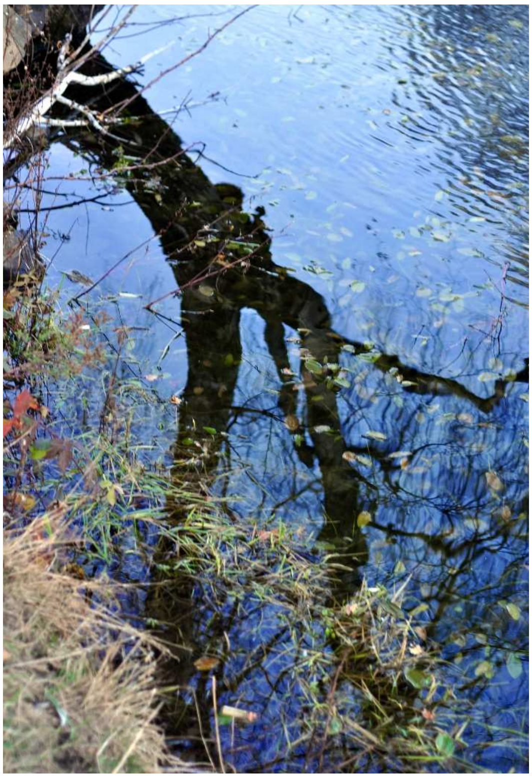
European Environment Agency
European Topic Centre on Water

Climate change impacts on water quality and biodiversity

Background Report for EEA European Environment State and Outlook Report 2010



ETC Water Technical Report 1/2010

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Contents

Introduction

The aim of this report is to provide an overview of the climate change impacts on water quality and freshwater biodiversity as a background report for the EEA European Environment State and Outlook Report 2010. This background report is based on the EEA/JRC/WHO Impacts of Europe's changing climate - 2008 indicator-based assessment (EEA, 2008) and updated with information in recent publications and with the 2010 update of the EEA climate change indicators: CLIM 19 Water temperature, CLIM 20 Lake and river ice and CLIM 21Freshwater biodiversity and water quality.

Climate change can result in significant changes in the variables and processes that affect water quality and freshwater biodiversity. These include:

- physical changes such as increased water temperature, reduced river and lake ice cover, more stable vertical stratification and less mixing of water of deep-water lakes,and changes in water discharge, affecting water level and retention time;
- chemical changes, such as increased nutrient concentrations and water colour (DOC), and decreased oxygen content
- biological changes, including northwards migration of species and alteration of habitats, affecting the structure and functioning of freshwater ecosystems.

Changes in these variables lead to impacts on all the socio-economic and environmental goods and services that depend on these systems directly or indirectly.

European freshwaters are already being affected by many human activities, resulting in changes in land-use, pollution with nutrients and hazardous substances, and acid deposition. Because of difficulties in disentangling the effects of climatic factors from other pressures, there is limited empirical evidence to demonstrate unequivocally the impact of climate change on water quality and freshwater ecology. On the other hand, there are many indications that freshwaters that are already under stress from human activities are highly susceptible to climate change impacts and that climate change may significantly hinder attempts to restore some water bodies to good ecological status.

Currently many national and European research activities are producing relevant and valuable results on climate change impacts on Europe's freshwater; see for example Euro-limpacs (Kernan et al., 2010 and http://www.eurolimpacs.ucl.ac.uk/index.php) or the IPCC report on climate change and water summarising the different water sections in the Fourth assessment report of the IPCC (IPCC, 2007). A new FP7-project (REFRESH: http://www.refresh.ucl.ac.uk/) has recently started with the main objective to develop adaptive strategies to mitigate the impacts of climate change on European freshwater ecosystems, including relevant restoration measures.

The content of this background report is structured into six main chapters:

1. Direct and indirect impacts on water quality (physical and chemical changes)
2. Impacts on biodiversity
3. Impacts on ecological status of lakes and rivers
4. Impacts on water use including human health
5. Possible adaptation measures
6. Conclusions and key messages

1. Direct and indirect impacts on water quality (physical and chemical changes)

Climate change will affect water quality both through increased temperature and through hydrological changes in rainfall affecting run-off and mobilisation of nutrients and other pollutants. The increased water temperature will affect ice cover and circulation patterns in lakes and rivers, as well as the rate of biogeochemical and ecological processes that determine water quality. In areas where river flow and groundwater recharge will decrease, water quality may also decrease due to less dilution of pollutants. Higher intensity and frequency of floods and more frequent extreme precipitation events are expected to increase the load of pollutants (organic matter, nutrients, and hazardous substances) washed from soils and overflows of sewage systems to water bodies. This may result in:

- *Increased water colour* due to increased input of humic substances as dissolved organic carbon (DOC) from the catchment.
- *Reduced oxygen content.* Increased biological respiration rates result in lower dissolved oxygen concentrations, particularly in summer low-flow periods and in the bottom layers of lakes. Higher temperature and lower oxygen concentrations will cause stress and may reduce the habitats of cold-water species such as salmonid fish in lakes and rivers.
- *Increased nutrients.* Increased mineralisation and releases of nitrogen, phosphorus and carbon from soil organic matter and increased run-off and erosion will result in increased nutrient loads. Also release of phosphorus from bottom sediments in stratified lakes is expected to increase, due to declining oxygen concentrations in the bottom waters.
- *Increase in pathogenic microbes:* Sewage overflows upon heavy rains combined with higher water temperatures and longer ice-free season may increase the number of pathogenic microbes in water.

Many of the diverse aspects of climate change (e.g. temperature increase, variations in rainfall and runoff) affect the distribution and mobility of hazardous substances in freshwater systems. Loading of hazardous substances may increase due to sewage overflow, as well as higher pesticide use and run-off due to heavy rains, while higher temperatures increase the degradation rate of some pesticides and organic pollutants, which may reduce their concentrations in rivers and lakes. Thus the net effect of climate change on hazardous substances is uncertain. However, higher air and water temperatures may change the migration and biological uptake of atmospherically-transported toxic organic pollutants including those already banned (Grimalt et al., 2001).

This chapter is structured into 8 sections focusing on climate change impacts on:

- water temperature and lake circulation
- ice cover
- underwater light conditions
- DOC concentration
- oxygen
- nutrient concentrations

- hazardous substances
- pathogenic microbes in water.

1.1. Impacts on water temperature and lake circulation

Mechanisms

Since water temperature is mainly determined by heat exchange with the atmosphere, higher air temperatures lead to higher water temperatures. For rivers, there are strong correlations between water and air temperature (Webb and Nobilis, 2007; Moatar et al., 2006; Versteeg et al., 2005; Webb et al., 2003; Langan et al., 2001).

Increased lake temperature in deep lakes is connected with higher thermal stability and a deeper thermocline (DeStasio et al., 1996; Hassan et al., 1998). Increased thermal stability and stratification has been found in Lake Maggiore (Ambrosetti and Barbanti, 1999) and Lake Zurich (Livingstone, 2003) in relation to increased air temperatures. Shallow lakes, due to their smaller volume and limited stratification in summer, are known to respond more directly to prevailing weather conditions (Gerten & Adrian, 2001). However, increased thermal stability may become more common also here. Wilhelm and Adrian (2008) studied the shallow polymictic lake Müggelsee in the period 2003 to 2006. They found that during summer, periods with stratified conditions exceeded those with mixed conditions, and particularly long stratification events developed during the exceptional heatwaves occurring in Europe in 2003 and 2006. Hypolimnetic temperatures were higher than what is common for dimictic lakes, because stratification builds up when lake temperatures are already high.

Water temperature may also be affected by other factors than air temperature. Especially in larger rivers, flow regulation, cooling water from power plants, ground water flow and extreme hydrological events (Moatar et al., 2006; Liu et al., 2005; Versteeg et al., 2005) may be important. For smaller rivers, removing tree cover along the stream can lead to dramatic increases in water temperature (Gomi et al., 2006). Earlier ice-off gives more rapid warming of lake water in spring/summer (Moore et al., 2009). Higher wind speed could reduce stratification stability (George et al., 2007), but rapid warming of lake surface water in spring can prevent spring circulation and cause higher phytoplankton biomass (Straile et al., 2003).

Deep water warming in coloured lakes is less responsive to changes in air temperature than in clear-water lakes (Snucins and Gunn, 2000). In Clearwater Lake, Ontario, a drop in bottom temperatures of 7 ºC and a thermocline increase of 4 m in the period 1973-2001 was related to a marked increase in DOC concentration (and thus vertical light attenuation) (Tanentzap et al., 2008). The increased thermocline was also related to decreased wind speed, due to afforestation.

Past trends

Long time-series, covering the past 100 years, show that the surface water temperatures of some of the major rivers and lakes in Europe have increased by 1–3 °C over the last century (Fig. 1.1; EEA, 2008). The temperature of the river Rhine increased by 3 °C between 1910 and 2006. Two-thirds of this is estimated to be due to the increased use of cooling water in Germany and one-third to the increase in temperature as a result of climate change (MNP, 2006). In the river Danube the annual average temperature increased by around 1 °C during the last century. A similar temperature increase was found in some large lakes: Lake Võrtsjärv in Estonia had a 0.7 °C increase between 1947 and 2006 and the summer (August) water temperature of Lake Saimaa, Finland increased more than 1 °C over the last century.

A number of shorter time-series of water temperature covering the past 30–50 years indicate a general trend of increasing water temperature in European freshwater systems of 0.05 to 0.8 °C per decade.

The temperature of Lake Windermere (England) and Lough Feeagh (Ireland) increased by 0.7–1.4 °C between 1960 and 2000 (George and Hurley 2007). The water temperature of Lake Veluwe (the Netherlands) has increased by more than 1 °C since 1960 (MNP, 2006). Mooij et al. (2008) modelled the water temperature in three shallow lakes in the Netherlands, and found an increase of almost 2 ºC over the period 1961-2006. Marked increases in water temperature were found in eight Lithuanian lakes (Pernaravičiūtė, 2004) and six Polish lakes (Dabrowski et al., 2004). Since 1950, water temperatures in rivers and lake surface waters in Switzerland have in some cases increased by more than 2 °C (BUWAL, 2004; Hari et al., 2006). In the large lakes in the Alps the water temperature has generally increased by 0.1–0.3 °C per decade: Lake Maggiore and other large Italian lakes (Ambrosetti and Barbanti, 1999), Lake Zürich (Livingstone, 2003), Lake Constance and Lake Geneva (Anneville et al., 2005).

Also in Lake Baikal, there was a significant increase in water temperature at the surface and at 25 m in the period 1945-2005 (Hampton et al., 2008). Surface waters warmed on average 0.20 ºC decade^{-1}, at 25 m on average 0.12 ºC decade^{-1}. The seasonal analysis showed that surface water warming in summer seems to have induced warming of deeper waters in the fall. Lake Baikal is important in a climate change context, because as the world’s largest lake by volume, it is expected to be among the most resistant to warming.

Dokulil et al. (2006) studied the trend in hypolimnion (bottom water) temperature in 12 deep European lakes and found generally a temperature increase of 0.1–0.2 °C per decade.

Projections

As water temperature is closely linked to change in air temperature, the predicted increase in air temperature due to climate change will be reflected in increased surface water temperature. This is in addition to temperature changes caused by other factors, such as changes in cooling water releases. Projected increases in surface water temperatures are often 50 to 70 % of the projected increases in air temperature. In line with the projected increases in air temperature, lake surface water temperatures may be around 2 °C higher by 2070, but with a clear seasonal dependency and depending on lake properties (Malmaeus et al., 2006; George et al., 2007).

Physical modelling studies of medium-sized lakes in the temperate zone predict that temperatures will increase more in the upper regions of the water column than in the lower regions, resulting in generally steeper vertical temperature gradients and enhanced thermal stability (Hondzo and Stefan, 1993; Stefan et al., 1998; Peeters et al., 2002). This may have significant effects on mixing of water in lakes, which in turn affects deep water oxygen conditions, nutrient cycling and phytoplankton biomass (Straile et al., 2003).

Conclusion

Surface water temperatures have been observed to increase and will continue to increase. During the last century the water temperature of some of the major European rivers and lakes increased by 1-3 ^{o}C, mainly as a result of air temperature increase, but also locally due to increased inputs of heated cooling water from power plants. In line with the projected increases in air temperature, lake surface water temperatures may be around 2^{o}C higher by 2070 than today. Increased temperature gives increased thermal stability and less mixing of the water column. Further increases in DOC concentrations may partly counteract the deep water warming by reducing the light penetration.

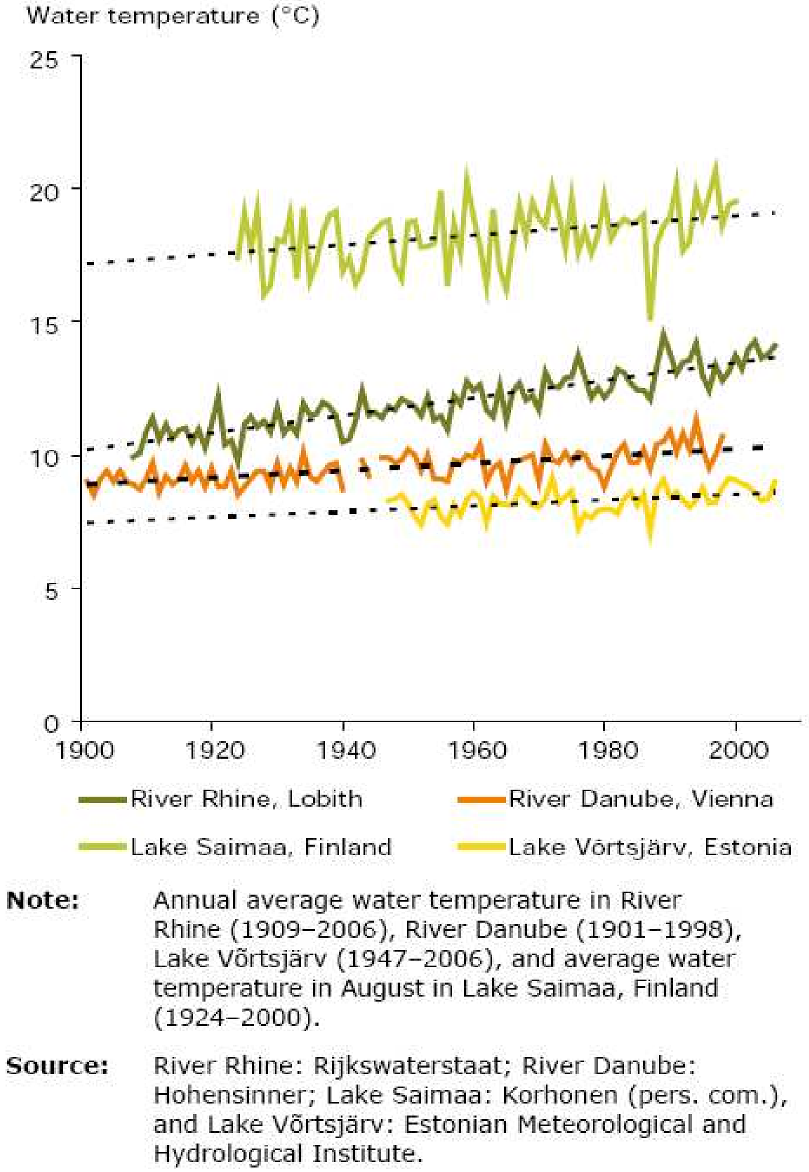


Figure 1.1 - Trend in annual water temperature in river Rhine (1909-2006), Danube (1901-1998), Lake Vörtsjärv, Estonia (1947-2006) and average water temperature in August in Lake Saimaa, Finland (1924-2000) (EEA, 2008).

1.2. Ice cover

Mechanisms

The appearance of ice on lakes and rivers requires prolonged periods with air temperatures below 0 °C. Franssen and Scherrer (2008) found good correspondence between the sum of negative degree days (NDD) and the probability of lake freezing. The sum of NDD needed for a 50% probability of freezing was positively related to lake depth, meaning that more cold is needed the deeper the lake is. This is because deeper lakes have a higher volume where heat is stored compared to the surface area where heat is lost (Franssen and Scherrer, 2008).

Higher temperatures will affect the duration of ice cover, the freezing and thawing dates and the thickness of the ice cover. Air temperature is the key variable determining the timing of ice break-up (Palecki and Barry, 1986; Livingstone, 1997). Wind has a strong impact on timing of both ice formation and break-up (Moore et al., 2009). For rivers, also discharge affects the ice cover, giving retarded freeze-up and accelerated break-up with increasing flow and vice versa (Beltaos and Prowse, 2009).

Climatic conditions not only influence the timing and duration of the ice cover, but also the thickness of the ice cover and the nature of break-up. This is especially important in rivers. In years with low snowpack and/or protracted spring melt, the ice-break will mainly be thermal, characterised by extensive ablation and weakening of the ice cover prior to an increase in flow, if any. With a thick snowpack and rapid melt, break-up will be mainly mechanical, characterised by a rapidly rising flow driven into a thick and mechanically strong ice cover. This process increases the chance of ice jamming and flooding (Beltaos and Prowse, 2009).

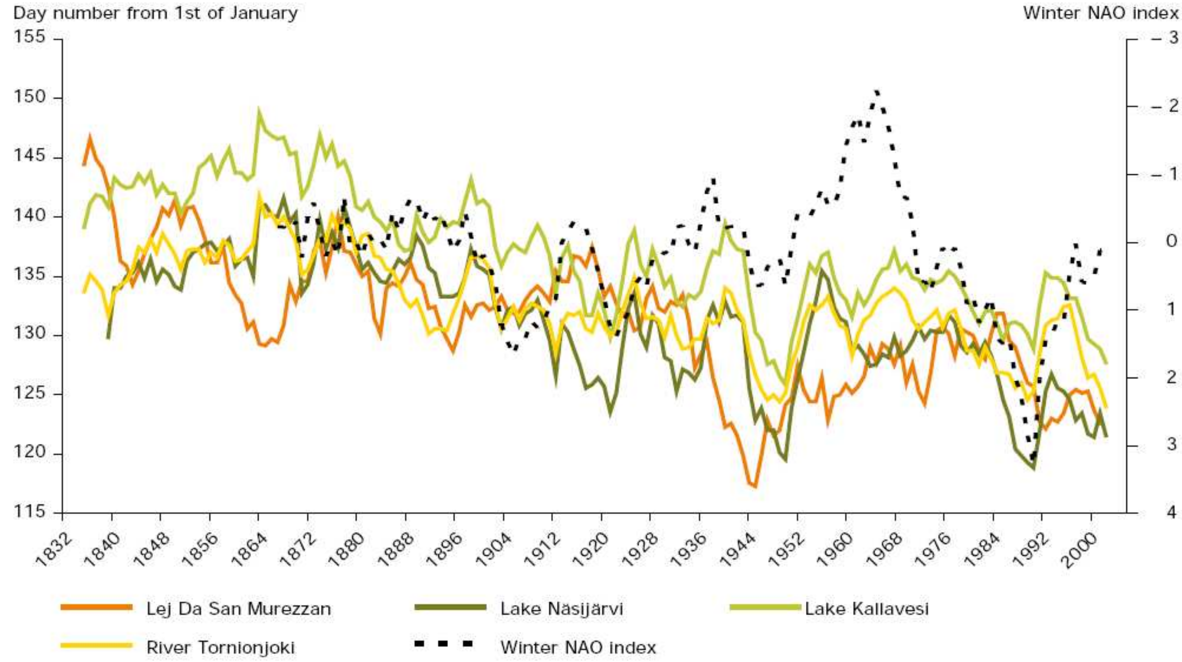
Past trends

An analysis of long (more than 150 year) ice records from lakes and rivers throughout the northern hemisphere by Magnuson et al. (2000) indicated that ice cover has been occurring on average 5.8 days later per 100 years, while ice break-up has been occurring on average 6.5 days earlier, implying an overall decrease in the duration of ice cover at a mean rate of 12 days per 100 years (Fig. 1.2). These results do not appear to change with latitude, or between North America and Eurasia, or between rivers and lakes.

A few longer time-series reveal reduced ice cover (a warming trend) beginning as early as the 16th century, with increasing rates of change after about 1850. The early and long-term decreasing trend in the ice break-up dates is the result of the end of the Little Ice Age, which lasted from about 1400 to 1900 (Kerr, 1999).

Beltaos and Prowse (2009) reviewed various studies of river ice in the northern hemisphere/Arctic, showing an overall trend towards earlier spring break-up. However, there was considerable spatial variability in freeze-up date trends. In most cases, the ice season decreased, but there were several exceptions. The changes were often most pronounced in the last few decades of the 20th century. As an overall approximation, the authors suggest that the autumn and spring warming occurring in the 20th century warming has produced a 10-15 days delay in freeze-up and advance in break-up in these areas.

Studying ice cover information from 11 Swiss lakes over the last century, Franssen and Scherrer (2008) found that the freezing frequency of lakes that freeze rarely was significantly reduced in the past 40 years, and especially during the past two decades. In Lake Baikal, the ice-free season has lengthened and the ice thickness decreased in the last century (Hampton et al., 2008; Shimaraev et al., 2002). However, Karetnikov and Naumenko (2008) found only minor changes in ice cover for Lake Ladoga, north-west Russia in the period 1943 to 2006.



Note: Data smoothed with a 7-year moving average. See Box 5.1 'Atmospheric circulation patterns in Europe'.

Source: Benson and Magnuson 2000 (Updated to 2006 by J. Korhonen, Finnish Environment Institute (SYKE) and D. Livingstone, Water Resources Department, Swiss Federal Institute of Environmental Science and Technology (EAWAG)).

Figure 1.2 - Ice break-up dates from selected European lakes and rivers (1835–2006) and the North Atlantic Oscillation (NAO) index for winter 1864–2006 (EEA, 2008)

The sensitivity of ice cover to increased temperature may vary with the initial climatic conditions. A study of 196 Swedish lakes along a latitudinal temperature gradient revealed that a 1 °C air temperature increase caused an up to 35 days earlier ice break-up in Sweden's warmest southern regions with annual mean air temperatures around 7 °C. It caused only about 5 days earlier break-up in Sweden's coldest northern regions where annual mean air temperatures are around − 2 °C (Weyhenmeyer et al., 2004; Weyhenmeyer, 2007). The same effect is observed in Finland, where ice break-up happens significantly earlier now than in the late 19th century, except in the very north (Korhonen, 2006).

Karetnikov and Naumenko (2008) studied long-term (1943-2006) ice cover data from Lake Ladoga, north-west Russia, the largest dimictic lake in Europe. In this case the trend analysis revealed only minor changes in ice cover. In Lake Baikal, the ice-free season has lengthened and the ice thickness decreased in the last century (Hampton et al., 2008; Shimaraev et al., 2002).

There are very few studies on historical trends in river ice-cover thickness, but there are reports on decreasing trends (Beltaos and Prowse, 2009). Even less reported is the trend in river ice jams, and the results point in different directions, underscoring the complexity of ice-jam processes (Beltaos and Prowse, 2009). However, there is some evidence of a reduction in ice-jam floods in Europe due to reduced freshwater freezing during the last century (Svensson et al., 2006).

Projections

Future increases in air temperature associated with climate change are likely to result in generally shorter periods of ice cover on lakes and rivers. The most rapid decrease in the duration of ice cover will occur in the temperate region where the ice season is already short or only occurs in cold winters (Weyhenmeyer et al., 2004). As a result, some of the lakes that now freeze in winter and that mix from top to bottom during two mixing periods each year (dimictic lakes) will potentially change into monomictic (mixing only once) lakes with no ice-cover during winter and a long stagnation period from early spring till late autumn. This has consequences for deep-water oxygenation, nutrient cycling and algal productivity, and may change the ecological status of previously ice-covered lakes in temperate regions.

By the end of the century, the ice cover is predicted to shorten dramatically in lake Baikal (Shimaraev et al., 2002; Todd and Mackay, 2003). Climate change may also change the ice quality, but it is difficult to predict the effects; a shift towards more rain rather than snow in the spring may give more cloudy ice if the rain falls on snow, but a great deal of rain produces clear ice (Moore et al., 2009).

Borsch et al. (2001) calculated the expected changes in dates of freeze-up and break-up of river ice for regions in Russia based on simple correlation with air temperature. Although rough, the calculation shows that the largest changes will occur in the most westerly parts of Russia.

Warmer autumns and higher flows would give the largest potential for river freeze-up jamming. The likelihood of break-up jamming depends both on the nature of the established ice cover and the climatic conditions during break-up. A very important factor is the snowpack in the catchment, as more intense snowmelt increases the probability of a mechanical breakup (Beltaos and Prowse, 2009). According to the IPCC (Meehl et al., 2007) winter precipitation in high-latitude regions will increase, but the corresponding effect on snowpack will depend on the present and future temperature, determining whether the additional precipitation will fall mainly as rain or snow. At present, more research is needed to be able to predict the trends in spring ice jams with a changing climate (Beltaos and Prowse, 2009).

Mid-winter melts are likely to become more frequent in a warmer climate. Intensified mid-winter thaws would enhance the severity of mid-winter breakups, but may also reduce the potential for spring jamming. More importantly, rivers that currently have a permanent ice cover will be susceptible to mid-winter ice-breakups, which may have severe consequences (Beltaos and Prowse, 2009).

Conclusion

A shortening of the ice cover season has been observed, and continued shortening is expected. The duration of ice cover in the northern hemisphere has shortened at a mean rate of 12 days per century, resulting from an average 5.8 days later ice cover and 6.5 days earlier ice break-up. The ice cover of lakes in the temperate region where the ice season is already short or ice cover only occurs in cold winters is much more affected by temperature change than lakes in colder regions, such as Northern Scandinavia. The largest changes are therefore expected in the medium cold regions. A warmer climate will also give thinner ice and more mid-winter break-ups.

1.3. DOC concentration

Mechanisms

Climatic factors may affect both concentrations and fluxes of dissolved organic carbon (DOC). Climatic factors can affect production of DOC in the terrestrial systems and thereby changing DOC concentrations without changes in runoff. However, increased runoff may also affect DOC concentrations through changing the degree of dilution and by altering water flow paths through the soil (Inamdar et al., 2006; Soulsby et al., 2003; Worrall et al., 2002; Roulet and Moore 2006; Fig. 1.3). Changes in DOC fluxes can result from changes in runoff without changes in DOC concentration. Simultaneous increases or decreases in concentration may add on to or cancel out the flux changes, respectively.

DOC production is stimulated by increased temperature (Freeman et al., 2001a; Michalzik and Matzner, 1999). Increased soil moisture is also shown to have a positive effect on DOC production (Christ and David, 1996). Drought and soil frost may also stimulate DOC production through increased fine root and microbial mortality, increased fragmentation of soil organic matter and fresh litter and increased aggregate instability, giving direct release of DOC or increased microbial activity (e.g. Fitzhugh et al., 2001; Lundquist et al., 1999; Schimel et al., 2007; Tierney et al., 2001).

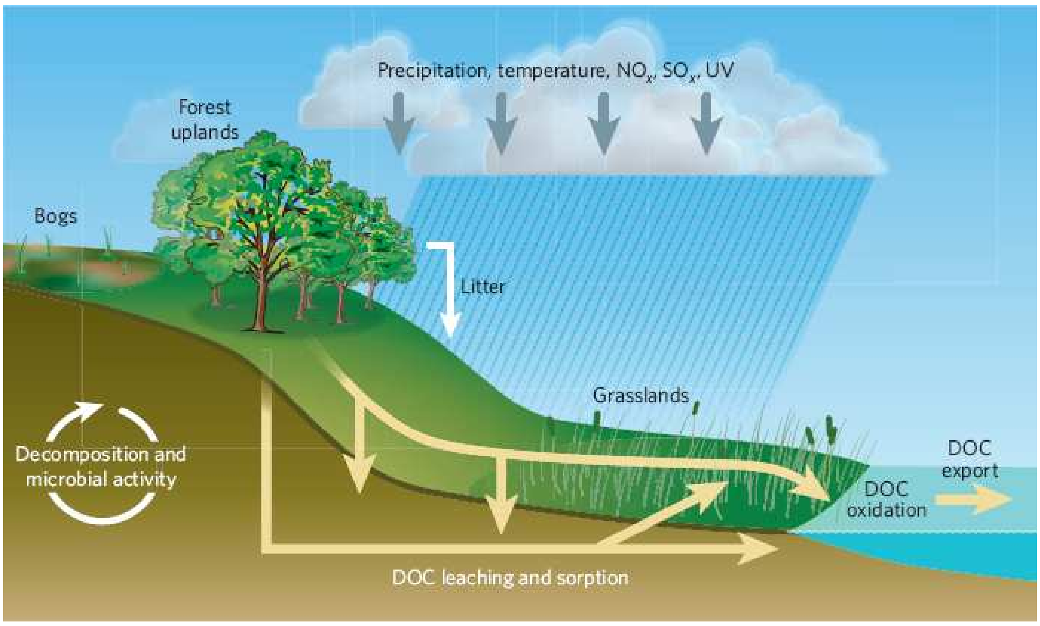


Figure 1.3 - Passage of dissolved organic carbon (DOC) through the landscape. The decomposition and subsequent leaching of organic litter in bogs, forests and wetlands are the principal sources of DOC in the terrestrial landscape. Production is mediated by several physical and biogeochemical factors, such as deposition of nitrates and sulphates from the atmosphere, moisture and temperature. The rate of export of terrestrial DOC is determined by the rate of production combined with the rate of sorption by mineral soils, and the availability of pathways for water through the

landscape. Evans et al. (2006) correlate an increase in hydrological DOC concentrations in the United Kingdom during 1988–2003 with the decreased deposition of sulphates in the form of acid rain in that period (Roulet and Moore, 2006).

In recent years an increasing trend in DOC concentrations has been observed (see Fig. 1.4)). However, there has been considerable debate on the main drivers of the observed increases, including the role of climate in these trends (e.g. Evans et al., 2005; Freeman et al., 2001a; Freeman et al., 2004; Monteith et al., 2007; Tranvik and Jansson, 2002). Freeman et al. (2001b) initially suggested that the increases could be due to the effect of temperature increase on soil enzyme activity. In reply, Tranvik and Jansson (2002) pointed out that the substantial increase in DOC concentrations in lakes and streams in Sweden during the 1970s and 1980s occurred despite a reduction in annual temperatures. Freeman et al. (2004) subsequently suggested that carbon dioxide mediated stimulation was responsible for the observed increases. However, Evans et al. (2005) argued that given the magnitude of the observed increase in CO_2, this mechanism can only account for 1–6% of the DOC increase at UK sites. Their study came to a tentative conclusion that deposition-related and climate-related factors both appeared to be significant. The increase in DOC concentration due to reduction in acid deposition is likely to be related to the effect of pH and ionic strength on soil organic matter solubility (de Wit et al., 2007; Evans et al., 2005). Statistical analysis of large datasets has later confirmed the strong relationship between the rise in DOC concentrations and the reduction in acid deposition (de Wit et al., 2007; Evans et al., 2006; Monteith et al., 2007). In a study of 522 spatially distributed data sets, Monteith et al. (2007) showed that changes in atmospheric deposition provided the only regionally consistent explanation for the observed increasing trend in lake and river DOC concentrations. However, the discussion is still ongoing, and several authors point to the importance also of climatic variables. It is evident that climatic factors can control concentrations and fluxes of DOC, but the question is to what extent. It is most likely that the trends so far have been governed by changes in acid deposition, but this does not exclude climatic factors to be important controls in the future (e.g. Weyhenmeyer, 2008). Erlandsson et al. (2008) showed that flow and SO_4 concentrations best explained the variability in DOC concentrations. They concluded that as SO_4 concentrations are stabilising, hydrology is likely to be the major driver of future variability and trends in DOC concentrations.

Past trends

In recent decades increased DOC concentration and water colour have been reported from many catchments in Europe and North America (Freeman et al., 2001a; Skjelkvåle et al., 2001; Löfgren et al., 2003; Hongve et al., 2004; Evans et al., 2005, 2006; Worrall et al., 2005; Vuorenmaa et al., 2006; de Wit et al., 2007; Erlandsson et al., 2008; Monteith et al., 2007). In the period 1988-2003 significant increases in DOC concentrations occurred at all 11 lakes (average increase 63%) and 11 streams (average increase 71%) included in the Acid Waters Monitoring Network in the UK (Evans et al., 2006) and increased significantly at 77% of 198 sites across the UK, none of which showed significant decreases (Worrall et al., 2005). In a study of long-term (1990-2004) trends in DOC concentrations for 522 data sets, covering six North European and North American countries, upward slopes outnumbered downward slopes. 88% of significant trends were positive. Upward slopes were most frequently significant in the southern Nordic region, the UK, and in the north-east USA (Fig.1.4).Long-term data sets have also shown increasing trends in DOC flux (de Wit et al., 2007; Worrall et al., 2008).

Projections

Due to the complexity of the biogeochemical processes, there are still large uncertainties in the modelling of future DOC concentrations and fluxes. The most advanced model including catchment and in-stream processes so far is probably the INCA-C model (Futter et al., 2007). The modelling suggests that a warmer and wetter climate could lead to higher DOC concentrations (Futter et al., 2007). Calculations by Erlandsson et al. (2008), based on the statistical relationship between DOC concentrations and flow, indicates that future changes are far smaller than the short term fluctuations observed. Hence, the short term fluctuations (hours to years) are likely to be of most concern for the water quality in the boreal zone. The modelling studies are mainly focusing on catchment and stream processes. It should be noted that changes in lake processes may also affect DOC concentrations, i.e. enhanced lake stratification due to higher temperature could lead to more photo-degradation of DOC (Epp et al., 2007).

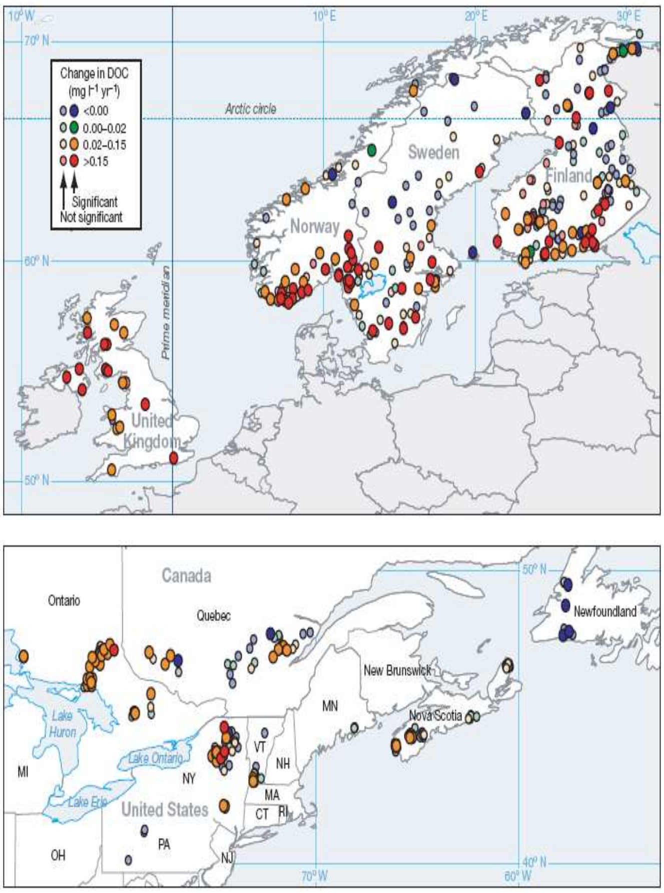


Figure 1 | Trends in dissolved organic carbon (mg l^{-1} yr^{-1}). Data are shown for monitoring sites on acid-sensitive terrain in Europe (upper panel) and North America (lower panel) for the period 1990–2004.

Figure 1.4 - Trends in dissolved organic carbon (mg l^{-1}y^{-1}). Data are shown for monitoring sites on acid-sensitive terrain in Europe (upper panel) and North America (lower panel) for the period 1990-2004 (Monteith et al., 2007).

Future DOC fluxes are likely to be mainly affected by the changes in precipitation levels. In a study of long-term DOC trends in peat-dominated catchments, Worrall et al. (2008) showed that the major control upon DOC flux from the catchments is the amount of runoff.

Conclusion

The increased DOC concentrations observed in lakes and rivers over the last couple of decades is probably mainly due to reduced acid deposition. However, both DOC concentrations and fluxes are affected by climatic factors, especially changes in precipitation. Increased precipitation is likely to give increased DOC fluxes, and may also give increased DOC concentrations. This will reduce light penetration and oxygen levels. Droughts may give lower lake DOC concentrations due to prolonged retention time and enhanced lake stratification, giving more photo-degradation of DOC.

1.4. Nutrient concentrations and loading

Mechanisms

A warmer climate can enhance the pollution load of nutrients to lakes and rivers. Changes in precipitation and resulting discharge patterns will also affect nutrient concentrations. Decreased summer flow will give less dilution of nutrient inputs and higher nutrient concentrations (Whitehead et al., 2009). Higher intensity and frequency of floods and more frequent extreme precipitation events will give increased surface runoff and erosion, increasing the nutrient load to the surface water (Jeppesen et al., 2009). Heavy rainfall may account for a significant proportion of annual phosphorus transfer from agricultural soils under arable crops (Fraser, 1999).

Also internal nutrient loading can increase with rising temperature. Higher temperature increases the release of phosphorus from bottom sediments (Feuchtmayr et al., 2009). The alternation of mixing and long stratification events make polymictic lakes especially vulnerable to this effect. This is because the mixing prevents a build up of high phosphorus gradients at the sediment-water interface which hamper further phosphorus release (Marsden, 1989), and because higher hypolimnetic temperature enhance mineralisation of organic matter and phosphorus release from the sediment (Wilhelm and Adrian, 2008). However, higher thermal stability due to increased lake temperature in deep lakes will delay mixing of nutrients (DeStasio et al., 1996). On the other hand, higher wind speed could enhance the mixing of nutrients (George et al., 2007).

A shorter ice-cover period can increase resuspension of phosphorus from the sediment owing to a longer mixing period, as seen in the Great Lakes (Nicholls, 1998). In contrast, earlier ice break-up in Lake Erken, Sweden, prolonged the P-limitation period for phytoplankton during the mixing period, but increased the nutrient availability in summer; probably owing to enhanced bacterial activity at higher water temperatures in combination with the extended mineralisation period (Blenckner et al., 2002).

For nitrate, extreme precipitation and snow melt events increases the nitrate load to surface waters as nitrate in deposition bypasses the biological sinks (Battarbee et al., 2008; Futter et al., 2009). Higher temperatures will increase mineralisation of soil organic matter. This is likely to be the effect which has the highest impact on nitrate leaching (Battarbee et al., 2008). Nitrate leaching has also been shown to increase both due to soil freezing (Davies et al., 2005; Monteith et al., 2000) and drought (Adamson et al., 1998) events, but the quantitative importance of these effects are still debated (Borken and Matzner, 2009; Matzner and Borken, 2008).

Past trends and projections

Because of difficulties in disentangling the effects of climatic factors and agricultural practices on nutrient dynamics, there is only a limited number of publications, which, based on long term observations, unequivocally demonstrate the impact of climate change on nutrient loads or concentrations in water bodies. On the contrary, there is a vast number of recent papers which describe modelling results of climate impact on nitrogen dynamics. Much less papers are dedicated to projections of phosphorus dynamics (Andersen et al., 2006).

Phosphorus loading is generally predicted to increase due to climate change potentially resulting in eutrophication problems in areas where precipitation and river flow are projected to increase (Moore, 2007). The seasonality is changed with a larger proportion of the external annual load coming in winter rather than in spring. Both increased agricultural run-off and sewage overflow are the sources for this increase. The internal phosphorus load is also projected to increase with climate change due to stronger water column stability in summer, longer summer stratification periods and decreased bottom water oxygen concentrations in stratified lakes (Wilhelm and Adrian, 2008).

Whitehead et al. (2009b) modelled (INCA) soluble reactive phosphorus (SRP) and nitrate concentrations in UK rivers. SRP concentrations were projected to decrease in winter and spring and increase in summer and autumn in rivers Lambourne and Lugg. The pattern was explained by dilution effects, corresponding to predicted changes in flow patterns (Fig. 1.5). River Tame is more affected by urban and agricultural runoff, giving a flush effect of increased winter flows, and a projected increase in SRP also in winter. Nitrate concentrations were predicted to increase slightly in winter and decrease in summer in rivers Tweed and Tamar. The winter increase may be related to flushing of upland soils. In summer increased temperature, reduced flow and increased residence time give enhanced denitrification and lower nitrate concentrations. However, further downstream, where the input from agriculture and point sources is larger, the effect of reduced dilution in summer becomes more important than that of increased denitrification, giving increased concentrations also in summer.

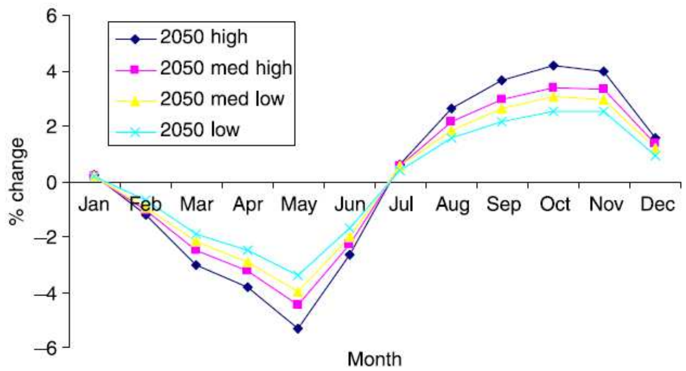


Figure 3 | Monthly (%) changes in phosphorus for the 2050s.

Figure 1.5 - Results for four different delta change scenarios for 2050 for SRP concentration in River Lambourne and River Lugg, UK (Whitehead et al., 2009b).

Futter et al. (2009) modelled (INCA) the combined effect of climate change and nitrogen deposition scenarios on nitrate concentrations in a Scottish mountain lake. The modelling projected a decline until 2030, larger under the maximum feasible reduction scenario for nitrogen deposition. From about 2030 the nitrate concentrations increase again. At this time the projected deposition levels are stable. The increase may be attributed to the small increase in temperature, stimulating soil microbial activity. Stimulated biomass production due to increased temperature may also contribute to nitrate leaching through increasing the organic nitrogen pool in the soil. The long-term trend indicates that climate change may offset the expected reductions in nitrogen emissions, but nitrate concentrations are predicted to remain lower than today, indicating that the main influence on long-term trends is the deposition.

Sjøeng et al. (2009) modelled (MAGIC) future nitrate leaching to surface water in an upland heathland catchment in southwestern Norway. The modelling was based on two scenarios of greenhouse gas emissions run with two global climate models. Estimates of future rates of nitrogen and carbon processes in the catchment were based on the downscaled temperature scenarios and two different storylines, one assuming changes only in soil processes due to future warming and nitrogen deposition (SLsoil), and the other assuming changes in both vegetation and soil processes (SLsoil + veg). Compared to the present, MAGIC simulated higher future NO_3 leaching for both storylines with much higher rates for SLsoil. The results suggest that differences between the two storylines were larger than differences between the different scenarios within each storyline.

Ducharne (2008) modelled (RIVERSTRAHLER) future nutrient concentrations in the river Seine. There was a positive climate change impact on ammonium concentrations if the point source input was high. The effect was shown to be mainly due to decreased dilution with decreased flow in summer. At low point source input, this effect was less important, and the climate change impact negligible.

Marshall and Randhir (2008) modelled (SWAT) the impact of temperature change only for the Connecticut River Watershed in the period 2060-2100. They estimated an average annual decrease in the load of organic phosphorus and nitrogen. The decrease was related to reduced annual runoff due to increased evapotranspiration, but also increased denitrification. The decrease in organic nitrogen compared to baseline (1960-2000) was 7 and 19 % for low and high warming, respectively. The corresponding numbers for organic phosphorus were 19 and 46%.

Conclusion

Increased rainfall can give both increased fluxes and concentrations of nutrients due to increased runoff and erosion, while summer droughts will give less dilution and higher nutrient concentrations, but also reduced annual run-off and thus reduced nutrient load. The net effect of climate change on nutrient load and concentrations depends on the hydrological conditions in the catchment, but is projected to increase in catchments with increased rainfall, also as a consequence of sewage overflow and increased surface run-off in agricultural areas. Increased temperature and lower oxygen levels in deep waters of lakes also give increased phosphorus release from sediments.

1.5. Water transparency and underwater light conditions

Mechanisms

Underwater light conditions are mainly affected by three factors: mineral turbidity, DOC concentration (colour) and phytoplankton density. Climate may change the underwater light conditions through changes in erosion, affecting both mineral turbidity and background input of nutrients, and through changes in the carbon cycling processes in the catchment affecting the input of humic substances. Moreover, if the stratification pattern of lakes are changed, the depth of the circulating water column will be affected and thereby also the mean underwater light for phytoplankton.

Past trends and projections

The amount of mineral turbidity may increase due to increased erosion caused by increased runoff. In a modelling study by Whitehead et al. (2009a), mineral turbidity was predicted to increase throughout the year, but particularly when increased flow follows a dry period. Turbidity may also increase due to increased wind-driven resuspension of sediments (Naumenko, 2008), but wind projections are uncertain (Moore et al., 2009).

The loading of DOC to surface waters may increase in a warmer and wetter climate (Whitehead et al., 2009a), but the projections for DOC concentration are uncertain (cf. DOC concentration section). Climate change may also have an opposing effect, as increased water temperature gives enhanced lake stratification and increased photo-oxidation of coloured dissolved organic matter (Epp et al., 2007).

For lakes and rivers in the colder regions, future trends in ice cover are also important, as the disappearance of ice cover increases underwater light availability in early spring and late autumn (Leppäranta et al., 2003). In Arctic regions, projected enhanced permafrost thawing is also likely to increase nutrient, sediment, and DOC loading to aquatic systems (Wrona et al., 2006), and thereby decrease underwater light.

A study of trends in Secchi depth in Lake Ladoga was conducted over the period 1905-2003 (regular measurements from 1955 only). The 7085 measurements collected in the ice-free period (May-October) were analysed for temporal trends on annual and monthly basis, and also on spatial basis, related to lake depth. Overall the results showed a significant linear trend, with a decrease in Secchi depth of 0.0073 m/year (Naumenko 2008; Fig. 1.6).

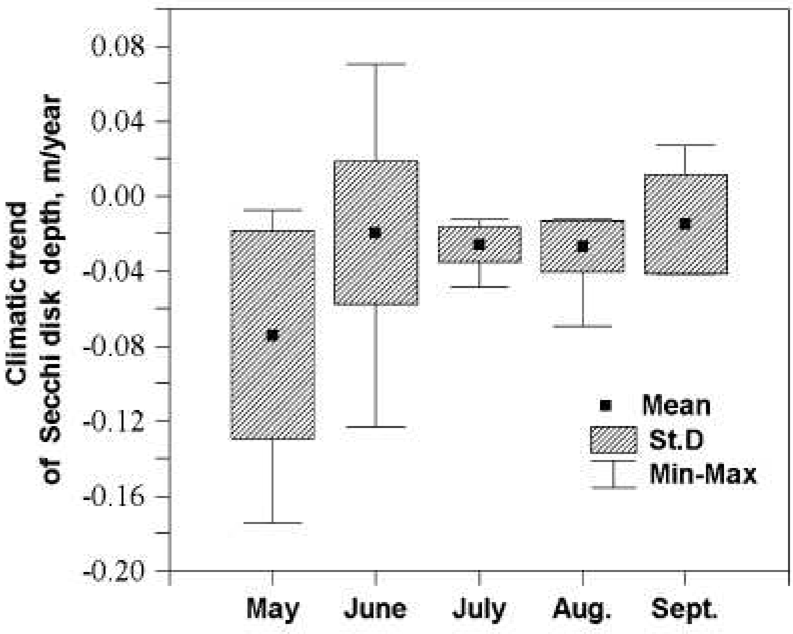


Figure 1.6 - Climatic (interannual) trend of Secchi disk transparency in Lake Ladoga from 1905 to 2003 (Naumenko, 2008).

Conclusion:

Where increased rainfall is predicted, transparency may be reduced due to increased erosion, as well as due to more nutrients and higher phytoplankton biomass in lakes. Potential increases in DOC concentrations will also decrease transparency, while reduced ice cover will dramatically increase light availability during early spring and late autumn.

1.6. Oxygen concentrations

Mechanisms

Dissolved oxygen concentrations decrease as a direct result of temperature increase, but also as an effect of increased respiration, either as a direct response to increased temperature or due to increased nutrient levels (Battarbee et al., 2008). This effect has been confirmed in mesocosm studies (Moss et al., 2003; Moss et al., 2004). The effect will be larger where the water warms faster, i.e. at shallow lake depth (Naumenko, 2008) and with summer low-flow periods in streams (Cox and Whitehead, 2009).

The effect of increased temperature is enhanced by the higher thermal stability, delaying vertical mixing of oxygen (DeStasio et al., 1996) which can lead to extreme hypolimnic oxygen depletion, and increases the risk of occurrence of deep-water anoxia (Jankowski et al., 2006). This may be particularly evident in polymictic shallow lakes. Here oxygen depletion is usually less severe than in dimictic lakes, but with increased temperature stratification events may occur more frequently, last longer and feature higher water temperature, increasing the severity of oxygen depletion (Wilhelm and Adrian, 2008).

Increased wind on the other hand, may increase mixing. At present, changes in wind dynamics cannot be projected, but wind has a strong impact on deep-water ventilation, as seen in lake Baikal (Moore et al., 2009). A reduction in wind speed or change of direction or timing could reduce the deep-water ventilation, while increased wind from the appropriate direction would give an enhancement. Finally, the presence of ice cover has a negative effect on dissolved oxygen concentrations (Karetnikov and Naumenko, 2008; Livingstone, 1993).

Past trends

Pan-European overview to show trends in oxygen concentrations in lakes and rivers are currently not available, but some case studies exist. One such case study is from Danish lakes, where some stratified, sheltered lakes with long retention time have declining oxygen concentrations (Fig.1.7). The total depth of the water layer with acceptable oxygen concentrations for fish is shrinking (Hansen et al., 2008).

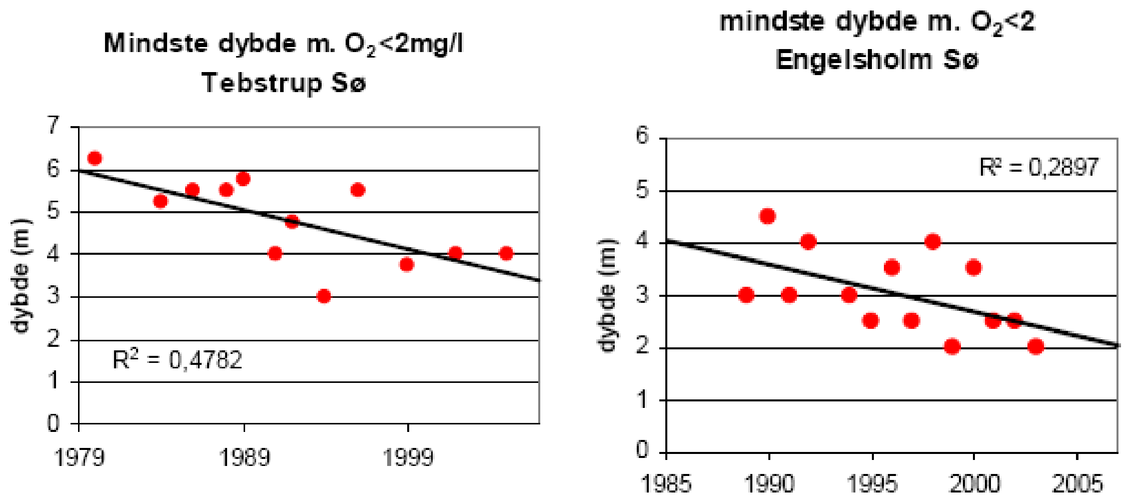


Figure 1.7 - Example of two Danish lakes with declining oxygen concentrations. The figures show the depth below which the oxygen concentration was lower than 2 mg l^{-1}. This concentration is the tolerance level for many fish species (Hansen et al., 2008).

Projections

Future climate projections indicate that longer and warmer growing season gives increased algal biomass and reduced hypolimnetic oxygen conditions (Battarbee et al., 2008). Modelling future dissolved oxygen concentrations in the Seine catchment, Ducharne et al. (2008) found reduced concentrations in a future climate, but the impact of climate change increased with the magnitude of point source inputs. With high point source input, the effect of discharge reduction in summer was important in addition to that of increased temperature, while at low point source input, only changes in temperature were important.

Cox and Whitehead (2009) modelled future dissolved oxygen concentrations in the River Thames. They showed that dissolved oxygen concentrations will decrease, and the changes are more significant with higher greenhouse gas emissions and in later periods. The authors conclude that the overall effect on water quality is unlikely to be dramatic, but may well be important. For instance, the changes were larger at the extremes, and more frequently approached or passed critical limits for fish survival. The authors also highlighted the importance of human activities, stating that the impact of future climate change will depend not only on climate change itself, but also on changes in water management practices.

Conclusion

Higher temperature is likely to give decreased dissolved oxygen concentrations in rivers and lakes, due to increased respiration and decreased solubility. Prolonged stratification in lakes will decrease the oxygen concentration in the deep waters of lakes. The effects are largely dependent on nutrient inputs. Future rainfall is also important, regulating the nutrient loads.

1.7. Impact on hazardous substances

Mechanisms, past trends and future projections

The research on the effect of climate change on concentration of hazardous substances is still limited (Barth et al., 2009). The effect will also to some extent depend on the type of hazardous substance in question. Most hazardous substances bind strongly to particles, and are thus likely to be affected by changes in hydrology. Direct temperature effects, on the other hand, will largely be limited to compounds which are volatile (organic pollutants, mercury) and/or subject to degradation processes (organic pollutants).

Climate change may affect future use of pesticides. It is not likely that changes in cropping patterns will affect the use of pesticides to a large extent (Bloomfield et al., 2006). However, increased prevalence of existing pests, weeds and diseases and increased pest resistance can lead to wider and more frequent application of pesticides, and introduction of new products (Bloomfield et al., 2006). On the other hand, in areas severely affected by drought the decline in agriculture will reduce the use of pesticides. Increased temperature will increase the rate of dispersal and give wider distribution of pesticides (Bloomfield et al., 2006).

Climate change will also affect remobilisation of previously released hazardous substances persisting in soil and sediment (Whitehead et al., 2009). Increased frequency of intense rainfall events and floods will give increased soil and sediment erosion, increasing the pollutant concentration in aquatic systems (Battarbee et al., 2008; Bloomfield et al., 2006). More intense rainfall can also give more by-pass flow and more rapid movement of pesticides from agricultural soils to surface water. This will be enhanced by warmer, drier summers giving cracking of the soils (Bloomfield et al., 2006).

Enhanced precipitation and groundwater levels may give increased flow through organic-rich soil horizons, causing mobilisation of mercury and methyl-mercury. Increased soil moisture can also give increased anaerobic conditions and enhance generation of methyl mercury (Battarbee et al., 2008).

Loading of hazardous substances may increase due to sewage overflow and increased leaching of hazardous substances from urban surfaces resulting from increased rainfall. Leaching of heavy metals from old mining tailings may decrease due to dilution by cleaner sediment from hillslopes (Coulthard and Macklin, 2003). However, low flows in summer may cause significantly reduced dilution potential and could generally give increased concentrations of hazardous substances (Bloomfield et al., 2006).

In cold regions, soil warming and thawing of the permafrost is likely to augment the release of stored chemicals to surface waters (Moore et al., 2009).

Increased temperature leads to a higher extent of partitioning of POPs to the air compartment. Since the air compartment is the compartment having the highest degradation potential, an increase in the fraction partitioning to this compartment will result in lower overall persistence (Dalla Valle et al., 2007; Bloomfield et al., 2006. Higher temperatures will also increase volatilisation and degradation of pesticides in soil and surface water (Bloomfield et al., 2006).

In their review of climate change effects on pesticides, Bloomfield et al. (2006) conclude that the overall effect of climate change on pesticide fate and transport is difficult to predict. It is also likely to be very variable. This is due to the uncertainties in climate predictions, the complexity of the natural processes and because of the conflicting implications of the climate-sensitive processes. The authors also suggest that in the long term climate change driven changes in land use may be more important for pesticides in the environment than the direct climate impacts (Bloomfield et al., 2006).

Conclusion:

There is so far too little knowledge on the effects of climate change on hazardous substances. Higher temperature may give increased volatilisation and degradation of some substances. Increased frequency of intensive rainfall will give increased soil and sediment erosion and higher loading to surface water. Drier conditions will give less loading, but may give higher concentrations due to lack of dilution. The use of pesticides is likely to increase, but will be reduced in areas where agriculture is abandoned due to too high drought frequency.

1.8. Pathogenic microbes in water

Pathogens derive both from point sources and diffuse sources (Ferguson et al., 2003). Increased precipitation and higher frequency of intensive rainfall events may lead to increasing occurrence of sewage overflow from drains and treatment plants (Kistemann et al., 2002; Nie et al., 2009). This can give a higher load of pathogenic microbes to surface waters.

High intensity rainfall may also give increased input of pathogens from pastures. Loading of livestock wastes to grasslands generates a potential surface store of pathogens, which may be released through for instance precipitation events (Oliver et al., 2005). Storm events may contribute to many orders of magnitude increases in pathogen loading in watersheds (Ashboldt et al., 2002; Atherholt et al., 1998). Statistically significant correlations have been observed between rainfall and outbreaks of waterborne disease outbreaks in the US (Curriero et al., 2001). Increased discharge may also lead to resuspension of microbes adsorbed in sediments (Ferguson et al., 2003).

Effects of dry periods on water quality have not been adequately studied (Takahashi et al., 2001), although lower water availability clearly reduces dilution and may e.g. concentrate pathogens in surface water (Giovanni, 2004; Byrne et al., 2006). See also Chapter 4.

Conclusion:

In regions which will receive more intensive rainfall, there may be a higher load of pathogenic microbes due to sewage overflow and runoff from agricultural land with animal manure. Drier conditions may give accumulation of pathogens, but also less input from the catchment. The net effect of droughts is therefore more complicated to predict and depends on local land use and hydrology, as well as urban waste water treatment efficiency.

2. Impacts on Freshwater Biodiversity

2.1. Introduction

Species and habitat dynamics in the face of climate change are complex and have many aspects. Increased temperatures and CO_2 concentrations will have an effect on different processes such as photosynthesis, respiration and decomposition and generally speed up these processes. Climate-induced changes in ice cover period, thermal stratification and nutrient availability and longer growing seasons affect species composition and food web structures.

Water temperature is one of the parameters that determine the overall health of aquatic ecosystems. Most aquatic organisms (e.g. salmonid fish) have a specific range of temperatures that they can tolerate, which determines their spatial distribution along a river or on a regional scale (Section 2.2). Climate change could lead to the extinction of some aquatic species or at least could modify their distribution in a river system or move their distribution northwards (Section 2.3). Several indications of climate impact on the functioning and biodiversity of freshwater ecosystems have already been observed, such as northward movement, phenology changes and invasive alien species (Section 2.4).

Enhanced harmful algal blooms in lakes resulting from climate change (Section 2.5) may counteract nutrient load reduction measures and also require a revision of classification systems for ecological status assessment. The inclusion of additional nutrient load reduction measures in river basin management plans may be needed to obtain good ecological status, as required by the Water Framework Directive. Public health may be threatened and the use of lakes for drinking water and recreation may be reduced.

2.2. Direct impacts on species distribution patterns

Mechanisms

The geographic distribution of aquatic organisms is partly controlled by temperature. Most aquatic organisms have (e.g. salmonids) have specific range of temperatures that they can tolerate, which determine their spatial distribution along a river or at a regional scale. Higher water temperatures therefore will lead to changes in distribution more northwards in Europe and to higher elevations. Interactions with other human induced hydromorphological or physico-chemical alterations to habitats further affect the geographical distribution patterns.

The timing of lake ice break-up is of critical ecological importance for lakes because the disappearance of the ice cover has a drastic effect on the underwater light climate, nutrient recycling and oxygen conditions, influencing for instance the production and biodiversity of phytoplankton and the occurrence of winter fish kills.

The projected world wide changes in parameters such as water temperature, light, nutrients availability and water fluctuations have the potential to affect species composition in different regions. These parameters acts as natural physiochemical filters and could become less effective in stopping invasive species (Battarbee et al. 2008; Rahel and Olden 2008; Fig. 2.1).

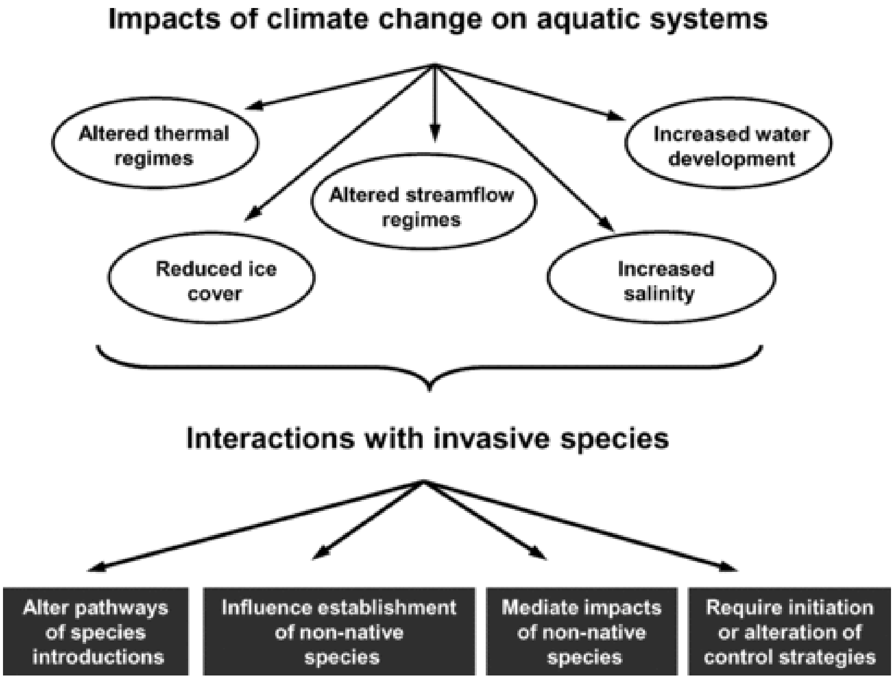
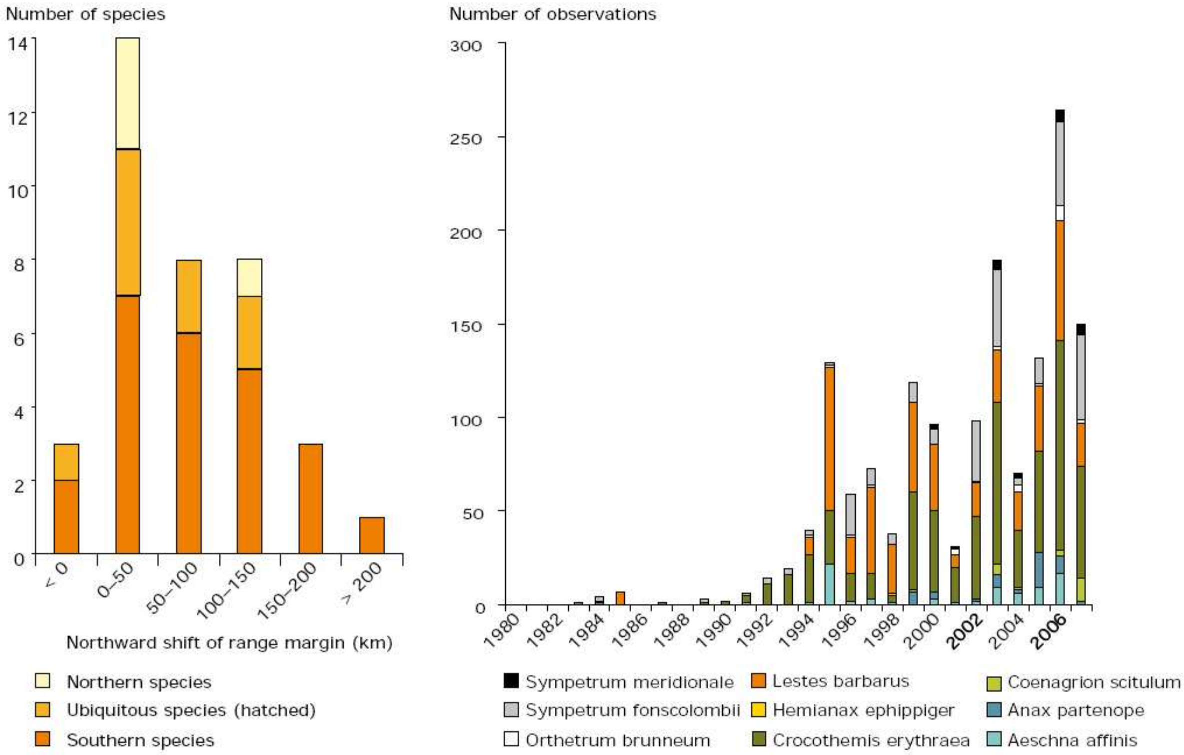


Figure2.1 - Characteristics of aquatic systems that will be altered by climatic change and how these will affect invasive species (Rahel and Olden, 2008).

Past trends and future projections

Many species are predicted to shift their ranges to higher latitudes and altitudes in response to climate warming.

- Hickling et al. (2005) presented evidence for 37 species of nonmigratory British dragonflies and damselflies shifting northwards at their range margins over the past 40 years, seemingly as a result of climate change. All but three species shifted northwards at their range margin (mean 74 km) between 1960–1970 and 1985–1995 (Fig. 2.2a).
- In Flanders the number of records of South-European dragonflies has markedly increased during the last 25 years. Some species such as Lestes barbarus, which were only occasional visitors in the past, now have permanent populations (Fig. 2.2b).



Note: Left: northward shift of range margins of British Odonata, dragonflies and damselflies, between 1960–1970 and 1985–1995. Right: observed occurrence of southern dragonflies in Belgium, 1980–2007.

Source: Hickling *et al.*, 2005 (left) and Biodiversity Indicators, 2006 (right).

Figure 2.2 - Northward shift and changes in occurrence of selected freshwater species

- The American slider turtle *Trachemys scripta* is a long lived species (at least 30 years) that has been introduced world-wide as pets. It is now considered a potential threat to freshwater ecosystems and import of *T. scripta* has been banned in the European Union. *T. scripta* has negative effects on the threatened European pond turtle *Emys orbicularis* through competition for food and basking places. In high numbers they may also modify wetlands and communities of macroinvertebrates and amphibians. Most populations of *T. scripta* are assumed to live in too cold habitats for a successful reproduction but will survive those conditions for decades. Ficetola et al. (2009) modeled projected climatic changes and found that by 2020 this species will increase its suitable breeding habitats dramatically. An A2 scenario would mean that 87 % where feral populations are currently present would be suitable for reproduction in 2020 (similar results for A1, B1 and B2 scenarios).
- Low water levels and a warm summer lead to eutrophication like effects in spite of decreasing external loads of nutrients. Visconti et al. (2008) studied a lake that had suffered from rapid eutrophication during the 1960s and 1970s and went back to oligotrophy during the 1980s and 1990s. In 2003, the warmest year of last century, both phytoplankton and zooplankton densities increased to values typical of those of the eutrophication phase. These values were strongly different from a normal year in the oligotrophic phase.
- A drier climate may reduce water flow and hence water depth in many streams and increase exposure of benthic invertebrates to ultraviolet radiation. Clements et al. (2008)

found through microcosm studies that benthic communities from a metal polluted stream were tolerant of the metal but more sensitive to UV-B exposure than invertebrates from a reference stream.

- Resting stages of eggs may be more prone to wind and animal dispersal in desiccated rock pools. Altermatt et al. (2008) studied metapopulation dynamics of the three planctonic Crustaceans (*Daphnia magna*, *D. longispina* and *D. pulex*) in small Finnish rock pools over a period of 24 years (1982-2006). These systems have high colonization and extinction rates. Colonization rates were much higher in summers with high temperatures and little precipitation.
- Warming may result in lower densities and diversities of aquatic hyphomycetes. According to Barlocher et al. (2008), the water temperature in a first-order stream increased by 4.3 ^{o}C, which is an approximate temperature increase projected within 2050 in Ontario, Canada. Their results show that an increase in temperature, by itself, may shorten the residence time of substrate available for aquatic hyphomycetes.
- Earlier spring warming may alter zooplankton dominance dynamics. Dupuis and Hann (2009) performed a laboratory experiment and further modeled a scenario with an earlier spring warming. Their results show a shift in the zooplankton community from larger daphniids to rotifer dominance. Hatching success from resting eggs of the dominant daphniid *D. ambigua* decreased by 50 % but the rotifers were not negatively affected. Rotifers use only temperature as a cue for hatching while *D. ambigue* use both temperature and photoperiod.
- Hering et al (2009) studied Trichoptera taxa potentially endangered by climate change in the European ecoregions. The study projected about 20 % of the Trichoptera species in most South European ecoregions and about 10 % in high mountain range to be potentially endangered For the Iberic-Macaronesia Region it is projected that 30.2 % of all species will be potentially endangered (endemic taxa meeting at least one sensitivity parameter that may lead to exclusion) (Fig. 2.3).

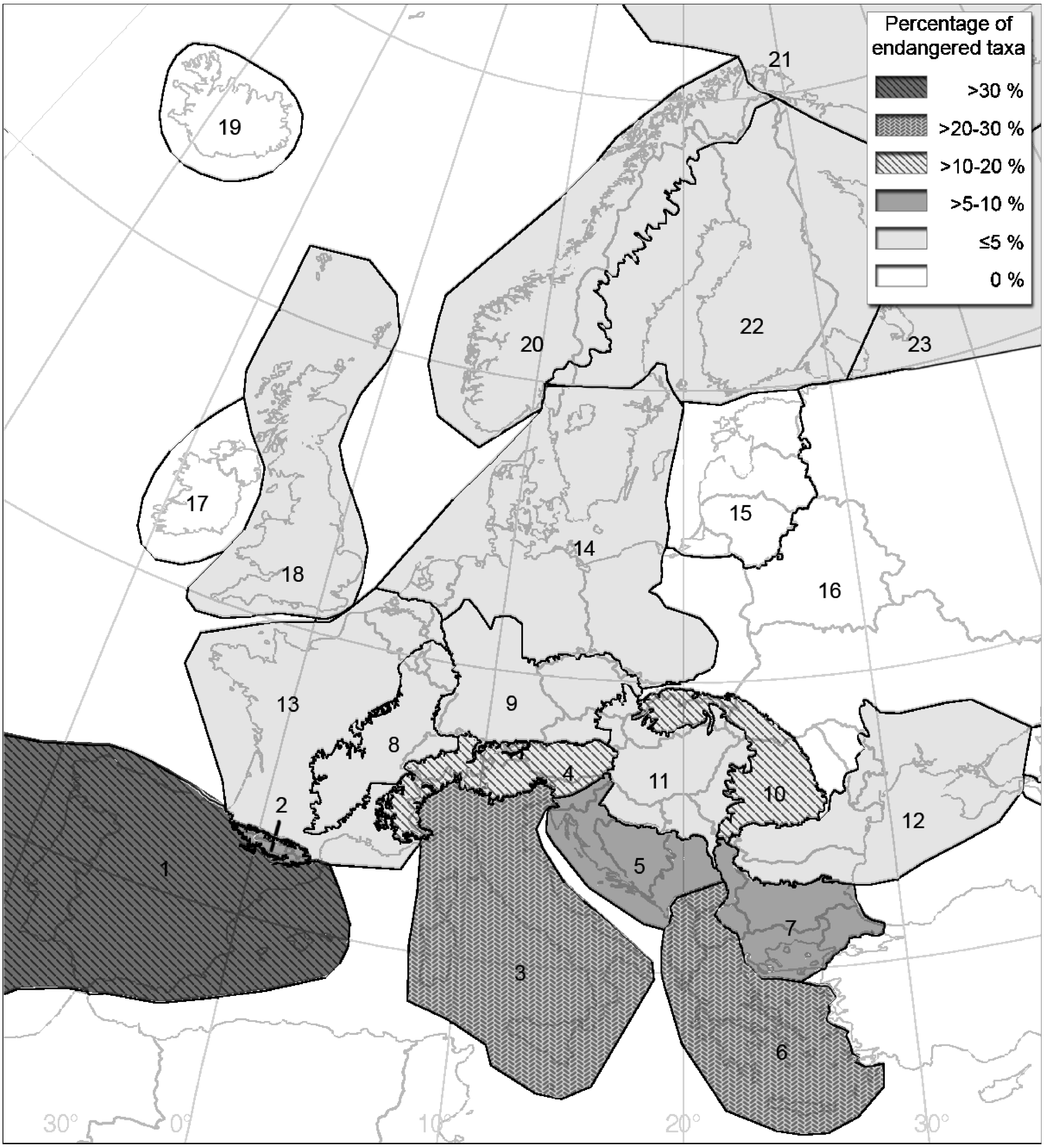


Figure 2.3 - Fraction of Trichoptera taxa potentially endangered by climate change in the European ecoregion (Hering et al., 2009).

Several indications of climate impact on freshwater ecology such as northward movement; phenology changes and invasive alien species thus have already been observed.

2.3. Extinction and reduction of cold-water species

Mechanisms

Most aquatic organisms are ectothermic and therefore temperature affects their physiology and biogeography (Rahel and Olden, 2008). Their dispersal is also limited within hydrographic networks (Buisson, Thuiller et al., 2008). Projected climate changes with increased temperatures world wide could lead to that cold water species may be excluded by warm water species and in worst case get extinct (Brown, Hannah et al., 2007).

Past trends and future projections

- Hari et al. (2006) has for brown trout in Alpine rivers shown, the warming during the last 25 years resulted in an upward shift in thermal habitat that was accelerated by an increase in the incidence of temperature-dependent Proliferative Kidney Disease at the habitat's lower boundary. Because physical barriers restrict longitudinal migration in mountain regions, an upward habitat shift in effect implies habitat reduction, suggesting the likelihood of an overall population decrease.
- Winfield et al. (2008) suggested that climate change has had a direct negative effect on the Arctic char (*Salvelinus alpinus*) living in Windermere, England, as a result of warming and eutrophication. Data from 1982-2006 show that the population of Arctic char had decreased and especially in the most eutrophic parts of the lake where oxygen levels in the hypolimnetic have been very low. At the same time there has been a marked increase in the abundance of warmwater roach (*Rutilus rutilus*). Arctic char is an important food source to the top predator Northern pike (Esox lucius) and the pike population in this lakes system has decreased.
- Buisson et al. (2008) assessed the future distribution of 30 common stream fishes in several French rivers and effects of changes in temperature and precipitation regimes. Overall fish species diversity was expected to increase but probably at the expense of cold-water adapted species. Cold-water adapted species living in headwaters would undergo deleterious effects whereas downstream species would have the possibility of migrating to new habitats.
- Burgmeer et al. (2007) analysed time series of entire communities of macrozoobenthos in lakes and streams in Northern Europe. There were no direct linear effects of temperature and climate indices on species composition and diversity, but using multivariate statistics they were able to show that trends in average temperature have already had profound impacts on species composition in lakes. These significant temperature signals on species composition were evident even though they analysed comparatively short time periods of 10–15 years. Future climate shifts may thus induce strong variance in community composition.
- Daufresne et al (2004) show that fish and invertebrate communities respond to increases in water temperature in the upper Rhône river in France. Fish abundance increased with water temperature, and thermophilic fish and invertebrate taxa replaced to a certain extent cold-water taxa. Daufresne et al. (2007) documented for the whole Rhône river (1985-2004) that changes in community structures of macroinvertebrates were related to high water temperatures and low oxygen content of the water.
- Brown et al. (2007) looked at data from the French Pyrènèes and found that a lower contribution of meltwater (from snow and glaciers) to the streams significantly increased

the diversity of macroinvertebrates. However some cold adapted taxa, like the Diamesa spp. (Diptera) and *Rhyacophila angelieri* (Trichoptera), decreased in abundance.

- Durance and Ormerod (2009) looked at macroinvertebrate data (family level) from 50 southern English streams in relation to water temperature, discharge and water quality over 18 years (1989-2007). They found that river temperatures are increasing year around but that this was insufficient to affect families of macroinvertebrates negatively. More variation was explained by the improving water quality through positive management than by increased winter temperatures.

2.4. Potential genetic adaptation / phenology changes

Mechanisms

Changes in growth season such as the ice free period or periods above a certain temperature will change the life cycle events such as earlier spring phytoplankton bloom, clear water phase, first day of flight and time of spawning of fish. Prolongation of the growing season can have major effect on species which as a result increase the number of cell divisions or generations per year.

Past trends

- Hassall et al. (2007) found that British Odonata have significantly advanced their phenology chronologically (on average by 1.75 days per decade) and with respect to temperature (on average by 3.37 days per one degree Celsius increase) over a 45-year period (1960–2004). This shift represents an extension to the preceding edge of the flight period (first quartile flight date) as opposed to a shift of the flight period as a whole.
- Dingemanse and Kalkman (2008) studied phenological data from 37 dragonfly species (Odonata) in the Netherlands 1995 through 2004. Flight period characteristics were investigated in relation to shifts in temperature. They found that the average Odonata species advanced its timing of flight by a mean of 8.8 days.
- In the Muggelsee, Germany Adrian et al. observed 25-30 days earlier occurrence of algae and Daphnids
- In the large Swedish lakes Vänern, Vättern and Mälaren there have for example been observed about one month earlier spring phytoplankton bloom in the 1990s compared to the 1970s and 1980s (Weyhenmeyer, 2001). In Lake Erken, Sweden, the spring peak of phytoplankton has come about one month earlier during the last 50 years (Weyhenmeyer, 1999; Noges et al., 2007)
- Increase in the length of growing season:
    - George and Hurley (2004) showed on the example of Lake Windermere (UK) that the slow growing late summer phytoplankton species, such as *Ceratium* and *Microcystis,* are particularly responsive to the extension of the growing season as one additional cell division in late summer can have a major effect on their maximum biomass.
    - Among animal species, the number of generations per year may increase with important implications on the balance in the trophic cascade. For example the *Baetis rhodani* (Ephemeroptera), one of the most common mayfly species in streams, has one generation in northern or higher altitude locations in Europe, a winter and a summer generation in Central Europe, and one winter and two summer generations in Southern Europe (Bauernfeind and Humpesch, 2001).

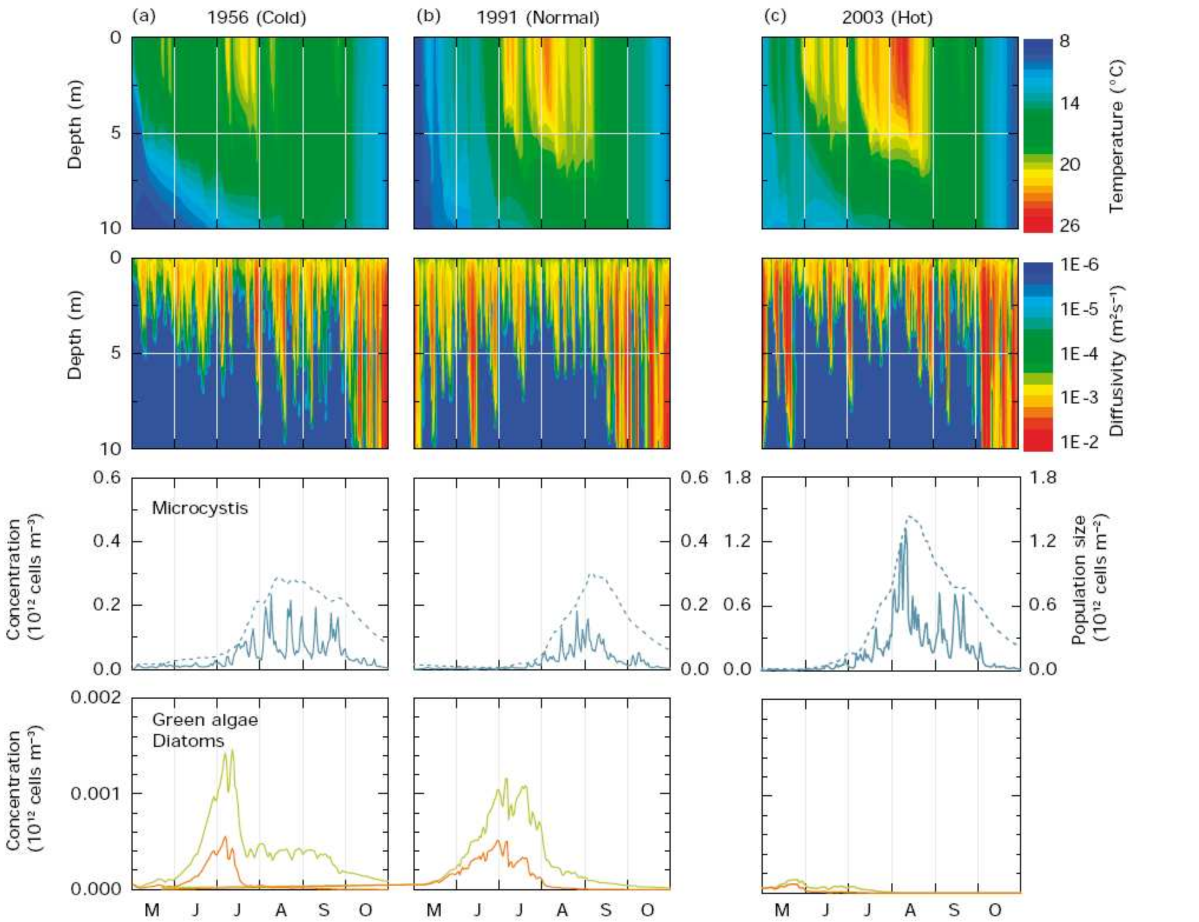
2.5. Risk of increased cyanobacterial blooms and other harmful algae

Mechanisms

Algal blooms in freshwater are predominantly cyanobacteria, some of which produce toxins. Cyanobacterial harmful algal blooms (CHAB) have been shown to increase worldwide for both natural forces and human activities. Climatic change with increased temperatures will allow temperature limited genera to expand both temporarily and spatially in subtropical regions (Hudnell, Dortch et al., 2008). Altered water runoffs and wind induced disturbance may also change population dynamics and potentially affect CHABs. The invasive Asiatic zebra mussel, *Dreissena polymorpha,* has been shown to filter feed on only non toxic phytoplankton thereby increasing CHABs.

Past trends and future projections

- The subtropical filamentous cyanobacterium *Cylindrospermopsis raciborskii* thrives in waters that have high temperatures, a stable water column and high nutrient concentrations: it has recently spread rapidly in temperate regions and is now commonly encountered throughout Europe (Dyble et al., 2002). The spreading to drinking and recreational water supplies has caused international public health concerns due to the potential production by this cyanobacterium of toxins.
- Climate change will generally favour and stabilize the dominance of cyanobacteria in phytoplankton communities, resulting in increased threat of harmful cyanobacteria and enhanced health risks, particularly in water bodies used for public water supply and bathing (Jöhnk et al. 2008; Mooij et al. 2005; Fig. 2.4).
- The increased occurrence of *mixotrophic* or motile algal species such as *Ceratium hirundinella* (Anneville et al., 2002) and metalimnetic maxima of the cyanobacterium *Planktothrix rubescens* (Walsby, 2005) are phenomena related to phosphorus depletion in the epilimnion. This depletion is caused both by loading reduction measures and intensified thermal stratification that isolates the upper mixed layer from the more nutrient rich bottom layers. Due to large body size, these algae are better protected from zooplankton grazing and may build up high standing stocks by the end of the stratification period.
- Shatwell et al. (2008) found that the timing of the phytoplankton peak was strongly dependent on the length of the ice cover while the peak of the cladocerans was dependent on ice cover length and photoperiod. This lead to an uncoupling in peaks, leaving a loophole between maxima of spring phytoplankton and zooplankton. Earlier and thus longer spring periods in years without ice-cover favor cold water adapted diatoms until water temperature increases or they get limited by silicon. This will leave more phosphate for filamentous cyanobacteria (Oscillatoriales) which may exploit the loophole and establish dominance. Therefore, warming and earlier springs may promote cold-adapted diatoms and filamentous cyanobacteria (Fig. 2.5).



Note: (a) the cold summer of 1956, (b) the average summer of 1991, and, (c) the hot summer of 2003. Top panels show the temperature contour plots. The second row shows contour plots of the turbulent diffusivities. The third row shows the surface concentrations of Microcystis (solid lines) and the depth-integrated population size of Microcystis (dashed lines). The fourth row shows the surface concentrations of diatoms (orange lines) and green algae (green lines). Note the difference in scale between the Microcystis concentrations (third row) and the concentrations of diatoms and green algae (fourth row).

Source: Jöhnk *et al.*, 2008.

Figure 2.4 - Model simulation of hydrodynamics and phytoplankton dynamics during three contrasting summers in Lake Nieuwe Meer (the Netherlands) (Jöhnk et al., 2008).

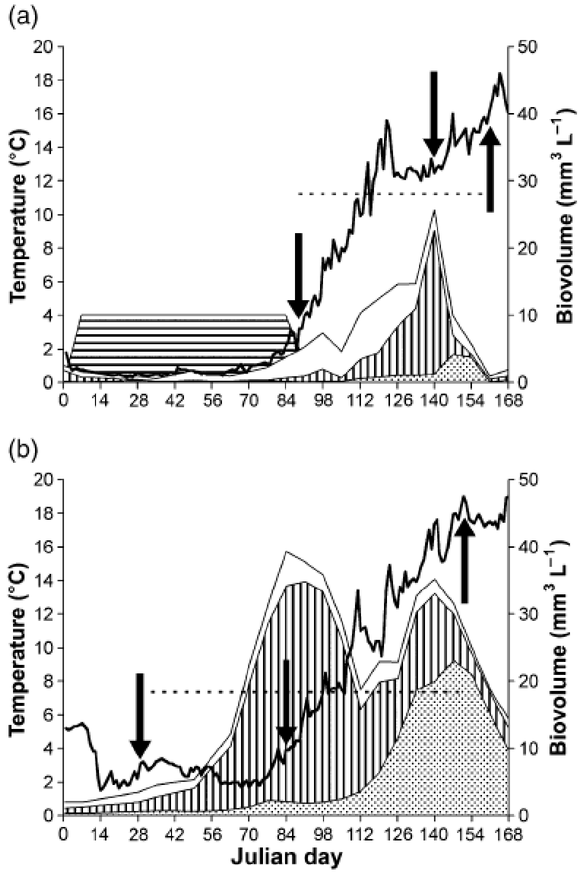


Fig. 1 Temperature and timing of the spring period in warm and cold years in Lake Müggelsee: (a) 1987, a cold year; solid line: water temperature; broken line: mean water temperature in the spring period; shaded areas: cumulative biovolumes of phytoplankton – dotted: cyanobacteria; vertical lines: diatoms; empty: other algae; horizontal lines: ice cover. Arrows from left to right: beginning of the spring period (when water reaches 3 °C); phytoplankton peak; end of the spring period (clear water phase or cladoceran peak); (b) 1988, a warm year without ice cover, legend same as for part (a).

Figure 2.5 - Phytoplankton development in Lake Müggelsee, Germany in a) a cold year (1987) and b) a warm year (1988) (Shatwell et al., 2008).

2.6. Conclusions

- Southern species will move further north and to higher altitudes. Species of colder regions will move north and to higher altitudes or will disappear when their migration is hampered (e.g. due to habitat fragmentation).
- Of all ecosystems, freshwater ecosystems will have the highest proportion of species threatened with extinction due to climate change (Millennium Ecosystem Assessment, 2005b, IPCC 2007).
- Climate warming will aggravate eutrophication effects with enhanced phytoplankton blooms, favouring and stabilising the dominance of harmful cyanobacteria in phytoplankton communities, resulting in increased threats to the ecological status of lakes and enhanced health risks, particularly in water bodies used for public water supply and bathing. This may counteract nutrient load reduction measures.
- There are European examples of changes in life cycle events (phenology) such as earlier spring phytoplankton bloom, appearance of clear-water phase, first day of flight and spawning of fish. In several European lakes, phytoplankton and zooplankton blooms are occurring one month earlier than 30–40 years ago.
- Effects of warming may interact with pollutants in the environment causing synergistic negative effects (Clements et al., 2008).

3. Impacts on ecological status of lakes and rivers

3.1. Introduction

The ecological status in lakes and rivers can be affected by climate change in a number of ways. The reference conditions can be affected, the pressures can be affected, and the thresholds often used to set the WFD target for different groups of freshwater biota can be affected through different mechanisms. These three main effects can also cause time lags for restoration /recovery after implementation of mitigation or adaptation measures to reduce the pressures. These effects are illustrated in Fig.3.1.

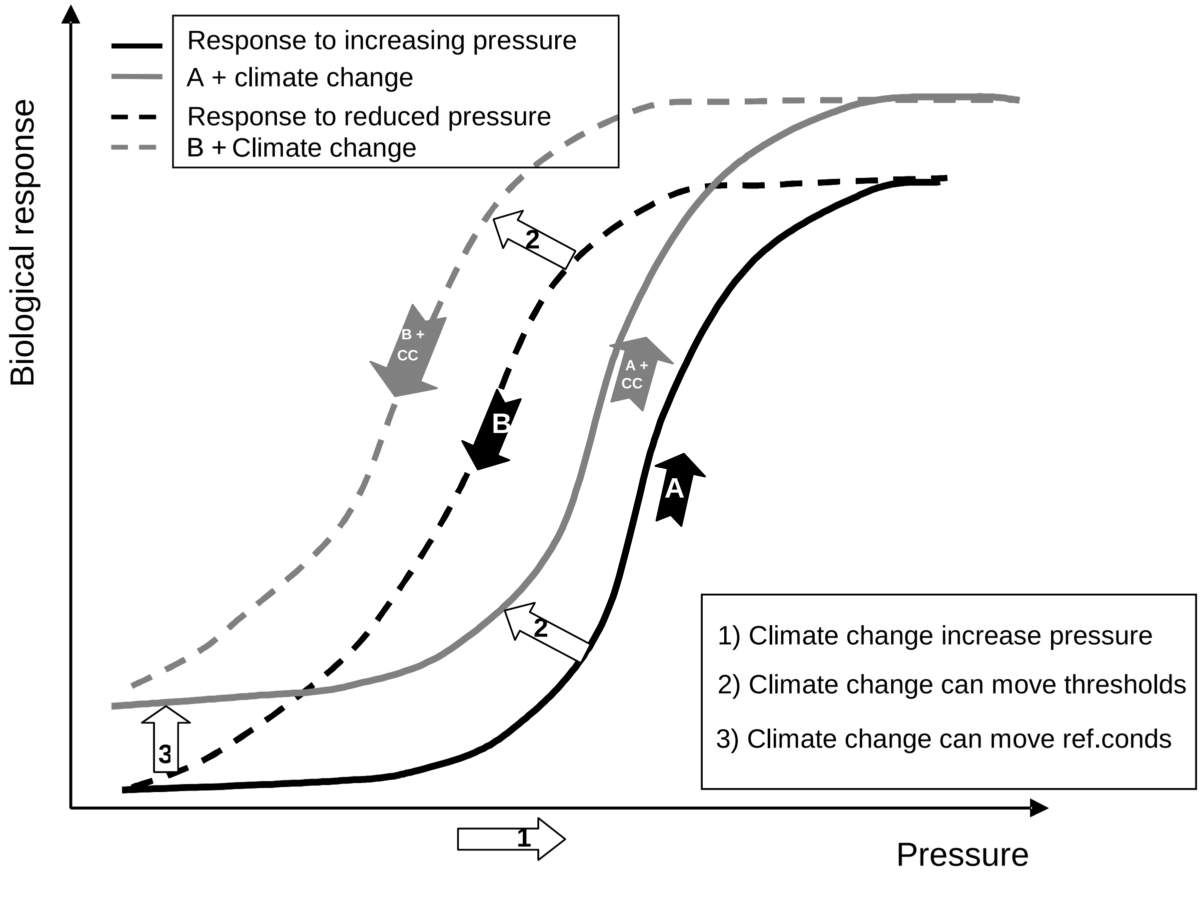


Figure 3.1 - Potential impacts of climate change on pressures, thresholds and reference conditions for a biological indicator that increases with pressure (such as tolerant taxa) (Thaulow and Lyche-Solheim, 2009).

The figure illustrates that climate change may cause degradation of water bodies to happen at lower pressure levels than before, and may also require more mitigation measures for the water bodies to recover. The recovery may also be incomplete if the reference conditions have changed.

While the impact of climate change on other pressures is outlined in Chapter 2, this chapter will focus more closely on the impact of climate change on reference conditions and on thresholds, since this is of major importance for water managers when making river basin management plans and programme of measures to reach the WFD target in European lakes and rivers.

3.2. Impacts on reference conditions

Mechanisms

Under climate change, reference conditions (sensu WFD) may change, as identified by the Euro-limpacs project (Battarbee et al., 2008). Reference conditions, as defined by a set of biological indicators for different quality elements (phytoplankton, benthic flora, benthic fauna, fish), may change because of changes in the abiotic conditions, such as increased water colour, increased temperature, changes in flow regimes and changes in the bottom substrate. Such changes may affect the biogeographic patterns, as well as the phenology of species. The abiotic changes may also cause the type of a water body to change, e.g. from clearwater to humic (Northern Europe), from non-stratified to stratified, from freshwater to saline, or even from a permanent to a temporary water body drying out in the summer (Southern Europe). In cases where flow regimes and temperature in small rivers become sufficiently altered, the biological indicators normally found under reference conditions can be endangered.

Past trends and future projections

Long-term trends from reference sites are difficult to find, since such dataseries are rare in Europe. One example is from the Norwegian reference lake Atnsjøen, that has been regularly monitored for all biological quality elements since 1990. The lake is situated at ca. 700 m altitude in a pristine catchment close to the mountain area of Rondane.

The results for phytoplankton (Fig. 3.2) do not yet show any trends in phytoplankton mean or maximum biomass (based on 5 samples per year), but this time-series is a good basis to analyse such trends in the years to come. If the biomass will exceed the high/good boundary for most years in the future, then this boundary should be revised.

The results for fish (Fig. 3.3) show that the biomass of char has become consistently lower than that of trout for all years after 2002, whereas this was not the case in the previous years, except for a couple of single years. This may be related to climate change, since the arctic char is more vulnerable to high temperatures than the brown trout.

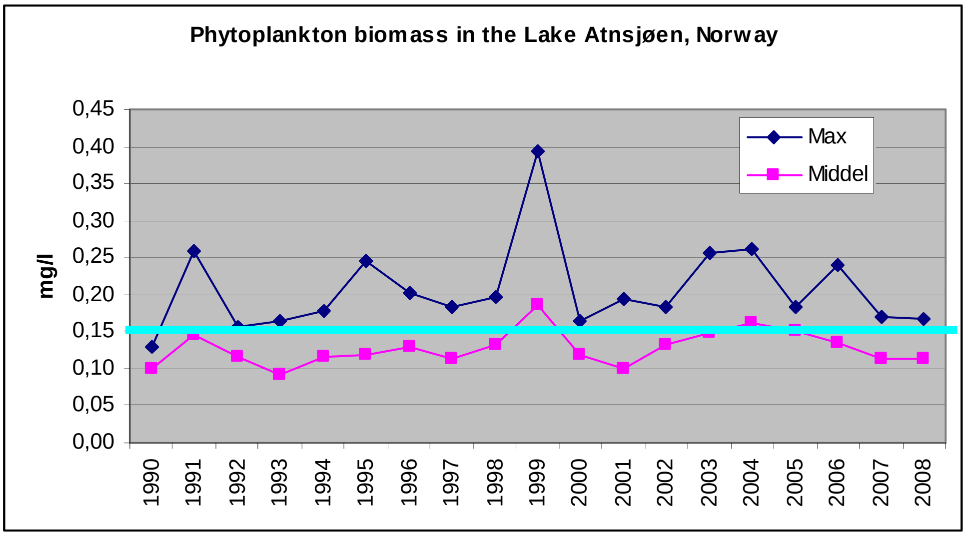


Figure 3.2 - Phytoplankton biomass in Lake Atnsjøen, Norway. Data from Pål Brettum, NIVA. Mean biomass is based on 5 samples taken monthly during the growing season (May-October) each year. The light blue line is the preliminary high/good class boundary for mean phytoplankton biomass for this lake type.

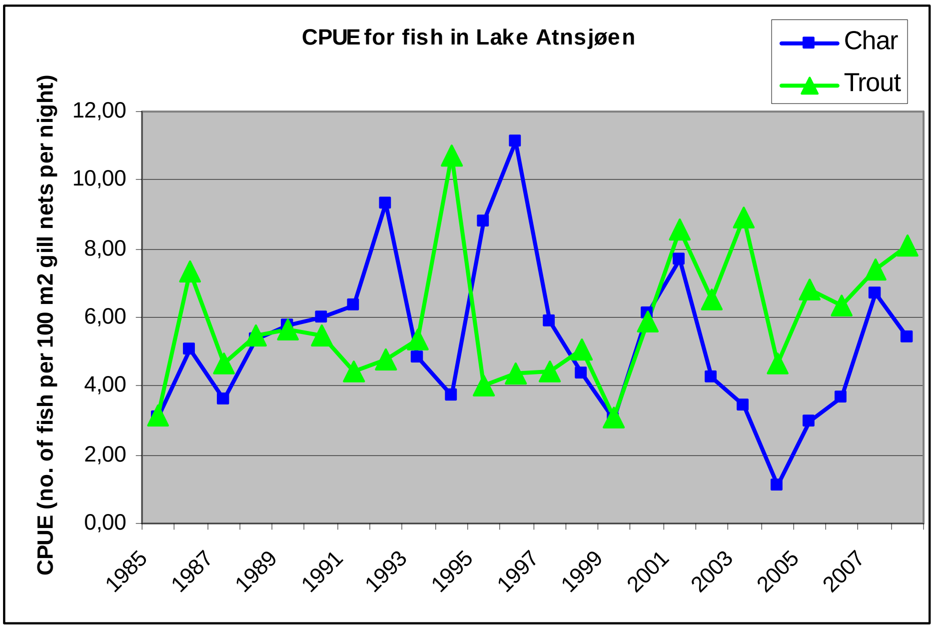


Figure 3.3 - Fish biomass in Lake Atnsjøen, Norway. Data from Trygve Hesthagen, NINA. Data shows catch per unit effort (CPUE) as number of fish per 100 m2 gill nets per night.

Studies from CEMAGREF (Lasalle and Rochard, 2008), done these last years, demonstrate the effect of increased temperature and salinity on the fish fauna in the Garonne/Gironde river in southern France. The water temperature has increased by 1 °C over the last 20 years in the Gironde Estuary. This increase of temperature has contributed to the disappearance of the fish "eperlan" in the estuary of the Gironde river and of the "flet" that have been replaced by "anchois" due to the decrease of fresh water in the Estuary. A decrease of the biomass has also been observed. At the same time, a lot of invasive species (fauna and flora) are appearing in the Garonne district having negative impacts on the local species (e.g. eperlan, anchois). The quality indicator used to set reference conditions is based on the endemic fishes and invertebrate species for the Garonne river, but this indicator may have to be modified in the years to come.

Future projections suggest that the reference values for indicators based on biomass will increase in most lake and river types for all biological quality elements, due to warmer water, longer growing season and increased background inputs of nutrients from the catchment. Concerning species composition it is likely that reference values for indicators based on sensitive taxa mostly will decrease (as in the Garonne river), since these taxa often are cold-water adapted, whereas the reference values for indicators based on tolerant taxa mostly will increase. Reference values based on species diversity (number of taxa) may increase or decrease depending on the type of water body (esp. location relative to altitude and latitude). Southern and lowland rivers and lakes may get lower number of taxa, whereas northern and highland rivers and lakes may get higher number of taxa. This is due to the unimodal (bell-shaped) relationship of species number with temperature: most species are present at intermediate temperatures (Moss et al., 2009).

Monitoring of reference sites will be essential to understand and appropriately respond to the potential impact of climate change on reference conditions of water bodies across Europe.

3.3. Potential impacts on thresholds for the good/moderate boundary

Mechanisms

A critical concern in the management of freshwater ecosystems is the attempt to prevent water bodies from crossing key **thresholds**, where systems may change abruptly and involve a switch to regimes that are difficult to restore (cf. Andersen et al., 2009). If thresholds are moving to lower parts of the pressure gradients, this is a serious challenge for the programme of measures. Such moving can be anticipated from direct and indirect effects of higher temperature and hydrological changes on the biological quality elements. For example, the tolerance level of cold-water species of fish and benthic fauna to eutrophication/organic load and acidification pressure may decrease due to lower oxygen concentration in warmer water. Likewise, sensitive macrophytes may have lower tolerance limits to eutrophication if the underwater light climate is reduced due to increased mineral turbidity or humic substances. With reduced underwater light, tolerant phytoplankton taxa, such as the shade-adapted bluegreens, may dominate at lower pressure levels than before. In Southern Europe, reduced water levels in rivers and lakes will probably decrease the tolerance levels to other pressures.

Past trends and future projections

Although thresholds and reference conditions for some systems and pressures have been well studied e.g. water bodies suffering from eutrophication or acidification (Lyche-Solheim et al., 2008), there is much less empirical evidence about how climate change may cause thresholds to be crossed, or may cause thresholds to move (Fig. 3.1).

One example of a climate induced crossing of the threshold for chlorophyll in deep Alpine lakes is seen in Lake Constance, where the chlorophyll threshold or WFD target was exceeded in a year with insufficient spring circulation due to rapid warming of the surface layer (Straile et al., 2003; Fig. 3.4). Such situations may become more common with climate change and will pose special challenges for water managers, since additional nutrient reduction measures will be needed to counteract this effect.

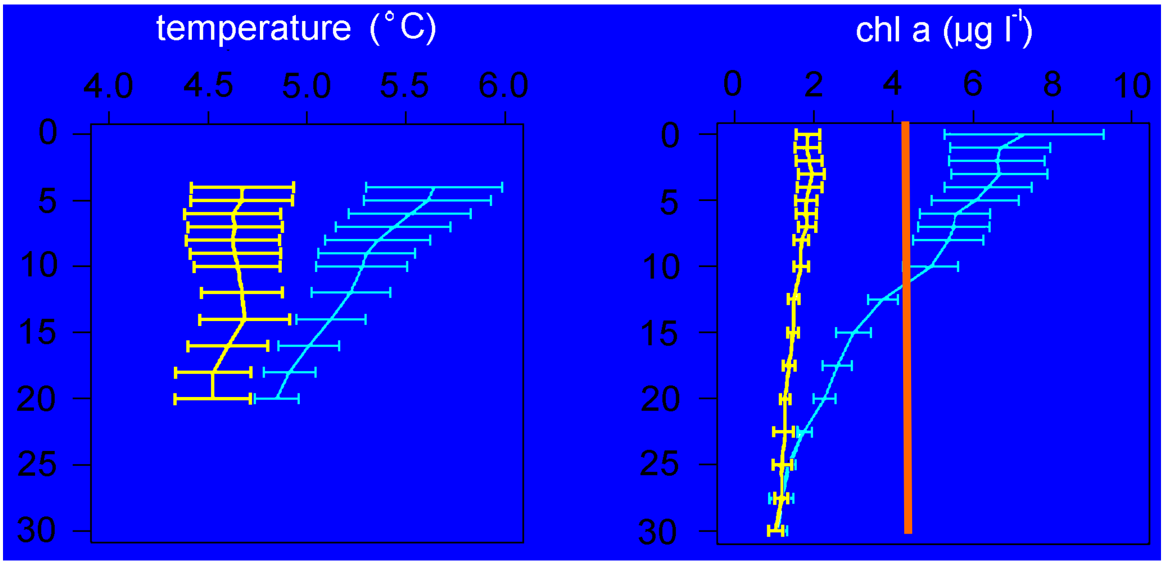


Figure 3.4 - Temperature (left) and chlorophyll (right) profiles in Lake Constance from Straile et al., 2003. Y-axis is depth in m. Yellow lines and bars represent years with normal spring mixing, and light blue lines and bars are years without complete spring circulation. The orange vertical line is the WFD target for this lake type.

Although some European freshwaters are now recovering from pollution, all face pressure from climate change directly or indirectly. The most vulnerable systems are those already close to critical thresholds, for example those that contain endemic taxa that are trapped geographically, or those most susceptible to a decrease in oxygen concentrations such as eutrophic lakes with small hypolimnia.

3.4. Projections for major ecological changes in rivers and lakes

A recent paper by Moss et al., 2009 summarizes the impacts of climate change on different types of rivers and lakes. Table 3.1 extracts the most essential information from this paper. The most common effects will be:

- increased algal blooms in lakes,
- increased biomass of benthic algae in rivers,
- loss of cold-water species, such as salmonid and coregonid fish, and other sensitive taxa,
- increased dominance of pollution tolerant taxa in all biological quality elements,
- increased invasions of exotic species.

Table 3.1 - Major impacts of climate change on ecological status in different types of lakes and rivers (modified from Moss et al., 2009).

Region	Lake or river type	Climate change impacts on biological quality elements **Most effects are caused by increased temperatures, changes in river flow/water level/floods/droughts and increased risks of eutrophication and hydromorphological alterations.**
Northern	Fastflowing upland rivers	Reduction of cold-water species: salmonids and coregonids, increased biomass of benthic algae.
	Slowflowing lowland rivers	Reduction of cold-water species: salmonids and coregonids.
	Shallow lowland lakes	Reduction of cold-water species: salmonids and coregonids, increased biomass of phytoplankton, reduction of sensitive macrophytes. Invasions by exotic species.More algal blooms.
	Deep lakes	Reduction of cold-water species: salmonids and coregonids, increased biomass of phytoplankton and more harmful algal blooms, invasions by exotic species.
Central	Fastflowing upland rivers	Replacement of cool-water species with warmer water species. Increased biomass of benthic algae and tolerant taxa. Invasion of exotic species.
	Slowflowing lowland rivers	Loss of many fish species. Fish kills due to deoxygentaion. Loss of sensitive benthic fauna species (deoxygenation). Invasion of exotic species.
	Shallow lakes	Greater, but less diverse plant growth, risk of summer fish kills, esp. piscivores. Dominance of cyprinid fish. Invasion of undesirable exotics.
	Deep lakes	Almost complete loss of salmonid and coregonid fish, total dominance of cyprinid fish, increased biomass of phytoplankton and more harmful algal blooms, invasions by exotic species.
Southern	Fastflowing upland rivers	Major loss of biodiversity for all quality elements due to deoxygenation and salinisation, dominance by pollution tolerant taxa and exotic species.
	Slowflowing lowland rivers	Fish kills. Loss of biodiversity for most quality elements. Severe invasions of exotic species.
	Shallow lakes	Loss of many species, more harmful algal blooms, major loss of endemic fish and amphibians.
	Deep lakes/reservoirs	Loss of littoral zone due to severe water level drawdown in reservoirs, more intense and frequent algal blooms.

The effects are caused by:

- increased temperature stress,
- changes in river flow/water level/floods/droughts,
- increased erosion in upland rivers in the northern and central European regions
- increased risks of eutrophication and deoxygentation (see Chapter 2),

- hydromorphological alterations (flood protection, urbanisation, new transport infrastructure, irrigation and hydropower),
- salinisation primarily in the southern region.

Evolutionary adaptations in cold-water species may compensate for some of these effects, but will most likely be too slow relative to the speed of climate change to prevent ecological damage. The selection pressure will also favour warm-water species, thereby making the situation even worse for cold-water species.

The sum of all these effects will be a deterioration of ecological status in most lake and river types.

4. Impacts on human health and socio-economic costs[1]

4.1. Mechanisms

Access to safe water remains an extremely important global health issue. The risk of outbreaks of water-borne diseases increases where standards of water, sanitation and personal hygiene are low. Extreme precipitation events leading to floods or droughts can have direct and indirect health effects. Flooding can cause diarrhoeal diseases, vector-borne diseases, respiratory infections, skin and eye infections. Floods also have other effects with health consequences: damage to infrastructure for health care and water and sanitation. Droughts or extended dry spells can impair provision of safe water leading to water-related health problems, for example through reducing the volumes of river flow, which may increase the concentration of effluent pathogens, posing a problem for the clearance capacity of treatment plants. Intestinal infectious diseases that are transmitted through water are sensitive to climate and weather factors. Such diseases are the main causes of infectious diarrhoea and cause significant amounts of illness each year in Europe.

Increased surface runoff after extreme precipitation events can flush pathogenic bacteria, other contaminants and nutrients into surface waters and thus influence bathing water quality negatively, and also increase the need for more thorough purification of raw water in drinking water supply systems based on surface waters.

Higher temperatures can result in a longer bathing season, but can also cause prolonged blooms of potentially toxic algae.

4.2. Past trends and projections

In Europe the risk of infectious disease outbreaks related to a climate change impact on microbial water quality is relatively small due to the standard of water treatment and distribution infrastructure. Nevertheless, the incidence of pathogens is increasing.

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[1] The first paragraphs in Sections 4.1 and 4.2 are extracted from Impacts of Europe's changing climate – 2008 indicator-based assessment, Chapter 5.10.4 (EEA, 2008)

Approximately 20 % of the population in Western Europe is affected by episodes of diarrhoea each year (van Pelt et al., 2003). Such infections have a significant economic impact in terms of treatment costs and loss of working time (Roberts et al., 2003).

Heavy precipitation has been linked to a number of drinking-water outbreaks of Cryptosporidium (a pathogen causing a diarrhoeal illness) in Europe, due to spores infiltrating drinking water reservoirs from springs and lakes and persisting in the water distribution system (Lake et al., 2005; Semenza and Nichols, 2007). In Germany, bacteriological and parasitic parameters spiked considerably during extreme runoff events (Kistemann et al., 2002). New pathogens have also emerged in recent years. Examples of an increased risk of infectious disease outbreaks have been found in the United Kingdom (Reacher et al., 2004), Finland (Miettinen et al., 2001), Czech Republic (Kříž et al., 1998) and Sweden (Lindgren, 2006). Key water-borne infections in Europe are monitored..

The European Union publishes every year a report on bathing water quality in the different member states. This report checks e.g. the compliance of water bodies with the new Bathing Water Directive of 2006 (EU, 2006). For inland water a positive trend towards more water bodies with a good bathing water quality is observable (Fig. 4.1). The Atlantic, North Sea, Baltic Sea and Black Sea regions do best compared to mandatory values, only the inland bathing areas of the Mediterranean fall below the European average in complying with mandatory values (EEA, 2009).

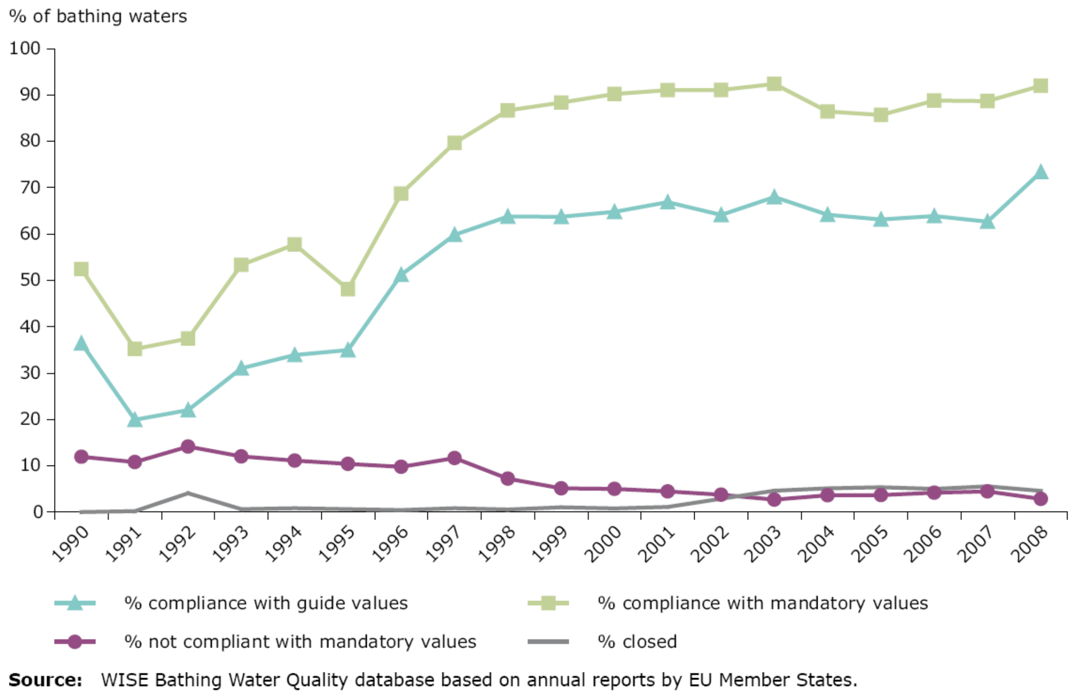


Figure 4.1 - Inland bathing water quality in Europe (EEA, 2009).

Predictions for future development of human health related disease vectors are highly uncertain. On one hand a further improvement of bathing water quality can be expected due to stricter implementation of environmental regulations. On the other hand the water quality in inland surface water bodies strongly depends on the future land use intensity and precipitation pattern.

A recent report on Climate change and Bathing water (Roijackers and Lürling, 2007) concludes that climate change in the Netherlands will result in favourable environments for a myriad of new pathogens and vectors, including invasions by new disease vectors to European surface waters from the (sub)tropics (Tab. 4.1). The same is probably largely true for Europe as a whole, with Central and Southern Europe being most vulnerable to these new vectors due to higher population densities and a warmer climate than Northern Europe.

Table 4.1 - Vector-borne pathogens transmitted by waterborne organisms (compiled from WHO, 1990; Chan et al., 1999): ++ = slightly increased risk, +++ = increased risk in The Netherlands (Roijackers and Lürling, 2007).

pathogen	disease	vector	presence	increased risk
Plasmodium spp.	malaria	*Anopheles* spp.	(sub)tropics	+++
dengue virus	dengue	*Aedes aegypti*	(sub)tropics	+++
Trematodes	schistosomiasis (bilharzia)	snails	(sub)tropics	++
West-Nile virus	West-Nile virus	mosquitos	tropics	++

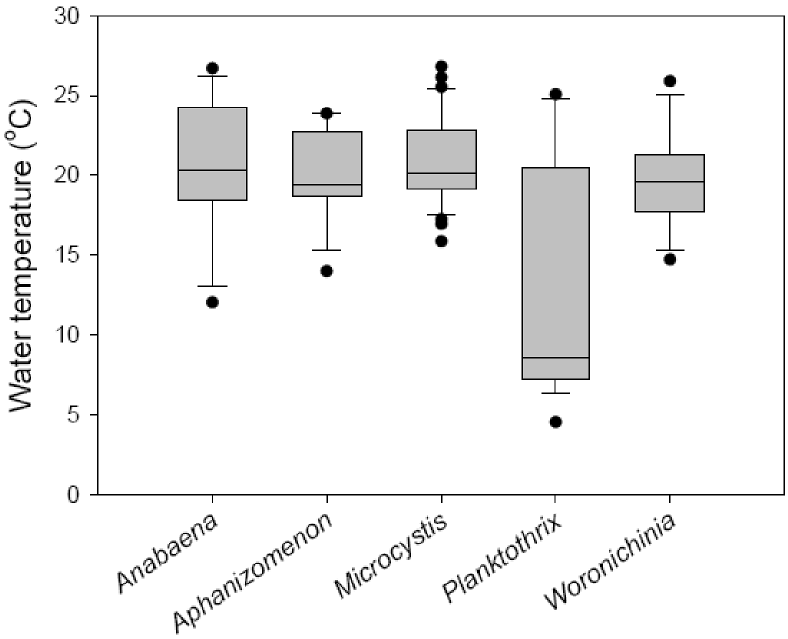


Figure 4.1 - Box-plots of water temperatures at which surface scums of five most prominent cyanobacteria genera were found. Boxes show medians and 25th and 75th quartiles, bars indicate 10th and 90th percentiles, and dot symbols represent outliers. (Roijackers and Lürling, 2007).

Even slightly increased temperatures could lead to higher biomass and dominance of cyanobacteria in some aquatic systems (Fernald et al., 2007). Moreover, climate related increased risk of eutrophication will further enhance the formation of algal and toxic cyanobacterial blooms (Fig. 4.1), that can cause human diseases. Bathing is not recommended in such water bodies due to the existence of cyanobacterial toxins that may exceed the WHO threshold (WHO, 2003).

The measures already taken to reduce eutrophication (reducing nutrient loading; improved water treatment) have been at least partly successful in many river basins of Europe (Lyche-Solheim et al., 2010). There is a risk that climate change will counteract this success (Moss et al., 2009), and that the public health risk for waterborne diseases is increased (Fig. 4.2).

Figure 4.2 - Children playing on the shore of Lake Reeuwijk (Roijackers and Lürling, 2007).

Increased colour (DOC) caused by a combination of climate change and reduced sulphate deposition (see Chapter 1, Section 1.3) increases the required DOC removal capacity in drinking water treatment, and significantly affects treatment process selection, design and operation. As an illustration, an increase in raw water color from 20 to 35 mg Pt L^{-1} increased the required coagulant dose, sludge production, number of backwashes per day and residual DOC by 64 %, 64 %, 87 % and 26 %, respectively. In addition, hydraulic capacity and filter run time decreased by 10 % and 47 %, respectively (Eikebrokk et al., 2004).

Sport fishing may also be negatively impacted by climate change, since most of the attractive fish species (salmonids, coregonids) will become reduced or lost (Reinhartz, 2007 and Chapter 3.2). The concentrations of hazardous substances (e.g. pesticides, mercury) are already too high in the fish in many areas, causing governmental regulations to restrict the consumption. An example from Norway is the Lake Mjøsa, where high concentrations of PCB and mercury were detected requiring diet regulations not to eat trout more than once a month (Løvik, 2008). As climate change has an influence on the water-chemistry, such as more coloured waters and thereby higher bioavailability of mercury, it will also have an effect on the concentration of contaminants in fish.

Boating, walking along the shore and bird watching are activities related to water-bodies but without direct contact with the water, so no health risk is associated with climate change impacts on water quality and biodiversity. Nevertheless, these activities can indirectly be influenced by climate change, e.g. algal blooms compromise the aesthetic perception of a water body, and dominance of alien species can reduce the chance to see rare bird species. Thus the positive impacts of these activities on human well-being can be reduced. Moreover, the socio-economic value of tourism and properties in riparian areas can be reduced.

4.3. Conclusion

The impact of climate change on water quality, biodiversity and ecological status of inland surface waters may cause increased risks to human health and pose a threat to recreational water use. Related socio-economic impacts can be expected, including increased costs for drinking water supply and sanitation, as well as reduced property prices and tourism in riparian areas. Future land use intensity, rain patterns and level of implementation of environmental and other regulations, such as the WFD and the Common Agricultural Policy, will affect the magnitude of these impacts.

5. Possible adaptation measures

5.1. Adaptation to climate change impacts on water quality and biodiversity

Mechanisms

Society needs to avoid the unmanageable through the reduction of greenhouse gas emissions and manage the unavoidable through adaptation measures (Scientific Expert Group on Climate Change, 2007). Adaptation is very much about managing the risks associated with future climate change impacts. Adaptation to climate change is defined by the IPCC as 'Adjustment in natural or human systems in response to actual or expected climatic stimuli or their effects, which moderates harm or exploits beneficial opportunities' (IPCC 2007). Adaptation aims particularly at reducing unavoidable negative impacts already in the shorter term, reducing vulnerability to present climate variability, and exploiting opportunities provided by climate change. Adaptation includes pro-active and reactive measures, which relate mainly to planned adaptation, as well as autonomous actions. Mitigation aims at avoiding the unmanageable impacts, while adaptation aims at managing the unavoidable impacts. Adaptation occurs primarily at transboundary (e.g. river catchments), sub-national and local levels, and therefore involves many levels of decision-making. As a consequence of that, adaptation has to be tailor-made to the specifics of the geographic area considered in terms of landscape types and sectors involved. National strategies provide the framework for adaptation actions, many of which have to be implemented at sub-national and local levels. The relevance of adaptation at the EU level is primarily concerned with coordinating information sharing, and encouraging an appropriate, proportionate and integrated implementation of adaptation measures at the different levels. The integration of adaptation into EU sectoral policies, structural/cohesion funds together with fostering research and involving stakeholders are key instruments in this respect (EEA, 2008).

Climate change can result in significant changes in the variables that affect the quality of water as described earlier in this report. Physical changes may occur in water temperature, ice-cover, stratification of water masses in lakes and water discharge including water level and retention. Chemical changes relate in particular to oxygen content, nutrient loading and water colour while the biological changes are affecting the structure and functioning of freshwater ecosystems. Changes in these variables lead to impacts on all the socio-economic and environmental goods and services that depend on these systems directly or indirectly (EEA, 2008 and Chapter 4 above). Key economic sectors, which will need to adapt through integration within sectoral policies at European and national levels, include energy supply, health, water management, agriculture, tourism and transport. Adaptation strategies should also consider the costs and the benefits of each measure and of the combinations of measures.

Adaptation to climate variability and change is a process of assessing and responding to present and future impacts, planning to reduce the risk of adverse outcomes, and increasing adaptive capacity and resilience in responding to multiple stresses. A key step is to make use of the best available science to identify conditions and risks as well as their relevance for adaptation strategies and actions to allow adapting to new boundary conditions due to climate change. Any analysis carried out as part of river basin management planning should take into account the existence of uncertainty and, where possible, use a range of climate projections including a variety of climate models, emissions scenarios and timescales. Proper staff training and cooperation between different levels of authorities and sectors, also need to be considered in efforts to build up adaptive capacity for management under climate change. Adapting to climate change implies new requirements regarding the type and the extent of data collected for river basin management. Long time series are judged essential for understanding the changes of physical variables and selected species over time. (EEA 2009a, EC, 2009).

Past trends

At the 2007 UN Framework Convention on Climate Change (UNFCCC) Bali conference, the urgency of responding effectively to climate change through both adaptation and mitigation activities was recognised by a larger number of countries than ever before (EEA, 2008).

The WFD, which entered into force on 22 December 2000, introduced a significant shift in regulatory approach from one of multiple instruments with separate (but overlapping) objectives to one providing an integrated framework covering all variables affecting the status of water bodies and the water needs of terrestrial ecosystems and wetlands that depend directly on aquatic ecosystems. The effectiveness of the WFD in the face of climate change clearly depends on the extent to which scenarios are introduced into the river basin management plan process. (EEA, 2007)

In the context of the WFD Common Implementation Strategy, an activity on Climate Change and Water was initiated in 2007 to produce guidance on how EU Member States should incorporate consideration of climate variability and change into the implementation of EU water policy. This activity is supported by a Strategic Steering Group (SSG). In 2008 the Water Directors agreed key messages on Climate Change and Water, and the SSG exchanged practices in incorporating climate change in the first River Basin Management Plan (EEA 2009a). EU Member States are at different stages of preparing, developing and implementing national adaptation strategies, depending on the nature of the observed impacts, assessment of vulnerability and capacity to adapt (EEA, 2008). Most European Member States have started drafting adaptation programmes or strategies, and a great number of regional adaptation projects were launched in recent years. However, experience in designing adaptation strategies and implementing policies is still limited (EEA 2009b). A brief overview of the status of European countries is presented later in this chapter.

Future projections

In 2006 the Stern Review (Stern 2006) recognised that adaptation in developed countries is still at an early stage despite well-developed market structures and the relatively high capacity to adapt. It suggested that governments had a role in providing a clear policy framework to guide effective adaptation by individuals and firms in the medium and longer term. In 2007, the European Commission Green Paper on adaptation (EC, 2007) stressed that early action is most important to prevent reactive and un-planned adaptation as response to more frequent crises and natural disasters. With late or no action, damages and associated economic cost would rise sharply until 2080. The Green Paper also calls for an integration of climate change adaptation into existing policies. It stresses that early adaptation will bring economic benefits and may even help to gain competitive advantages through the development of new technologies. The Green Paper analyses how adaptation efforts could be integrated into existing sectoral EU policies, how Community funding programmes could take climate change and adaptation into account, and also explores the scope for developing new policy responses, in particular with respect to financial services and insurance, and spatial planning. The Green Paper announces that a systematic check of how climate change will affect all Community policy and legislation should be carried out by 2009.The White Paper of the European Commission “Adapting to climate change: Towards a European framework for action” (COM/2009/147) was issued in April 2009 and sets out a framework to reduce the EU’s vulnerability to the impact of climate change (EEA, 2009a; EC, 2009).

5.2. Concrete adaptation measures

i. Measures for adaptation related to the WFD

WFD offers important tools for adapting to climate change impacts. In particular, the integrated approaches to land, water and ecosystem management, combined with the cyclical review of progress, are all consistent with the ideals of adaptive management. However, sometimes it is not easy to distinguish regular water management issues and measures from adaptation measures. Some of the river basin management planning steps are considered more critical than others in our ability to prepare for climate change, especially in the short term (EC, 2009; Table 5.1):

Table 5.1 - Criteria to help select adaptation measures (EEA, 2009a).

Example – Criteria to help selecting adaptation measures

Criterion	Sub-criteria	Guiding questions to be asked
Effectiveness of adaptation	Adaptation function	Does the measure provide adaptation in terms of reducing impacts, reducing exposure, enhancing resilience or enhancing opportunities?
	Robustness to uncertainty	Is the measure effective under different climate scenarios and different socio-economic scenarios?
	Flexibility	Can adjustments be made later if conditions change again or if changes are different from those expected today?
Side-effects	No regret	Does the measure contribute to more sustainable water management and bring benefits in terms of also alleviating already existing problems?
	Win-win (or win-lose)?	• Does the measure entail side-benefits for other social, environmental or economic objectives?
	Spill-over effects	Does the measure affect other sectors or agents in terms of their adaptive capacity? Does the measure cause or exacerbate other environmental pressures?
Efficiency/ costs and benefits	Low-regret	Are the benefits the measure will bring high relative to the costs? (If possible, consider also distributional effects (e.g. balance between public and private costs), as well as non-market values and adverse impacts on other policy goals)
Framework conditions for decision-making	Equity and legitimacy	Who wins and who loses from adaptation? Who decides about adaptation? Are decision-making procedures accepted by those affected and do they involve stakeholders? Are there any distributional impacts of the climate change impacts or of the adaptation measures?
	Feasibility of implementation	What barriers are there to implementation? • Technical • Social (number of stakeholders, diversity of values and interests, level of resistance) • Institutional (conflicts between regulations, degree of cooperation, necessary changes to current administrative arrangements)
	Alternatives	Are there alternatives to the envisaged adaptation measure that would e.g. be less costly or would have fewer negative side-effects?
	Priority and urgency	How severe are the climate impacts the adaptation measure would address relative to other impacts expected in the area/river basin/country? When are the climate change impacts expected to occur? At what timescales does action need to be taken?

ii. Adaptation measures for point source pollution control

Sewage treatment plants continue to be one of the main sources of point pollution discharges in Europe. The main pollution problems associated with these are the nutrients nitrates and phosphorus. Uncontrolled or accidental discharge of untreated wastewaters into environmental waters is particularly problematic from an environmental perspective, due to the high levels of nutrients contained. The risk of this kind of discharge is particularly high in sewerage systems which combine storm water and wastewater; overflows of the system into water bodies, due to flash floods for instance, can contain important amounts of untreated wastewater and its pollutants (EEA, 2009b).

Pre-emptive reduction of point source pollution

The decrease in rivers' summer discharge predicted for various regions in Europe implies a reduction of the carrying capacity of the water bodies, i.e. discharges of pollutants that are currently acceptable and unproblematic can result in unacceptable pollution levels in future. The pre-emptive reductions of point source pollution in view of these decreases in discharges are a clear example of a pro-active adaptation measure. The Netherlands are currently addressing point-source pollution taking into consideration the predicted impacts of climate change on low water flow. Measures of this type specifically targeting nutrient loading from point sources are also recommended for reducing the risk of eutrophication in lakes.

Improving industry risk management

Industrial accidents leading to pollution of water bodies can have strong impacts on ecosystems already under additional stress due to climate change impacts. Finland has addressed this risk by heightening requirements for industrial security and for contingency plans. In addition to environmental benefits, the measure has clear additional benefits for human health.

Reducing phosphate load in wastewaters

As a nutrient the phosphate present in wastewaters can lead to eutrophication of surface waters. Climate change increases this risk due to the reduced river flows and increased frequency of flash floods – with increased storm and sewage water overflow – predicted for some regions. One measure addressing this is the banning or minimising of phosphate in relevant domestic products; many European countries have banned (e.g. UK) or drastically reduced (e.g. Germany) phosphates in laundry cleaning products by law. A second measure which makes use of the economic value of phosphorus and addresses its presence in other domestic sources is reclaiming the phosphorus content of wastewaters. This technique has been applied in pilot projects in Germany. Phosphorus reclamation is considered as possibly a very effective solution for maintaining high water quality in a changing climate, at the same time delivering economic benefits.

Separation of rainwater and sewage

Untreated wastewater effluents reaching surface waters constitute a serious pollution problem. A possibility, both when building new canalisation and renovating existing one, is to separate wastewater and storm water. This is usually a comparatively expensive option, but it strongly reduces the risk of untreated wastewater entering environmental waters. More cost-effective ways of adapting old infrastructure which mixed rainwater and sewage to this newer standard have been implemented in pilot projects in the cities in Germany. In one of the approaches used - all of which separate rainwater from sewerage - new tubing was placed within existing canalisation, thus separating the two streams.

Incorporating climate change considerations into discharge licensing schemes

New forms of discharge license agreements include provisions that address uncertainty due to climate change. Licenses from point sources discharges are reviewed periodically; changes related to climate change impacts, as well as changes in other parameters such as population behaviour and available technology can thus be reflected in the revised agreements. In the UK, discharge licenses of wastewater treatment plants are reviewed every 5 years to this purpose. In Belgium, regulations aim to tie up wastewater discharges to the carrying capacity of the receiving water body. Changes to the system's carrying capacity due to climate change will be reflected in the pollution load the system is allowed to take up.

Improvement of wastewater infrastructure capacity

Untreated wastewater effluents reaching surface waters constitutes a serious pollution problem, mainly due to the risk of eutrophication of surface waters due to nutrients, but also due to a variety of other pollutants present in wastewater. In many regions the frequency of these extreme events such as flash floods is expected to increase due to climate change. Improvements in the capacity of the treatment plants or in the storage capacity can address this problem. An example of a way to increase capacity that does not imply major new infrastructure works is the wastewater control system (LISA) implemented by the Berlin water service provider. In case of heavy rainfalls a computer-controlled system automatically coordinates all 148 pumps to shift the mix of storm- and wastewater through the connected sewerage system to those treatment plants that still have capacity. Thanks to this system, the overall amount of storm- and wastewater overflows has been reduced by approximately 20%.

iii. Adaptation measures for diffuse pollution source control

Agriculture is the main source of diffuse pollution affecting environmental waters in Europe. Agriculture is a sector that can make large contributions to adaptation to climate change and water quality. Thus, policy coherence with the EU Common Agricultural Policy's provisions should be ensured with regard to adaptation objectives in water management. Conventional agriculture implies a certain level of leaching or washout of nutrients and pesticides, either to surface waters or to groundwater bodies. Some pollutants, such as phosphorus, adhere to soil particles; in this case they reach surface water bodies not in the water that makes up surface run-off, but on the soil particles transported due to erosion processes. The measures presented below aim to contain the pollutant loads that take these transport paths (EEA, 2009b). From the several national and European research activities there is limited empirical evidence to demonstrate impacts unequivocally, because of difficulties in disentangling the effects of climatic factors from other pressures. On the other hand, there are many indications in freshwaters that are already under stress from human activities which is vulnerable to climate change impacts. In some cases climate change may significantly hinder attempts to restore some water bodies to good status on the long term (EEA, 2009a).

Support for switching to organic farming

Organic farming strongly limits the possibility of agriculture-related diffuse pollution entering groundwater or surface water bodies. In addition to restrictions typically prohibiting the use of artificial pesticides and fertilisers and regulating the use of natural fertilisers, requirements also typically include rules for improved soil structure and functioning, thus reducing the risk of soil erosion and unwanted sediment transport. The main drawback of this measure is the loss of productivity in organic systems; due to this some schemes compensate the farmers economically for their using this production system.

Precision farming

Precision farming is based on the principle of meeting crop requirements as precisely as possible, using high technology (GPS yield mapping, variable rate delivery of seed, pesticides and fertilisers) at a very fine scale (a few square metres), and aiming for application at the best possible moment in time. This results in maximisation of efficiency of all inputs, and reduces the risk of nutrient and pesticides run-off to surface water, leaching to groundwater, etc.

Buffer strips between water bodies and agricultural fields

Vegetated and unfertilized buffer zones alongside watercourses act as a shield against overland flow from agricultural fields and reduce run-off from reaching the watercourse, thus decreasing erosion and the movement of pollutants into watercourses. Research suggests very high potential for controlling nitrogen pollutants (including nitrate), but performance is strongly dependent on characteristics such as buffer zone width, slope of the drained field, soil type and variety, and density of zone vegetation. Additionally, suspended solids and sediments (with adsorbed pollutants such as phosphorus) are filtered and channel erosion is reduced; protection against pesticides and heavy metals is also judged to be very positive. Buffer strips also effect a reduction in pollution by changing land use. Buffer strips are being widely supported as agri-environmental measures in European rural development programmes.

Buffer strips at field margins (e.g. next to roads), between fields and within fields

The principles behind and the effects of this measure are the same as for the previous one, but whereas the previous measure addresses run-off in the immediate vicinity of a particular water body, this measure addresses the problem close to its origin. The strips' location aims to reduce surface run-off, as well as soil transport, coming out of individual fields. The measure shows particularly good results for soil retention. Locating buffer strips within fields makes sense under extreme conditions such as strongly sloping fields located close to watercourses.

Erosion reduction measures

Agricultural measures that reduce erosion and/or aim at soil conservation reduce the input of solid and suspended particles into water bodies, and consequently the input of diffuse agricultural pollutants such as phosphorus. Some of these measures (e.g. winter plant cover) also take up soil nutrients, reducing the nutrient load available for leaching. Many of these measures improve soil structure and increase the water retention capacity of soils. The measures also have the potential of strongly reducing the run-off from agricultural fields.

- **Continuous plant cover (catch crops, intercrops) / winter plant cover:** A cover crop is planted in late summer or fall to provide soil cover during the winter. A cover crop will take up residual nitrate and other nutrients from the soil, and helps stabilise soil thus reducing soil erosion and the mobilisation of associated pollutants.
- **Green stripes between fields:** The establishment of green stripes of some meters width between agricultural fields, either with permanent or with temporary (yearly) cover, strongly reduces the erosion of field soil and reduces surface run-off, thus reducing the input of nutrients and pesticides into surface waters. If local varieties of plants/grass are used, the measure also has positive effects on biodiversity.
- **Mulch sowing:** Maintaining plant rests within the soil and abstaining from the use of ploughs naturally creates a protective layer of mulch while new plants are sowed directly into the soil through the mulch with adequate equipment. This helps maintain organic matter and preserve good soil structure, thus improving infiltration and retention of water and thereby decrease erosion and pollutant concentrations in surface run-off.
- **Preventing soil compaction:** Soil compaction reduces water infiltration into soils, thus favouring surface run-off and erosion; compaction-free soils are also more drought resistant for crops Options for preventing compaction including use of low ground pressure tyres or tracks on vehicles, avoiding wet soils, and adding organic matter to soil.

Various programmes, including some for environmentally friendly agriculture, support these measures. The European Regional Development Programme for instance sets out the amount of support provided to farmers for their implementation of some of the above-mentioned measures.

Fertiliser, manure and slurry management measures

A rational and planned application of fertilisers/manure/slurry, which is well-timed and which takes account of local parameters such as soil type and structure, is a wide-spread tool for reducing nutrient leaching and which can have an enormous impact on water quality. By keeping nutrient levels as close as possible to plant requirements over time, excess nutrient in the soil, which can be flushed out e.g. by precipitation, is reduced to a minimum. These management measures include:

- **Nutrient balances:** Nutrient balance spreadsheets inform farmers on the efficiency of nutrient utilization and help identify the cropping phases in which nutrients are lost. They help to accurately account for fertilizer use and reduce unnecessary nutrient inputs. The reductions have positive effects on both surface and ground waters.
- **Manure application techniques:** A measure of this type involves for instance cutting slots in the soil, injecting the slurry and then closing these slots after application. Injecting slurry as opposed to applying it on topsoil makes it possible to directly reach the active soil layer in order to reduce nutrient leaching. This reduces groundwater and surface water pollution from nitrate leaching and phosphate run off.
- **Integration of fertiliser and manure nutrient supply:** Determining the amount of nutrients supplied to soils during manure application helps farmers judge the amount and ideal timing of additional fertilizers required by the crop. Taking better account of the nutrients contained in manure can reduce the need for fertilizer inputs, which in turn minimises nitrate and phosphorus losses.

Additional options include the adoption of improved precision techniques such as soil analysis, manure analysis, adaptation of fertiliser and pesticide application on demand, matching fertiliser to seasonal conditions, and slow and controlled release fertilisers. Requirements regarding fertiliser and manure management measures are being used widely, ranging from simple restrictions regarding the timing of manure or slurry application to widespread implementation of nutrient balance spreadsheets.

Risk-based fertiliser and manure restrictions

Risk areas include areas with flushes draining to a nearby watercourse, cracked soils over field drains, or fields with a high phosphorus index. By avoiding the spreading of mineral fertilizers or manure at high risk times, the nitrate leaching and loss of phosphorus through surface run off is diminished. High risk times include when there is a high risk of surface flow, rapid movement to field drains from wet soils, or when there is little or no crop uptake. The measure requires adequate collection and storage facilities.

iv. Adaptive measures for biodiversity

This section presents measures related to biodiversity which are not directly associated with the two main water quality problems of the previous sections.

Barrier removal for improved species migration

The predicted increase in water temperature is expected to effect a change on distribution of water species; species are expected to move to cooler waters, which implies a general move northwards as well as a move to higher elevations (i.e. upstream). Numerous initiatives are currently being implemented that remove man-made fish barriers or provide fish passage facilities at man-made barriers as a response to WFD requirements.

Mitigation of water temperature increases by establishing wooded riparian areas

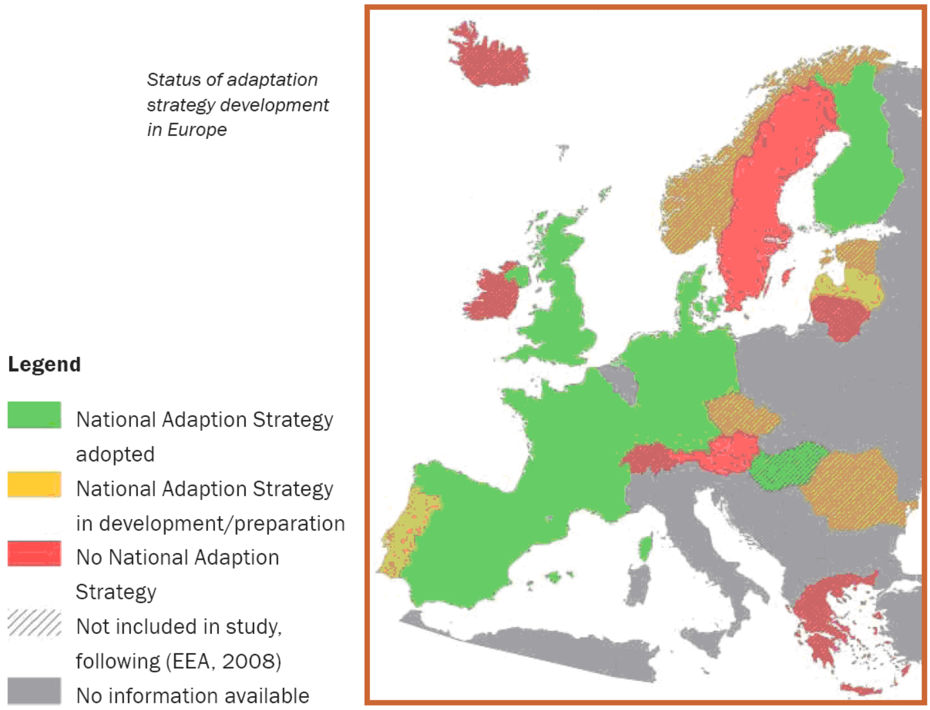
To the present only one measure has received widespread recognition as having potential to influence water temperature increases. Riparian areas with trees provide direct shade for the water body, reducing the influx of solar radiation on it and thus avoiding the corresponding increase in water temperature. In the case of wide riparian wooded areas, these can also increase the relative air humidity, which also contributes to reduce water temperature. This measure is considered particularly relevant for headwaters; its positive influence on water temperature and related biological processes extends to downstream regions. It is typically not considered applicable downstream because rivers are usually too wide to be influenced in their temperature by canopy cover in these regions; however, narrower rivers could benefit from this measure's implementation independently of its in-basin location.

Cash crops

Cash crops are usually sown after the harvest of one crop and before the sowing of the next. They offer forage or green manure (providing fertility for the soil thereby reducing nitrogen applications for the next crop) potential and are usually based on quick growing plants that will establish before winter. Their mitigation benefits include reducing N_2O emissions or leakage, improving N-use efficiency and carbon sequestration in the soil. When it comes to impacts on biodiversity, catch crops reduce nitrate leaching which can cause eutrophication in watercourses. They can also provide cover for many farmland bird and insect species and reduce soil erosion which would have negative effects on local biodiversity.

5.3. Status of adaptation strategies in European countries

A number of national adaptation strategies have been developed by European countries, and several are in the development process. Many of them address impacts on water and draw up a strategy to adapt water management accordingly (Map 5.1).



Map 5.1 - Status of adaptation strategy development in Europe (Swart et al., 2009).

Key messages

- Adaptation aims particularly at reducing unavoidable negative impacts already in the shorter term, reducing vulnerability to present climate variability, and exploiting opportunities provided by climate change.
- WFD offers important tools for adapting to climate change impacts. The integrated approaches to land, water and ecosystem management, combined with the cyclical review of progress, are all consistent with the ideals of adaptive management.
- The European Commission has set out a framework to reduce EU's vulnerability to the impact of climate change (European Commission, 2009).
- A number of national adaptation strategies have been developed by European countries, and several are in the development process.
- Several possible adaptation measures related to water quality are presented in this chapter classified as measures for point pollution source control, diffuse pollution source control and biodiversity.
- There is a need for integration of climate change adaptation into existing policies.
- With late or no action, damages and associated economic cost would rise sharply until 2080.

6. Conclusions and key messages

The most important conclusions and key messages presented in this report is that climate change has already started to show impacts on the surface waters of Europe and on their biodiversity and ecological status, and that these first symptoms will increase over the coming years. These impacts have already clear consequences for water use and for public health. A series of adaptation measures are urgently needed to counteract the negative impacts of climate change on European freshwater ecosystems and on the services they provide to human well-being.

The clearest impacts related to water quality, biodiversity and health are listed below:

- Warmer water, reduced ice cover and longer growing season.
- More stable stratification and less mixing in lakes.
- Browner water in the northern region (more DOC).
- Increased eutrophication with more harmful algal blooms, reduced water transparency and declining oxygen concentrations.
- Increased risk of inputs and bioavailability of hazardous substances, such as Hg and pesticides.
- Reduction of biomass and biodiversity of cold-adapted species like salmonids and coregonids.
- Major loss of biodiversity in the southern regions due to increased droughts and hot weather.
- Increase of invasive exotic species, including pathogenic micro-organisms.
- Deterioration of ecological status in most lake and river types.
- Increased risk of exceeding the WFD good status target.
- Increased health risk due to more toxic algae, more pathogens and sewage overflow.
- Increased socio-economic costs related to water supply and sanitation.

A series of adaptation measures are underway to counteract these negative impacts, such as improved point source control and measures to reduce diffuse pollution esp. agricultural measures, change of industrial processes to reduce effluents, banning phosphorus in detergents, restoration measures in rivers and lakes (barrier removal, shading). The uncertainty is whether they will be sufficient and be implemented soon enough to prevent these negative impacts on the inland waters of Europe.

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Abbreviations

CHAB	Cyanobacterial harmful algal blooms
DOC	Dissolved organic carbon
EC	European Commission
EEA	European Environment Agency
ETC	European Topic Centre
Euro-Limpacs	The impact of global change on European freshwater ecosystems
IPCC	Intergovernmental Panel on Climate Change
JRC	Joint Research Centre
NAO	North Atlantic Oscillation index
PCB	Polychlorinated biphenyls
POPs	Persistent organic pollutants
REFRESH	Adaptive strategies to mitigate the impacts of climate change on European freshwater ecosystems
WFD	Water Framework Directive
WHO	World Health Organisation