Impact of climate warming on European inland waters

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Abstract

Climate change impacts include such an overwhelming number of facets that the extent of references is impossible to cover. The selection presented in this review on global warming is presumably subjective, although the author has tried to assemble relevant and/or most recent information. The references specified should be regarded as examples that direct the reader to more information on a specific subject.

Motto: How to ruin the planet? Ignore global warming

Introduction

Today, apparent environmental changes are global and largely anthropogenic. The world's rapidly growing human population and its associated consumption of resources affect the whole planet (Vitousek 1994, Vitousek et al. 1997). In the context of recent multiple shifts in paradigms, we have presumably entered the "anthropocene," a quasi-geological era (Gerten 2008).

Although about 67% of the Earth's surface is covered by water, less than 2.7% of global water is freshwater, with the majority (2.05%) locked in ice caps and glaciers; less than 0.7% is available for human use. These inland water resources are particularly under threat by a multitude of impacts originating mainly from man. Fresh water is an increasingly scarce resource caused partially by increasing demand and climate warming (Vörösmarty et al. 2000). Changes in the pattern of consumption and a broad spectrum of human-mediated hydrological impacts beyond the greenhouse effect have been identified (Vörösmarty 2002).

As an additional complication, most inland lakes, rivers, and groundwater aquifers are transboundary waters shared by 2 or more nations. International basins cover about half of the Earth's land surface, and about 40% of the world's population relies on these shared water sources (Cooley et al. 2012). Because climate warming is essentially a catchment related process, transboundary strategies need to be developed (Tetzlaff et al. 2008, Everard et al. 2009, Heathwaite 2010). Expected changes in the hydrology of watersheds and processes within the catchment will affect runoff and ultimately inland waters (Jeppesen et al. 2009, Schneiderman et al. 2010).

Inland surface waters are directly and almost immediately affected by changes in temperature because of the close coupling between air temperature (AT) and surface water temperature (SWT; McCombie 1959, Smith 1981, Livingstone and Lotter 1998). Additionally, impacts are mediated via processes in the catchment (Fig. IV.B.5 in Dokulil 2005). All potential direct and indirect impacts are modulated and filtered by the landscape and the different types of waters (Blenckner 2005). The resulting multitude of internal changes affects water quality and trophic level (Dokulil and Teubner 2011, Winder 2012). Additional stress can come from tourism in many regions (Dokulil 2014). Whether eutrophication promotes climate change through positive feedback is still under debate (Moss et al. 2011).

Groundwater generally responds slower than surface water systems. Effects correlate more strongly with precipitation than with temperature, which is more important for shallow aquifers and during warm periods (e.g., Schartner and Kralik 2011). Aquifers are depleted worldwide by over-consumption as a result of population growth, and the groundwater abstraction rate has reached more than 100% of mean recharge in several countries (Gleick 2012).

Climate forcing

The reality of climate warming seems now widely accepted (IPCC 2007); however, scepticism remains as to the degree changes are natural or anthropogenic. The controversy essentially centres on whether greenhouse

gases or the sun cycle is more responsible for the observed changes (e.g., Hulme 2009 or http://www.skepticalscience.com/). Climate warming is amply documented by a number of long-term records of mean global or regional AT, which will not be iterated here. These long-term temperature records can also be modelled (e.g., Samuelson 2010). A few remarks on the European climate are necessary, however. Although most of Europe's climate depends to a large extent on weather patterns across the Atlantic, the dominant weather types vary spatially and seasonally among geographical regions. The classification schemes devised by different authors for European regions are summarised in George et al. (2010). In Northern Europe the anticyclonic weather type dominates, with highest frequencies in winter and spring. Westerly weather types are more common during winter. A similar pattern occurs in Western Europe, where the anticyclonic weather type also dominates, but the highest frequencies are shifted to late spring and early summer, and the westerly type becomes more pronounced particularly in autumn and winter. In both regions northerly weather types are of minor importance. The seasonal variation of weather types is different in Central Europe. The westerly and northerly types are most important, but the southerly type is also common (Fig. 16.2 in George et al. 2010). Climate trend analyses from more than 100 stations across Europe for 1946–1999 indicate an increase in temperature variability due to stagnation in the warming of cold extremes (Klein Tank and Können 2003).

Surface AT increased on average 0.7 °C ± 0.2 globally during the 20th century (IPCC 2007). In Northern Europe, AT increased by 0.3–1.0 °C in different parts of Fennoscandia (Tveito et al. 2000); in Germany the rise was estimated as 0.9 °C (http://www.dwd.de); and an increase of 0.5–0.8°C was reported for various parts of the Mediterranean (Xoplaki et al. 2003, Philandras et al. 2008).

Climate warming was even more pronounced in the greater perialpine region, where air temperatures have increased stepwise by almost 2 °C, twice the global average (Beniston et al. 1997, Auer et al. 2007). Spatial variation, both horizontal and vertical, was negligible (www.zamg.ac.at/cms/de/klima/informationsportal-klimawandel/klimavergangenheit/neoklima/ lufttemperatur). This pronounced temperature increase is due to the well-documented northward shift of the subtropical high pressure system, resulting in parallel development and variation of air pressure, AT, and sun duration at all elevations (Auer et al. 2007).

Despite these variable climatic conditions across Europe, impacts on inland waters are surprisingly coherent, both regionally and vertically, and over greater distances (Dokulil and Teubner 2002, Livingstone and Padisák 2007, Livingstone et al. 2010a). Physical properties such as temperature and the timing of ice-out exhibit a high degree of spatial coherence (Livingstone 2008). Chemical variables such as pH, conductivity, or oxygen concentration also seem to respond coherently to climate forcing. Elemental and nutrient concentration can be coherent or opposite, depending on the origin of the data. In waters within the same watershed, coherent concentrations of nutrients are more likely than between catchments or over larger distances, and coherence of biological parameters is usually weak. In other words, the odds of finding the same response of biological entities in neighbouring lakes is about 50:50 (Dokulil and Teubner 2002).

Current warming of inland waters

In the western part of Europe, SWT increased by 1.4 °C in Lake Windermere, English Lake District, between 1960 and 2000 and by 0.7 °C in Lough Feeagh, Western Ireland, between 1960 and 1997 (George et al. 2007). Mean lake temperature of Lake Zurich, Switzerland (Livingstone 2003), increased at a rate of 0.16 °C per decade from 1950 to 1990 (epi/metalimnion 0.24 °C per decade; hypolimnion 0.13 °C per decade). Decadal increase in mean lake temperature was 0.1 °C in Lake Constance, Switzerland, at approximately the same time (Straile et al. 2003). Epilimnetic temperature increases in July in a number of lakes from Switzerland, Germany, and Sweden ranged from 0.2 °C per decade in Greifensee, Switzerland, to 0.9 °C per decade in Lake Vättern, Sweden, with the majority of the European lakes increasing between 0.5 and 0.6 °C per decade. An exceptionally high increase of 1.6 °C per decade was reported from Lake Stensjön, Sweden (Adrian et al. 2009). SWT increased in April and August 0.37–0.75 and 0.32–0.42 °C per decade, respectively, in the large and shallow lakes Peipsi and Võrtsjärv in Estonia from 1961 to 2004 (Nõges et al. 2010a).

Long-term temperature changes of 19 European lakes from Finland, UK, Austria, and Switzerland were summarized by Arvola et al. (2010). Mean summer SWT increase per decade was 0.29 °C in Switzerland, 0.35 °C in UK, 0.38 °C in Finland, and 0.43 °C in Austria (see also Dokulil 2013). In the 9 Austrian lakes larger than 10 km^2, increments of SWT ranged from 0.4 to 0.66 °C per decade (mean 0.55 °C) for June to September 1965–2010 (Dokulil 2013); 4 of the 9 lakes increased on average by more than 0.6 °C per decade.

Warming rates were analysed for 23 Austrian lakes of varying size and location, and in almost all cases, long-term monthly mean AT are closely related to monthly mean SWT. The relation (slope) of SWT to AT ranged from 0.90 to 1.52 and their associated variances (r2) from 0.76 to 0.99. The average SWT anomaly from the Intergovernmental Panel on Climate Change (IPCC) base

period (1961–1990) was 0.86 °C for the 1991–2010 time window (Fig. 1; median 0.82 °C, range 0.14–1.41 °C). The average SWT anomaly increased to 0.93 °C when the time frame was changed to the 10-year period 2001–2010 (median 1.06 °C, range 0.04–1.61 °C).

The greatest deviation from the IPCC base period of 1.41 °C occurred in the 5 lakes south of the Alps, and the average increase was 0.78 °C in the 10 lakes north of the Alps (Fig. 2). The lowest mean anomaly of 0.58 °C was recorded for 6 lakes located in inneralpine valleys. SWTs have increased by 1.08 °C in the 2 lakes from the plains of eastern Austria. The size of the lakes seems to have some influence because the warming rate in 9 lakes larger than 10 km² was 1.05 °C while the 14 smaller lakes increased on average by 0.74 °C. Differences among groups were significant between lakes in the south and in inneralpine valleys and between lakes larger and smaller than 10 km² (Fig. 3). About half of the variance in the annual SWT anomalies can be explained by an empirical factor derived from lake morphometry and average heat influx. The function describes the heat transfer from air to water expressed by the inverse of the mean slope "b" of the long-term regression of air versus water temperature (1/bWT:AT), corrected by the hydraulic overflow rate $qs = z/T$, where z is the mean lake depth and T is the theoretical retention time.

Long-term mean seasonal SWT anomaly was 0.60 °C in winter, 1.11 °C in spring, 1.30 °C in summer, and 0.31 °C in autumn (Dokulil 2013). The maximum temperature increment of 2.54 °C occurred in spring in Weissensee, Germany, and the minimum SWT increment of −0.7 °C in Traunsee, Austria. Variability between lakes was largest in spring (2.14 °C) and smallest in fall (1.26 °C). Monthly long-term median SWT anomalies for the winter half year (Oct–Mar) were significantly correlated with positive indices of the long-term monthly median North Atlantic Oscillation (NAO) anomalies ($r^2 = 0.84$, $p = 0.10$), while the summer half year (Apr–Sep) was not significantly related to negative indices (Dokulil 2013). The overall relation for all months can be described by a nonlinear regression ($r^2 = 0.55$, $p = 0.080$). AT and SWT of Austrian lakes were significantly determined by the NAO winter index except for lakes isolated in alpine narrow valleys or south of the Alps (Livingstone and Dokulil 2001, Fig. 20.4 in Dokulil et al. 2010). On a larger scale, however, lakes respond coherently to a long-distance climatic signature across Europe (Blenckner et al. 2007).

Average long-term records of AT and SWT for 18 Austrian lakes show at least 3 phases of temperature increments between 1950 and 2010 (Fig. 4; Dokulil 2013). In the first phase from 1950 to 1980, average AT increased by 0.2 °C per decade and mean SWT remained constant. In the second phase from 1980 to 2000, both temperatures

increased at a rate of 0.5 and 0.6 °C per decade respectively. The 10-year period 2000–2009 is characterised by reduced warming; mean AT decreased by 0.02 °C per year and average SWT remained almost constant, similar to global observations (Hansen et al. 2013). This slowdown of climate warming (Knight et al. 2009) was recently explained by enhanced heat storage in the ocean, but the reasons for the warming pause have not been clearly identified (Guemas et al. 2013).
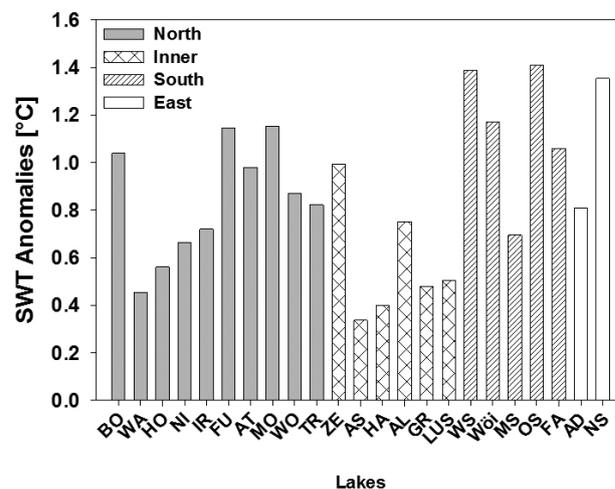

Fig. 1. Surface water temperature (SWT) anomalies as temperature difference in °C between the 1991–2010 period and the 1961–1990 base period as defined by the IPCC (2007) for 23 Austrian lakes. The lakes are grouped into lakes north of the Alps (BO = Bodensee or Lake Constance, WA = Wallersee, HO = Holzöster See, NI = Niedertrumer See, IR = Irrsee, FU = Fuschlsee, AT = Attersee, MO = Mondsee, WO = Wolfgang See, TR = Traunsee); inner alpine lakes (ZE = Zeller See, AS = Achensee, HA = Hallstätter See, AL = Altausseer See, GR = Grundlsee, LUS = Lunzer Untersee); lakes south of the Alps (WS = Weißensee, WÖS = Wörther See, MS = Millstätter See, OS = Ossiacher See); and lakes in the eastern plains (AD = Alte Donau, NS = Neusiedler See).
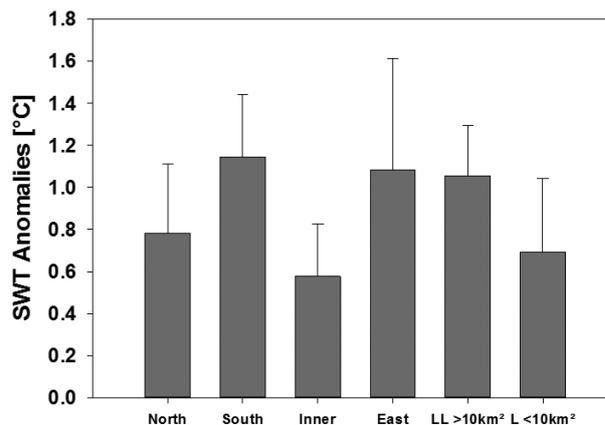

Fig. 2. Mean and standard deviation of SWT anomalies summarised for lakes north, south, and in and east of the Alps (see Fig. 1). Lakes are additionally grouped into lakes larger and smaller than 10 km².
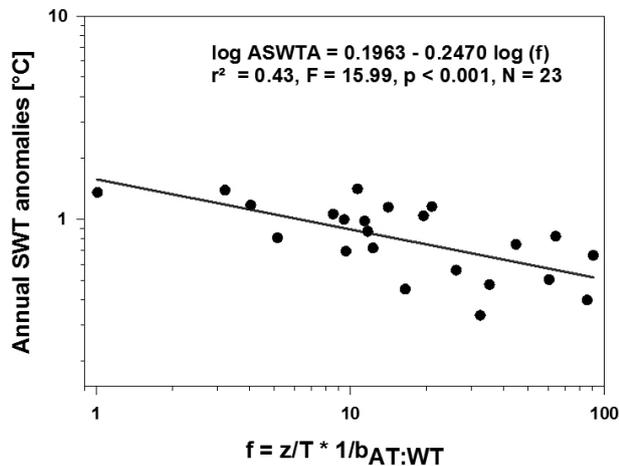
Fig. 3. Annual SWT anomalies (°C) for the 23 lakes shown in Fig. 1 against an empirical factor derived from the air temperature (AT) versus surface water temperature (SWT) relation and lake morphometry. $b_{AT:WT}$ = slope of the air–water temperature regression as expression of heat influx; z = mean depth; T = theoretical retention time; and z/T is the hydraulic overflow rate.
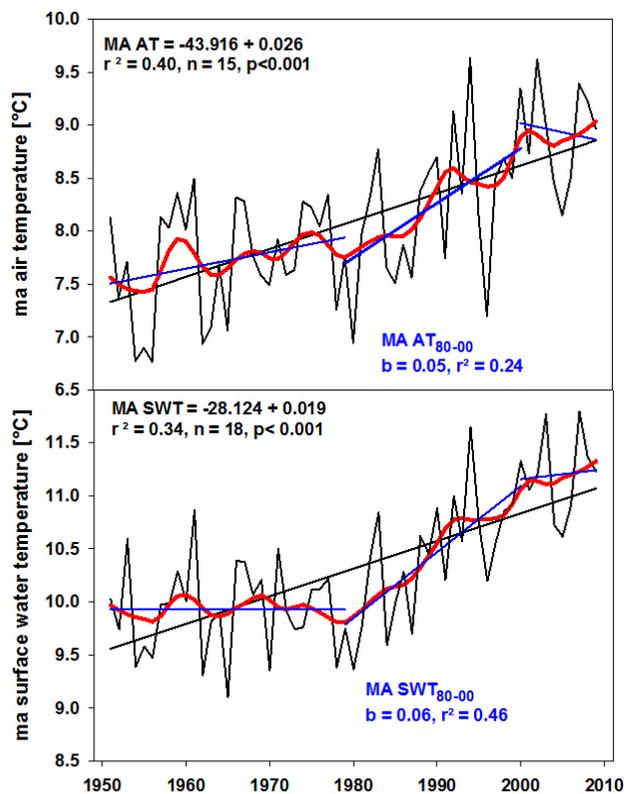

Fig. 4. Time series of mean annual (MA) air temperature (AT) for 15 stations close to the lakes (upper panel) and mean annual surface water temperatures (SWT) for 18 lakes (lower panel) for 1953–2010. Regressions and statistics are indicated for the entire time period and for the 1980 to 2000 period.

Increments in SWT are transmitted by convection and turbulence to deeper layers in the water column and eventually affect hypolimnetic water temperatures (HWT). Water temperatures had increased at all depths during 1963–1998 in Lago Maggiore, Northern Italy, but particularly in the deep water below 300 m (Ambrosetti and Brabanti 1999). Similar increases were found in several other northern Italian lakes. Salmaso (2005) reported HWT warming of 0.7 °C per decade in Lago di Grada from 1990 to 2003, and Livingstone (1993) observed warming rates ranging from 0.1 to 0.5 °C per decade in 4 Swiss lakes north of the Alps. Summarising results from 12 lakes across Europe, Dokulil et al. (2006) concluded that HWT had increased by about 0.1–0.2 °C per decade during the last half of the 20th century. In 10 of the 12 lakes, HWT was significantly related to the winter NAO index. HWT in one lake in Switzerland and one in Austria, both situated in remote narrow alpine valleys, were not determined by the long-distance climate signal (Dokulil et al. 2006). Deep water warming in some cases occurred in episodes of slow temperature rise and abrupt cooling events, creating a saw-tooth temperature structure (Livingstone 1997, Ambrosetti and Brabanti 1999).

Annual average water temperature in large rivers such the Rhine, Meuse, and Danube have increased by 1–3 °C over the last century. The water temperature of the downstream part of the Rhine increased by 3 °C between 1910 and 2010, with two-thirds of the increase attributed to the increased use of cooling water and one-third to climate change (EEA 2013). Water temperature increases in streams and rivers in Austria varied from 0.47 to 1.26 °C in the 20th century (Webb and Nobilis 1995), with average increments of 1.5 °C in summer and 0.7 °C in winter between 1976 and 2007 (Blöschl et al. 2011). The water temperature anomaly of the Danube at Vienna was 1.09 °C for 1991–2006 compared with the IPCC base period 1961–1990 (Fig. 5). Warming occurred regionally coherent at all altitudes in the European Alps (Hari et al. 2006); changes mainly reflect variations in regional AT. The absolute differences in water temperature from river to river are primarily a result of the general decrease that occurs with increasing altitude. Much of the warming occurred abruptly around 1987–1988, similar to that observed for lakes (North et al. 2013), possibly caused by a change in the Arctic Oscillation.

Groundwater is the most important source of raw drinking water worldwide, yet the potential impact of climate change on this vital resource is uncertain because relevant long-term data are scarce. A detailed analysis of Swiss aquifers fed by river-bank infiltration revealed an abrupt increase in annual mean groundwater, air, and river water temperature around 1987–1988 (Figura et al. 2011). The rise in groundwater temperature (GWT) of 0.5–0.7 °C

per decade was associated with a severe change in the performance of the Arctic Oscillation. Higher GWT affect biogeochemistry such as redox conditions in aquifers fed by river-bank infiltration. Reduced concentrations of dissolved oxygen in groundwater were already observed in the dry summer of 2003 (Figura et al. 2013).

Temperature data from groundwater not fed by bank infiltration and from springs do not show a regime shift (Schartner and Kralik 2011, Figura et al. 2013). Temperature in groundwater fed by primarily by precipitation is much less variable, and the reaction to changes is slow and delayed. GWT in porous groundwater bodies of Austria revealed an approximate increase of 0.7 °C per decade with a range of 0.4–1.3 °C (Schartner and Kralik 2011). Water temperature measurements in Austria springs indicate an increase of 0.5–1.5 °C between 1992 and 2008 (Kralik 2010); springs in Switzerland show no increase (Figura et al. 2013); and 48 springs in Hessen, Germany, vary from no change to an increase of 1.9 °C in one case. Average water temperature increase was 0.9 °C (http://www.hlug.de/?id=7626).

Temperature increase by 2050 and beyond

Dokulil (2009, 2013) analysed present SWT for 15 Austrian lakes and projected the potential temperature increase until 2050 for the June to September period (summer tourist season). Results indicated that changes in water temperature were the greatest in spring and summer. A conservative estimate of the average SWT increase during June to September by 2050 is 2 °C (1.2–2.9 °C), depending on present thermal regime and geographical region. High alpine lakes seem highly sensitive to global warming. A study on 15 Austrian lakes at altitudes 1500–2000 m predicts the summer epilimnion water temperatures to rise to an average temperature higher than 10 °C (Thompson et al. 2005). Models predict an increase of 1.4 to 5.4 °C for several lakes in Europe during 2071–2100 relative to 1961–1990 (Persson et al. 2005). Water temperature profiles and monthly mean water temperatures were simulated for 2041–2050 for the German Ammersee by Weinberger and Vetter (2012). Simulation results indicate an increase in water temperature at all depths. The predicted impact directly affects stratification characteristics and mixing processes. In the future, a greater quantity of energy derived from external forces will be required to initialise complete mixing because warming will likely inhibit complete mixing in deep, temperate lakes, even during intense wind storms (Ambrosetti et al. 2010, Rempfer et al. 2010).

Climate change will affect hydrologic and thermal regimes of rivers. The seasonality of river discharge will increase due to an increase in high flow and a decrease in low flow. River water temperatures are projected to increase on average by 0.8–1.6 °C (1.0–2.2) for 2071–2100 relative to 1971–2000, with the largest increase occurring in Europe (Van Vliet et al. 2013). The sensitivities are exacerbated by the projected decreases in low flows, which will reduce thermal capacity. Europe could therefore be among the regions potentially affected by increased deterioration of water quality and freshwater habitats as well as reduced water available for human uses (Van Vliet et al. 2013). River water temperature of the Danube near Vienna will possibly increase 0.5–2.1°C by 2015, depending on which period is extrapolated from various methods (Fig. 5).

Climate scenarios to 2099 predict an increase in average GWT of about 2.5 °C for Swiss riverbank-influenced aquifers and 0.7 °C for subsurface waters mainly infiltrated by precipitation (Figura et al. 2013); however, results are highly variable and associated with large uncertainties. Similarly, model projections to 2065 for shallow GWT in Canada predict an annual average increase of approximately 1.06 °C (Kurylyk et al. 2013).

Increased GWT will potentially lead to decreased dissolved oxygen levels and blooms of microbial populations, which can negatively impact human health; however, groundwater quality depends on effective self-purification through biodegradation. When groundwater is used as drinking water, added chlorine reacts with organic material producing trihalomethanes, some of which are potentially carcinogenic. Because the reaction is temperature dependent, higher GWT will increase the
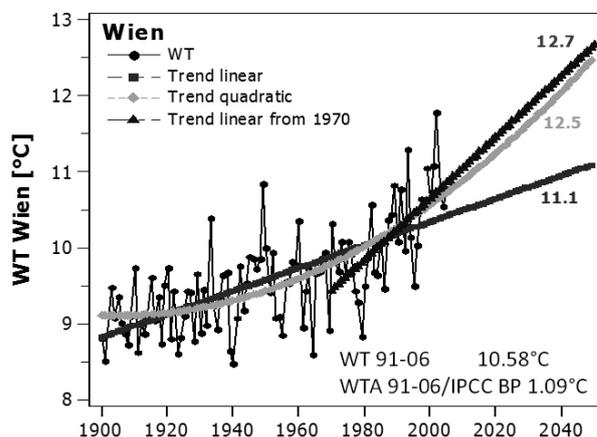

Fig. 5. Mean annual water temperature (WT) of the River Danube near Vienna (Wien) for 1900–2006. Average water temperature for 1991–2006 and water temperature anomaly (WTA) for that period from the IPCC base period 1961–1990 are indicated. Linear and quadratic regressions for the period 1900–2006 and extrapolated to 2050 are inserted. In addition, the linear trend and extrapolation beginning in 1970 is shown. For each extrapolation the expected temperature in 2050 is inserted.

production rate of trihalomethane as well as the consumption of residual chlorine; therefore, more chlorine must be supplied in the water purification process (Kralik 2010).

Catchment related processes

Impacts of climate change on inland waters are essentially a catchment-related problem. Changes in weather patterns within the watershed affect streams, rivers, and groundwater, especially changes in frequency and intensity of extreme events such as heavy rainfalls, storms, and extremely warm or mild seasons (Dokulil et al. 2010). In addition to the increase in precipitation, global warming is also expected to intensify the hydrological cycle by increasing evapotranspiration (Douville et al. 2012). Alterations in the hydrologic regime, glacial melting, and amount of ice and snow cover in the catchment will indirectly impact all water resources.

Streamflow observations in 441 catchments in 15 countries across Europe during 1962–2004 present a regionally coherent picture of annual streamflow. Trends were negative in southern and eastern region and generally positive elsewhere (Stahl et al. 2010). Changes are highly coherent across the continent and are generally congruent with the hydrological responses expected from future climatic changes, as projected by climate models.

An analysis of probabilities of high and low stream flow for 18 European river basins in the 21st century concludes that Nordic basins have the highest probability of high flow threshold violation in Europe, mainly during winter; in Central and Southern European basins, the probability of low flow threshold violation is highest in summer. More frequent single extreme events seem at least partly responsible for an increase in low or high flow conditions (Weiß 2011).

Abrupt changes in AT can change biogeochemical processes in the catchment, thus altering water chemistry. These changes become apparent when discharge patterns change. Overall abruptly rising temperatures can cause ecosystems to become more sensitive to further changes in precipitation (Temnerud and Weyhenmeyer 2008).

Many lakes are oversaturated with carbon dioxide (CO_2) emitted to the atmosphere, often attributed to the oxidation of dissolved organic carbon in the lake (Duarte and Prairie 2005, Tranvik et al. 2009). Recently Maberly et al. (2013), using data from 20 lakes and 26 years, were able to show that CO_2 is directly delivered from inflowing streams and not produced by lake degradation, suggesting that CO_2 emission to the atmosphere is not necessarily a symptom of heterotrophic lake ecosystems but a reflection of the productivity of the catchment. Any alterations in catchment management and concomitant effects of global warming will affect future CO_2 concentrations in lakes and their emissions to the atmosphere, which add up to the CO_2 emissions emanating from rivers and streams (Butman and Reymond 2011, Melack 2011).

Attempts to model the effects of climate change on catchment processes are numerous, classified as empirical–statistical, physical, or conceptual models. Two examples for conceptual models are the generalized watershed loading functions (GWLF) model (Schneiderman et al. 2010) and the Soil and Water Assessment Tool (SWAT) model (Glavan and Pintar 2012).

Consequences: stratification, mixing, ice duration, loading, and biological changes

The impacts of a warmer world impose a multitude of consequences on inland waters. Physical variables and related parameters such as mixing, stratification, winter conditions, ice phenology, among others, are primarily affected. Environmental changes then influence chemical variables, external and internal nutrient loading, and hence eutrophication. Ultimately these changes impact aquatic organisms at all levels, affecting food web structure and biodiversity.

The effects of a warmer winter climate on the variability of physical and chemical conditions analysed for 77 Swedish lakes for 1988–2005 indicated significant earlier ice breakup dates and higher nitrate concentrations along with increasing winter AT. Variability of nitrate concentrations increased in spring and early summer between and within lakes, affecting primary production (Weyhenmeyer 2009). The formation and decay of lake ice during winter and the resulting ice phenology are complex phenomena that largely depend on meteorological forcing functions (Leppäranta 2010, Livingstone et al. 2010b). The duration of total ice cover shortens, the timing of ice disappearance (ice-out) occurs earlier in the year, and in some cases ice coverage varies irregularly between years or disappears completely, as shown for several lakes in Europe at all altitudes. In a high mountain lake in the Tatra, Poland, for instance, ice phenology shows ice-out advancing significantly at a rate of 4.5 days per decade during 1971–2010 (Pociask-Karteczka and Choinski 2012). These changes in winter conditions consequently affect winter assemblages of phytoplankton and zooplankton (e.g., Dokulil and Herzig 2009). River lake ice also declines at increased rates. The duration of ice cover in rivers of the Baltic region is decreasing. The ice-covered period has been declining by 2.8 to 6.3 days per decade during the past 30 years (Klavins et al. 2009). According to the European Environment Agency (EEA 2013), the duration of ice cover on European lakes and

rivers has shortened at a mean rate of 12 days per century over the last 150–200 years. Projected changes in cold season temperatures and associated changes in ice phenology for 2020 and beyond imply trends largely mirroring trends in global warming (Prowse et al. 2007, 2011). In almost all cases, ice regimes are shown to be strongly correlated to large-scale atmospheric circulation processes over the North Atlantic (e.g., by the NAO).

The projected increase of winter precipitation in southern Europe will increase nutrient loading to lakes and contribute to a climate-induced eutrophication (Jeppesen et al. 2010). The impact will be proportional to external loading and the in-lake nutrient concentration and therefore will be more pronounced in waters of lower trophy. Retention time is of minor importance as evidenced from chemical and phytoplankton results from 2 stratified lakes in Northern Italy (Salmaso and Cerasino 2012).

Several variables, such as winter air and spring SWT, spring epilimnetic phosphorus, and hypolimnetic oxygen, varied coherently in several deep lakes south of the Alps (Garda, Iseo, Como, Lugano, and Maggiore). The synchronization originated from effects of the winter climate triggered by atmospheric modes relevant for the Mediterranean (East Atlantic and East Mediterranean pattern) rather than the NAO. Spring water renewal and nutrient replenishment were of prime importance for the interannual variation of the lakes (Salmaso et al. 2013). The depth reached by convective mixing has gradually been reduced in recent decades, and in the future a greater quantity of energy derived from external forces will be required to initialise complete mixing (Ambrosetti et al. 2010). In some deep, temperate lakes, future climate warming will likely inhibit complete mixing, even during intense wind storms (Rempfer et al. 2010). Reduced turnover rates, however, will favour harmful filamentous cyanobacteria; mixing represents the main mechanism of population control because toxic filamentous cyanobacteria are less affected by predator consumption (Posch et al. 2012).

In addition to mixing, thermal stratification of lakes during summer is most directly influenced by climate warming, as shown by numerous studies. In Mondsee, a monomictic oligo-mesotrophic, deep stratifying lake in Austria, the onset of stratification has advanced, on average, by 20 days since 1982, and the average period of thermal stratification is now 26 days longer (Dokulil et al. 2010). In lake Erken, a shallow lake in Sweden, the timing of the end of summer stratification has advanced, on average, by 28 days into autumn since 1962 (Arvola et al. 2010). Using the strength of Schmidt stability as an index for the degree of stratification, Dokulil et al. (2010) were able to show that the onset of stable temperature profiles

occur later in years with negative NAO indices during January to March (cold winters) and are significantly advanced in years with positive winter indices. The greatest difference in the timing of stability onset was 50 days. Similarly, maximum stability was reached earlier in years with negative NAO indices and advanced significantly, up to 60 days, in positive winter NAO years (Dokulil et al. 2010).

Potential consequences of global warming on bacterial metabolism associated with fine sediment particles from 3 Danish lowland streams were tested by Sand-Jensen et al. (2007). Results indicated no differences in temperature dependence, and parameters of bacterial respiration proved to be independent of season, stream site, or degradability of organic matter. If stream temperatures increase by 2.2 to 4.3 °C as a regional warming scenario for 2071–2100, predicted mean annual bacterial respiration will likely increase by 26–63% and bacterial production by 18-41%.

Inland water resources will also be under pressure by elevated amounts of humic content and, consequently, increased water colour, a phenomenon called "brownification," an effect particularly pronounced in Northern European countries. Increased temperature led to an earlier peak in phytoplankton and changed the relative timing of different zooplankton groups. Elevated water colour however reduced chlorophyll *a* concentrations in mesocosm experiments by Nicolle et al. (2012). Increased colour resulted in stronger top-down control of algae by zooplankton, and no mismatch between primary producers and grazers was found. Overall, results indicate an earlier onset of plankton spring growth in shallow lakes in the future with a stronger top-down control of phytoplankton by zooplankton grazers. Another mesocosm experiment combining a 3 °C increase in temperature and a doubling in water colour by Ekvall et al. (2013) showed an increased response of the cyanobacterial species *Microcystis botrys*, although total cyanobacterial biomass remained unaffected. Moreover, the species variability was responsible for a more than 300% higher microcystin concentration in the future scenario compared to the control. Such species-specific responses may significantly affect bloom toxicity, biodiversity, and ecosystem functioning in a warmer world.

Cyanobacteria are in general of major interest because their potential toxicity might have negative effects on the environment, health, economics, and society. Summer blooms of *Microcystis* species are often boosted by high temperatures combined with reduced wind speed, which enables better growth for this genus with a high temperature optimum (Jöhnk et al. 2008). Excessively warm and cold periods of AT can alter the biomass of cyanobacteria in either direction, but Gallina et al. (2011)

found these temperature extremes did not lead to cyano-bacterial dominance in 5 perialpine lakes of different trophic status. Both extreme hot and extreme cold events generate a loss of diversity among cyanobacteria.

Phytoplankton spring peaks have already advanced in a variety of aquatic systems induced by climate warming. The timing and onset of growth of *Daphnia* populations has also been affected, but the mechanisms for these shifts are different. The earlier transition from strong to weak vertical mixing is decisive for phytoplankton development. For *Daphnia* growth, however, the increase in water temperature is more important. A mismatch in timing of the two processes may thus interrupt food web interactions in spring (Peeters et al. 2007). The early summer period of algal suppression by herbivores resulting in the clearwater phase is a regular and distinct feature in temperate lakes. The timing of this food web interaction has advanced by approximately 2 weeks in some central European lakes within the last 30 years. Faster population growth of herbivores in warmer water is strongly related to climate dynamics over the North Atlantic. The NAO causes a striking temporal coherence of a food web interaction by synchronizing plankton succession over long distances (Straile 2002). In Mondsee, Austria, the timing of the phytoplankton spring peak has advanced by almost 30 days since 1982 and is significantly related to the mean January–February NAO indices. The more positive this index, the earlier maximum phytoplankton development was attained (Dokulil et al. 2010). In more northerly Lake Erken, Sweden, Weyhenmeyer et al. (1999) related the mean advancement of the phytoplankton spring peak by 30 days to earlier ice break up and to the NAO index of March.

Analysing long term data from a 58-year period (1945–2003) for 4 lakes in the English Lake District, Feuchtmayr et al. (2012) were able to show that the timing of the phytoplankton spring peak not only advanced, but species phenology reacted differently. The timing of the *Asterionella formosa* maximum advanced (see Fig. 14.6 in Nõges et al. 2010b) but was delayed for *Aulacoseira* spp. and *Cryptomonas* spp. Among several driving variables tested, temperature had an inconsistent effect on species. In Windermere, *A. formosa* and *Aulacoseira* spp. advanced with temperature while *Cryptomonas* spp. was significantly delayed in 3 lakes. Overall soluble reactive phosphorus concentration had a more consistent effect upon phenology.

Model projections for Windermere, England, using up to a 4 °C temperature change and a proportional increment of 0.5–2.0 in nutrient load, indicate minor effects on spring diatom biomass, but the peak occurred 2–3 days earlier per 1°C (Elliott 2012). Modelled cyanobacteria responded positively to both treatments and dominated

summer and autumn. Temperature alone mainly affects the phenology early in the growing season.

The thermal structure of lakes can be influenced by feedback mechanisms from phytoplankton via light attenuation. Biomass affects vertical light attenuation and therefore also affects vertical short-wave radiation. High light attenuation results in stronger stratification, shallower thermoclines, higher surface temperatures, and reduced heat content during the heating phase. Thermal structure was particularly sensitive to changes in the attenuation coefficient below a value of 0.5 m^{-1}. Model predictions of these theoretical mechanism correspond to findings from Lake Constance (Rinke et al. 2010).

Expanding the Plankton Ecology Group (PEG) model from the 1980s, Senerpont Domis et al. (2013) forecast a switch from a seasonal pattern with 2 plankton biomass peaks, one in spring and one in summer, to one with a single but longer and larger biomass peak as nutrient loading increases, with associated higher populations of zooplanktivorous fish. Climate change will exacerbate these trends because more precipitation in the catchment and higher internal nutrient release as a consequence of warming will increase nutrient loading.

Temperate freshwater systems are often dominated by various *Daphnia* species, which play a key role in the food web. Climate warming can influence *Daphnia* population dynamics, altering predator–prey interactions and algal biomass control. Results of several studies reviewed by Wojtal-Frankiewicz (2012) indicate that warming has direct and indirect effects on *Daphnia* biology and ecology via its influence on their life history processes (metabolism, growth, reproduction) and the properties of their habitats. The plasticity of daphnids in terms of adaptive responses is generally high and includes phenotypic adaptations and changes in genotypes, although it also depends upon the strength of selection and the available genetic variation. The seasonal timing and magnitude of temperature increases are important for seasonal biomass fluctuations of *Daphnia* and similarly influence the potential synchrony of daphnids and phyto-plankton succession (the timing hypothesis). Even a minor warming during short but critical seasonal periods can cause factors that disturb *Daphnia* population dynamics to coincide, which may destabilize lake food webs by decoupling trophic interactions. Both winter and spring are important critical periods for determining future seasonal fluxes of *Daphnia* spp. and, consequently, the timing of the clear-water phase and the occurrence and duration of *Daphnia* midsummer decline.

Winter conditions may also affect the impact of fish predation on daphnids during summer; however, the effects of global warming on *Daphnia* population dynamics and on ecosystem functioning are often difficult

to predict due to their complexity and the presence of both antagonistic and synergistic drivers. Thus, the diverse responses of daphnids to climate anomalies depend on both biotic (e.g., predator abundance and seasonal phytoplankton succession) and abiotic (e.g., hydrodynamics, intensity and duration of thermal stratification, trophic state, and geomorphology) factors of lakes, which are directly influenced by weather changes. Such influences have been analysed by George and Hewitt (2006) and others in Esthwaite Water, a thermally stratified lake in England. The *Daphnia* produced 3 cohorts during the year, and the strength of the cohorts was determined by year-to-year variations in the physical characteristics of the lake and the abundance of edible algae. All evidence suggests that the dynamics of the *Daphnia* in the lake are strongly influenced by seasonal variations in the mixing regime, the recycling of nutrients, and the episodic growth of edible algae.

Winter conditions, particularly extremely cold or warm, may play an important role in shaping phytoplankton spring peaks and *Daphnia* spring development in deep lakes (Straile et al. 2010). In the temperate eutrophic lake Müggelsee, Germany, fast-growing spring plankton diatoms and *Daphnia* populations showed a significant and synchronous forward shift by about 1 month, induced by concurrent earlier ice break-up dates (Adrian et al. 2006). The slow-growing summer zooplankton species with longer and more complex life cycles (copepods, larvae of the mussel *Dreissena polymorpha*) do not display such synchrony. Aquatic food web models using small, simple systems can be used to assess climate change scenarios on a larger scale (Vadadi-Fülöp et al. 2009). All results point to decreasing abundance and changes in the seasonal timing of 3 zooplankton species by the scenarios tested.

Fish play a key role in the trophic dynamics of inland waters. Complex changes in fish assemblage structure may be expected with climate warming due to direct effects of temperature and indirect effects operating through eutrophication, water level changes, stratification, and salinization. In a review of published long-term data from 24 European lakes covering a north–south gradient from Sweden to Spain, Jeppesen et al. (2012) report profound changes in fish assemblage composition, body size, and/or age structure during recent decades and a shift toward higher dominance of eurythermal species associated with an increase in temperature of about 0.15–0.3 °C per decade, The abundance of Arctic charr (*Salvelinus alpinus*) has decreased in the majority of the lakes as well as the harvest of brown trout (*Salmo trutta*), both cold-stenothermic species. Similarly, the warming of streams and rivers observed by Hari et al. (2006) resulted in an upward shift in thermal habitat for brown trout

populations. In general warmwater fish are stimulated by climate warming while coldwater and coolwater fish are negatively affected, as shown by Comte et al. (2013), who summarising observed and predicted trends on the distribution of freshwater fish.

Fish communities in temperate regions are usually dominated by coldwater species with physiological optima below 20 °C. These communities are vulnerable to displacement by invasive nonnative coolwater species (optima 20–28 °C) or even warmwater species (optima >28 °C) when temperatures rise. In an example from Wales, 24 nonnative introduced and established fish species were analysed for their response to climate change in 2050 (Britton et al. 2010). Results show that several fish species will benefit from elevated temperature regimes, including common carp (*Cyprinus carpio*) and European catfish (*Silurus glanis*). The ecological consequences from an invasion of *C. carpio* are potentially severe because of possible habitat destruction, macrophyte loss, and increased water turbidity.

The International Decade "Water for Life" 2005–2015 has designated freshwater biodiversity as priority target for conservation. Within this framework, 5 major threats are defined: overexploitation, water pollution, flow modification, destruction or degradation of habitat, and invasion by exotic species. Population declines and reduction of freshwater biodiversity have occurred through combined and interacting influences of these and other threats. Biodiversity therefore is perhaps the ultimate conservation challenge in Europe and worldwide to save intact inland waters (Dudgeon et al. 2006).

Species under stress from climate warming can respond in several ways: by moving to track climatic conditions, stay in place and evolve to the new climate, or go extinct. By far the most common response is movement that tracks climate change. The phenology and distribution of freshwater plants and animals have shifted in the last 30–40 years in the direction predicted by climate scenarios. General responses of biodiversity to global warming further include amplified metabolism of organisms, increased productivity of populations, changes in the geographic range of species, increased turnover of species composition, and increased or decreased species number (Vescovi et al. 2009). Combined with oligotrophication, however, climate warming may also lead to an increase in spatial heterogeneity with depth in the water column of deep lakes, which may lead to greater richness in phytoplankton and zooplankton (Pomati et al. 2012).

Climate change is a major driver for changes in species distribution, invasion, contracting or expanding territories, and species loss, but other factors such as habitat loss may be equally important (Pimm 2008). Adaptation and management of freshwater biodiversity must consider

that inland waters are strongly influenced by anthropogenic and other activities in the wider landscape. These impacts on inland waters are most often indirect and, although they pose a challenge for conservation, are not included in current legislation, which usually focus on the water body itself (Clark 2009). Precise information on the distribution for most freshwater taxonomic groups is currently not available, complicating predictions on geographical shifts of freshwater species in response to climate variables. The requirements of freshwater organisms must, however, be considered when delineating protected areas in a warmer world or to estimate the degree protected areas accommodate freshwater biodiversity (Heino et al. 2009).

Conclusions

The impact of climate warming on inland waters is such a diverse field that the manifold facets extend from long distance weather and climate influences over watershed-related processes to molecular and genetic variations at the (sub)species level. This extremely wide field has not been fully covered here. Effects of climate warming are direct and indirect. Indirect effects are mainly mediated through processes impacting the catchment that result in hydrological changes and increased nutrient loading. The emerging picture clearly indicates warming of most inland waters, streams, rivers, lakes, and groundwater at all altitudes. Temperatures are affected at all depths in deep lakes, in turn influencing mixing patterns, onset and duration of thermal stratification, and winter ice conditions. As a result, many internal lake processes such as nutrient regeneration, oxygen concentration, and others are affected; consequently, all organisms living in fresh-waters must adapt to new situations. The relations and impacts between forcing functions and organisms have been summarised several times in different conceptual diagrams (e.g., Nõges et al. 2010b, Dokulil and Teubner 2011). Climate warming can be related to local weather situations and to long-distance climate influences.

Acknowledgements

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