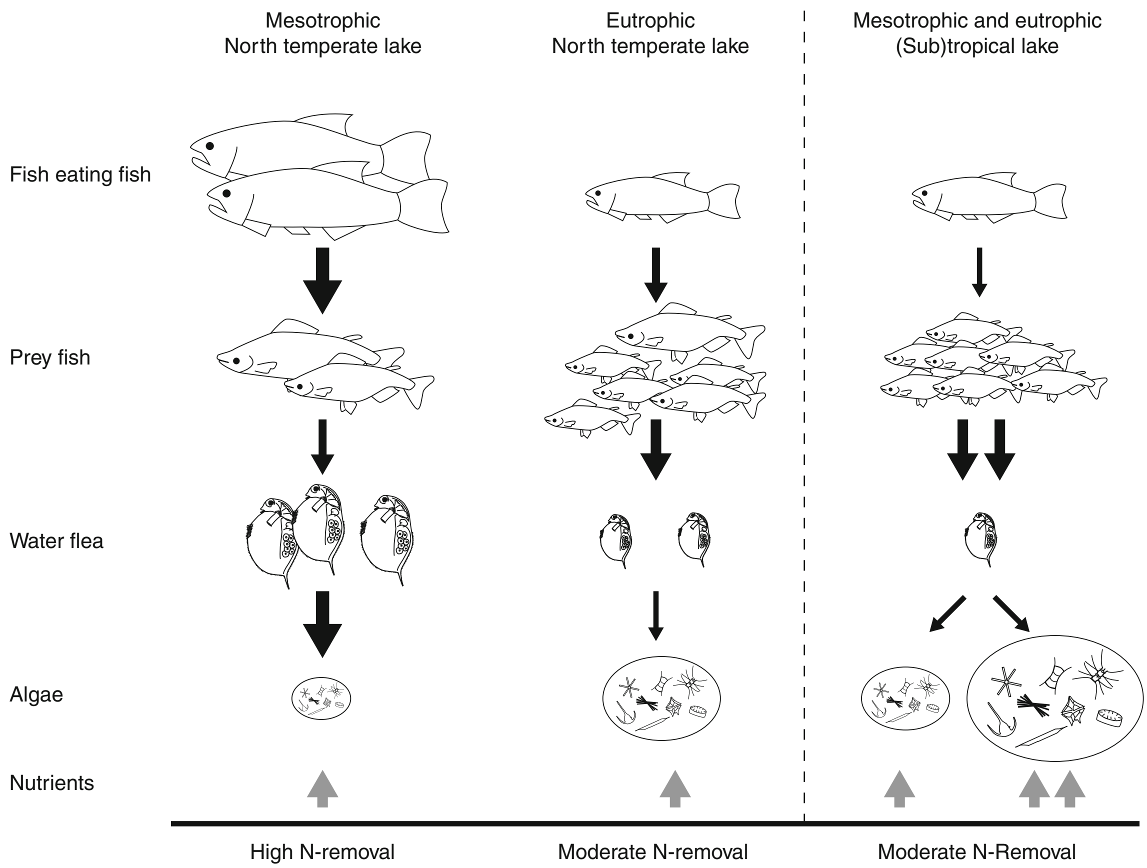

Fig. 11 Conceptual model showing trophic structure change in mesotrophic and eutrophic shallow lakes in the north temperate and subtropical lakes and its consequences for sensitivity to nutrient loading and the effects on nitrogen retention. See further in the text (modified from Jeppesen et al., 2010b)

and increases accumulation of ammonium. Higher nutrient concentrations may result in an increase in the summer phytoplankton biomass, which is dominated by cyanobacteria, and less macrophytes (Beklioglu & Tan, 2008; Özen et al., 2010). The role of N for macrophyte loss in these arid lakes has yet to be clarified. In field enclosure experiments conducted in Turkey, macrophyte presence was not sensitive to enhanced N loading, despite that the periphyton biomass, potentially shading out the plants, increased significantly (Özkan et al., 2010). However, a drop in water level during the experimental period, which occurred in summer in this area, may have counteracted the shading effect of the periphyton and therefore reduced the risk of plant loss (Özkan et al., 2010). Water level drawdown has had positive effects on plant growth in several (Beklioglu et al., 2006) but far from all Turkish lakes, especially the eutrophicated ones (Beklioglu & Tan, 2008).

Adaptation to counteract negative effects of climate change

Autonomous adaptation to climate change in the agricultural sector is likely to lead to intensified production in Northern Europe and less intensive production in Southern Europe, including reduced use of irrigation due to enhanced water scarcity (Olesen & Bindi, 2002; Alcamo et al., 2007). The intensification of North European agriculture under climate change in combination with warmer and wetter winters will most likely lead to enhanced nutrient cycling and runoff. The higher winter rainfall will make current low-lying agricultural soils in riparian areas less attractive for cultivation, which could be exploited to lessen the negative consequences of climate change for the aquatic environment by promoting the establishment of (larger) buffer zones along river corridors.

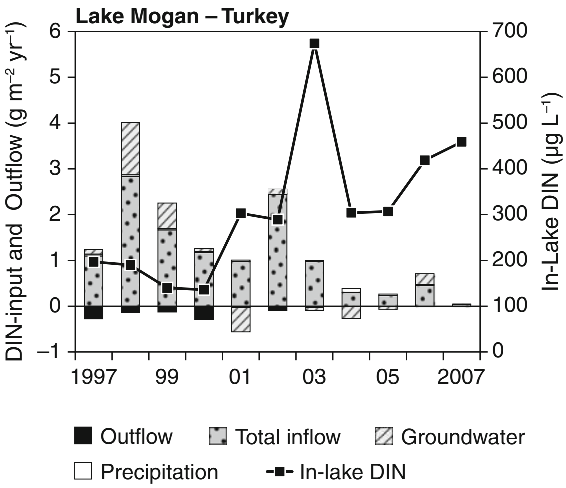

Fig. 12 Mass balance of inorganic nitrogen from Lake Mogan, a large and shallow lake in Turkey (drainage area: 925 km^2, surface area: 5.4–6 km^2, Z_{max}: 3.5 m, Z_{mean}: 2.1 m), showing annual input, output as well as the annual mean lake inorganic nitrogen concentration. The mass balance is based on water samples collected fortnightly in spring, summer and autumn and at monthly intervals in winter (Beklioglu et al., 2006), on the lake level recorded daily from a fixed gauge, on flow rates in inlet and outlet, and ground water input calculations (General Directorate of Electrical Power, Resource Survey & Development Administration, EIE, 2007). Monthly air temperature, rainfall and evaporation data were recorded by a meteorological station located within the lake catchment (Özen et al., 2010). Dry years are 2001 and 2003–2007

The major sources of N to be affected by climate changes are agricultural soils in river catchments. N loss from agricultural soils is primarily associated with vegetation-free periods, where high soil mineral N concentrations may accumulate and be lost during periods of excess rainfall. As stated above, climate change is likely to increase the period of bare soil in autumn in many regions and this, in combination with higher soil organic matter turnover and increased winter rainfall, is likely to increase the risk of N loss through runoff and leaching. To avoid excessive leaching of N to aquifers and surface waters, it becomes necessary to reduce intensive tillage in autumn and extend the period of active crop growth, either by growing catch crops, cover crops or longer-duration crops (Berntsen et al., 2006; Meza et al., 2008).

To avoid a prolonged period with bare soil after harvest, catch crops or early sown winter cereals may be grown. Catch crops grown under the prevailing growth conditions reduce nitrate leaching considerably (Thomsen & Christensen, 1999) and model simulations indicate that this may also be the case under a warmer climate (Olesen et al., 2004). However, autumn growth is mainly determined by solar radiation (Vos & van der Putten, 1997), and under the low winter solar radiance of Northern Europe neither catch crops nor cereals may respond fully to increased soil N availability and higher soil temperature (Thomsen et al., 2010). The choice of catch crop (e.g. broad leaf species vs. grasses/cereals) may be more important under raised temperatures as the response to, for instance, N availability and light intensity may interact (Vos & van der Putten, 1997), and root proliferation may be influenced (Kaspar & Bland, 1992). Many of the options available to reduce N loading under climate change will also lead to enhanced soil C sequestration and reduced N_2O emissions, thus contributing to mitigating climate change (Smith & Olesen, 2010).

To further reduce the load of N to surface waters and transport to the sea several methods are available. A key measure is to enhance the retention time of water which will lead to higher N loss by denitrification. This can be achieved by re-meandering of channelised streams, (re-)establishment of wetlands, including also constructed wetlands in the farmland areas, reduction of P loading to improve lake water clarity and thereby enhance the N loss in lakes, and maintenance of a reduced abundance of cyprinids in lakes.

To reduce the risk of eutrophication and lowering of the water table (salinisation) of shallow lakes in Southern Europe, restrictions on the human use of water are needed. Since more than 80% of the freshwater abstraction in most Mediterranean countries is used for irrigated agriculture (Flörke & Alcamo, 2005), this will imply less use of irrigation water, more effective irrigation and water distribution systems, and improved recycling of water, including waste water, within catchments (Iglesias et al., 2006).

Conclusions

The projected climate change will most probably enhance the N (and also P, Jeppesen et al., 2009) load to lakes in Northern Europe, especially in winter. Even though the concentration of N in lakes may only be negligibly affected, the direct and indirect effects of a higher temperature would likely lead to impoverished, more eutrophic conditions and reduced water clarity, not least when P is moderately high or high. However, it must be emphasised that the prediction of N transport and thus effects is uncertain as the prediction of regional precipitation and changes in land use is uncertain. In arid southern Europe water quality and ecological status might also decrease despite reduced nutrient loadings, as N and P concentrations increase due to evaporation. Therefore, the critical nutrient loading for a shift from moderate to good ecological state according to the EU Water Framework Directive will decrease. To counteract this deterioration, the external nutrient loading must be further reduced. Adaptations in the North Temperate Zone should include more sustainable agriculture with less loss of nutrients to surface waters. The measures applied in the agricultural production system should focus on improved utilisation of animal manure, fertilizer and crop rotation plans, improved exploitation of feed-stuffs and limitations on TN application. Other methods to reduce the N input are (re-)establishment of wetlands in the lake catchment, as these have a high N removal capacity, introduction of riparian buffer zones and re-meandering of channelised streams. In arid Southern Europe, restrictions on the human use of water are needed, in particular concerning irrigated agriculture.

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