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Chapter 3

OLD WINE IN NEW SKINS - EUTROPHICATION RELOADED: GLOBAL PERSPECTIVES OF POTENTIAL AMPLIFICATION BY CLIMATE WARMING, ALTERED HYDROLOGICAL CYCLE AND HUMAN INTERFERENCE

Martin T. Dokulil
EX Institute for Limnology, Mondsee, Austria
Assoc. University Vienna, Dept. Limnology, Vienna, Austria

ABSTRACT

Natural or anthropogenic enrichment of surface waters through input of nutrients, commonly referred to as Eutrophication, is essentially a catchment related process. The relative importance of different hydrological pathways in the water shed are therefore of crucial significance. Although eutrophication has a rather long history, the problem and its implications became particularly apparent in the mid-20th century as a consequence of population density, urban development, tourism, industry and agricultural practices. To maintain sustainable human societies profound water management was and is required including concepts to restore or rehabilitate surface waters and to prevent further deterioration. Mitigation of nutrient input was successful in many regions but failed or responded slowly in others, often as a result of in-lake processes. The growing water demand and the lack of clean water in large parts of the world necessitate elaborate models in the near future particularly under warmer climate scenarios. In a warmer world many consequences of eutrophication will potentially be amplified. Interaction of climate change with eutrophication will proliferate harmful algal blooms (HABS), spread infectious diseases, changes pathogen communities and favours microparasites among several other abiotic and biotic components affecting ecosystems. Persistent eutrophication may exceed ecological thresholds and lead to regime shifts. The symptoms of cultural eutrophication will certainly worsen when global temperatures increase and human impact intensifies further. Concepts and models are needed for future mitigation specifically for developing countries of the inter-tropical zone because initial attempts at applying temperate zone control measures in these regions have been largely unsuccessful.

Keywords: Nutrient input, lakes, rivers, algal blooms, catchment

INTRODUCTION

Eutrophication is perhaps the greatest threat to water quality worldwide particularly in developing countries. It affects all types of inland waters, rivers and streams. More recently a new paradigm on coastal eutrophication emerged (Duarte 2009). This new paradigm emphasizes its global dimension and the connections with other global environmental pressures. Population growth, economic development and lifestyle changes have added to the problem (Ansari 2011). The massive impact of cultural eutrophication on natural waters in general, and on water availability, water quality and water usage has generated an enormous volume of literature on causes, consequences, monitoring and management. It is the aim of the present review to summarize the state-of-the-art in eutrophication research in the context of climate warming.

DESCRIBING EUTROPHICATION

Eutrophication is defined as the enrichment of surface waters by inorganic plant nutrients, mainly phosphorus and nitrogen, as a result of slow natural or human induced accelerated processes. This nutrient load originating from the water shed increases productivity of the receiving waters (Dokulil 2013a, b). Hence, eutrophication essentially is a catchment orientated process. Phosphorus has generally been identified as the most important nutrient limiting productivity in inland waters (e.g., Vollenweider 1968, Likens 1972, Schindler 1974, 1977, OECD 1982). Nitrogen limitation has recently attained more attention evolving into a controversial discussion (Howarth, & Marino 2006, Schindler et al., 2008, 2012, Sterner 2008, Lewis & Wurtsbaugh 2008, Dolman et al., 2012). Nitrogen limitation seems responsible for macrophyte decline justifying the control of both phosphorus and nitrogen, at least in shallow lakes and estuaries (Moss et al., 2013). Phytoplankton biomass is affected by an increase in inorganic nitrogen due to atmospheric deposition in unproductive lakes in the northern hemisphere (Bergström & Jansson 2006). The authors conclude that these systems are limited by N in their natural state. Dai et al., (2012) tested the hypothesis that ammonia regulates the succession of cyanobacterial blooms. They could show that ammonia can be an important factor to determine the distribution of common algal species and cyanobacterial bloom in freshwater systems.

In addition, it must be emphasised that some authors see eutrophication also as an increase in the supply of organic matter (e.g., Nixon 2009). This view might be rectified when dealing with coastal or marine eutrophication which certainly needs viewing ecosystem changes on a larger scale. In general and in almost all textbooks however, the term eutrophication is used to describe the increase in concentrations of plant nutrients in aquatic ecosystems (e.g., Harper 1992, Mason 2002). Besides the already above mentioned phosphorus and nitrogen, silicon, iron or manganese are sometimes cited as potential limiting substances.

The nutrient status and/or productivity of a body of water is commonly quantified by a range of variables characterising water quality (Figure 1) The continuum of trophic levels is divided into several levels ranging from very poor in nutrients (ultra-oligotrophic) to over-saturated with nutrients (hyper-eutrophic). For more details refer to e.g., OECD 1982; Scholten et al., 2005; Andersen & Conley 2009; Ansari et al., 2011 among the vast literature in the field. Nutrient enrichment enhances primary productivity resulting in growth and finally excessive development of phytoplankton, sessile algae and macrophytes (Figure 2). When nutrient loading is small, phytoplankton, submersed macrophytes and sessile algae on submersed vegetation prevail. As loading increases, macrophytes decline and phytoplankton gaines more and more importance. At further increased nutrient input, phytoplankton dominates the ecosystem after submersed macrophytes have vanished. At very high nutrient concentrations, emergent or free floating vegetation competes with phytoplankton (Figure 2). This sequence is often associated with a regime shift (Figure 3) when ecological thresholds are exceeded leading to the existence of multiple stable states (Scheffer 1998, Dokulil & Teubner 2003, Dokulil et al., 2006, 2011).

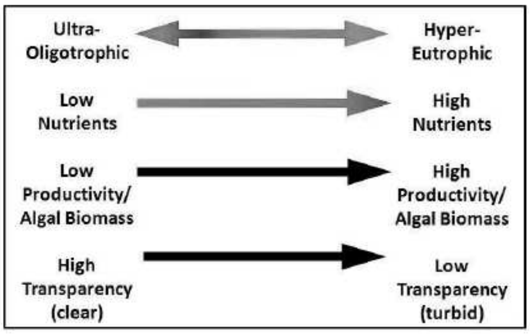

Figure 1. Conceptual diagram showing the gradients of nutrients, productivity, algal biomass and transparency in the trophic continuum from ultra-oligotrophic to hyper-eutrophic. Nutrients, productivity and biomass increase while transparency decreases at higher trophic status.

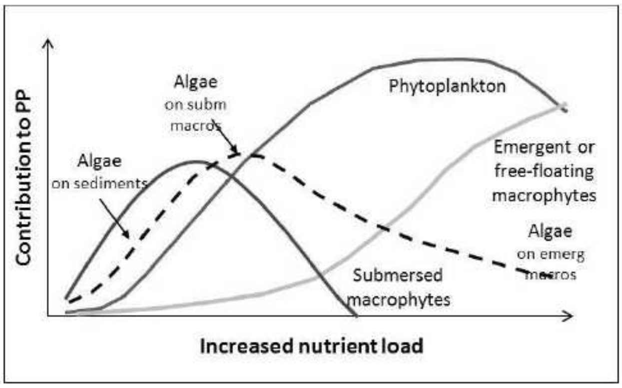

Figure 2 Conceptual illustration of the contribution of various vegetation forms to primary productivity as nutrient load increases. Submersed vegetation, algae on sediments dominate at low nutrient loading. As loading increases, phytoplankton contribution becomes dominant until at very high loading emergent or free-floating vegetation co-dominates.

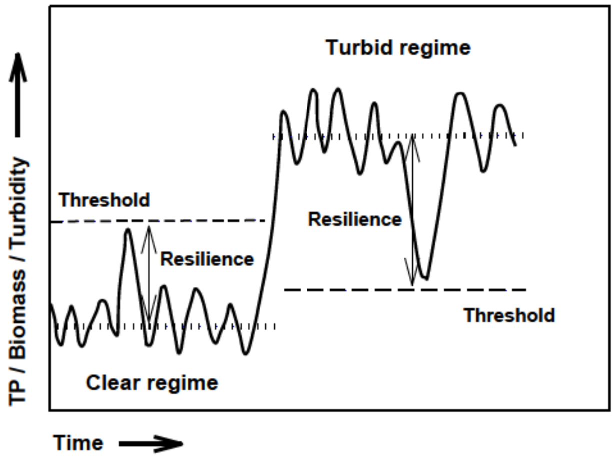

Figure 3. Eutrophication as a regime shift. The fluctuating line indicates the concentration of certain variables such as total phosphorus (TP), algal biomass, turbidity etc. Dotted lines show the means for the clear-water regime (oligotrophic) and turbid regime (eutrophic). The thresholds for both are shown by dashed lines. Distance of the mean to the thresholds is interpreted as resilience. Modified from Carpenter (2003).

History of Cultural Eutrophication

Anthropogenic eutrophication is perhaps the oldest environmental impact imposed on the biosphere by man. During the pre-agricultural hunting and picking stage total human population was small and eutrophication insignificant. As a consequence of agriculture, settlement and increasing population growth cultural eutrophication became progressively apparent. Eutrophication is thus not a recent phenomenon. It has continuously accompanied human activities in a variable but steadily increasing degree. Locally cultural eutrophication can have been far more significant in the past than today.

Cultural eutrophication has fundamentally changed the cycling of carbon, nutrients and water. The circulation within small regions characteristic in earlier days was changed and the circles were first *opened* when natural space and times scales were exceeded, and subsequently *widened* to global scales. Biogeochemical cycles are significantly changed nowadays. Stored carbon particularly from fossil fuels is reassigned with the consequence that the atmospheric CO_2 concentration has increased from 240 to 380 ppm since the beginning of the industrial period. Phosphorus is used in large amounts. Nitrogen gas is fixed from the atmosphere in similar amounts as nitrogen fixers and the hydrological cycle is altered in many ways. This fracturing of the original cycling, introduces a new, global cycling pattern that changes the overall functioning of the globe. Human population growth and increased eutrophication are therefore two aspects of the same cause (Wassmann & Kalle 2004).

SOURCES, SYMPTOMS AND RESTORATION

The causes of enhanced nutrient input to surface waters are manifold but can be divided into natural and anthropogenic components. Natural eutrophication usually is a slow geochemical process affecting lakes over long periods of time. A lake or reservoir can be naturally eutrophied for instance when situated in a fertile area with naturally nutrient enriched soils. Human influences are commonly categorised into two classes, point and non-point sources (Table 1).

Table 1. Point and nonpoint sources of inputs to receiving waters

• Point Sources
• Wastewater effluent (municipal and industrial) • Runoff and leachate from waste disposal sites • Runoff and infiltration from animal feedlots • Runoff from mines, oil fields, unsewered industrial sites • Storm sewer outfalls from cities with a population >100,000 • Overflows of combined storm and sanitary sewers • Runoff from construction sites >2 ha
• Nonpoint Sources
• Runoff from agriculture (including return flow from irrigated agriculture) • Runoff from pasture and range • Urban runoff from unsewered and sewered areas with a population <100,000 • Septic tank leachate and runoff from failed septic systems • Runoff from construction sites • Runoff from abandoned mines • Atmospheric deposition over a water surface • Activities on land that generate contaminants, such as logging, wetland conversion, construction, and development of land or waterways

A point source is a single identifiable localized source of nutrient discharge which has a small extent, distinguishing it from other geometries. The most significant point sources for nutrient loads to waterbodies are generally domestic, farming or industrial wastewater runoffs or treatment plant effluents, discharged directly to the water body or to a tributary in the catchment. Overflows of combined storm and sanitary sewers are a particular problem during severe rainfalls or thaw periods.

Non point sources originate from diffuse, not easily recognizable origin. Diffuse runoff from the catchment primarily depends on land use, urbanisation and catchment morphometry. Prime sources are agriculture including irrigation return flow, atmospheric deposition and runoff from sealed surfaces.

The symptoms generated by enriched nutrition are summarized in Table 2. Enhanced nutrient availability results in an increase in productivity leading to higher algal biomass and/or submersed macrophyte vegetation. As a consequence, biomass of consumer species may increase. When phytoplankton dominates water transparency decreases. Persistent high nutrient load leads to regime shifts which change species composition, reduces biodiversity and can culminate in algal blooms, often inedible to water organisms or even toxic. The high autotrophic biomass effects mineralization ultimately causing oxygen depletion, incidence of fish kills and therefore reduction in harvestable fish biomass. Moreover, the perceived aesthetic value of the water body largely declines (e.g., Smith & Schindler 2009).

Restoration measures can be divided operationally into external and internal restoration management practices. External techniques have a preventive character eliminating largely the causes of nutrient loading to surface waters. Controlling internal processes is curative only removing mainly the consequences of eutrophication. Both approaches can be used in parallel but external measures are usually more important in any case. Without controlling the causative factors, at least to a large extent, restoration will never be successful and sustainable (Beklioglu et al., 2011).

External reduction of nutrient inputs can be achieved using a number of concurrent approaches. Land use management in the catchment can be modified by changes in the nature of the crop, regulation of fertiliser application, treatment of animal farms and their wastes etc. Management of point source discharges primarily depend on sewage treatment including P-removal and/or N-removal. Political controls can help in addition by e.g., setting effluent standards, regulation of P in detergents, and controlling sewage collection and treatment. More rarely, inputs may be diverted from the catchment, a method not widely available. In certain instances, 'upstream' sedimentation may be applied using a chain of lakes or an impoundment. In some cases, phosphorus entering a reservoir or impoundment is removed by settling out in a 'pre-reservoir' (e.g., Harper 1992, Mason 2002).

Numerous methods are available for the treatment of eutrophication symptoms within lakes. Internal management techniques reduce nutrients by flushing, dilution, or hypolimnetic withdrawal (Olzsewski tube), an artificially installed water outlet designed to siphon anoxic water from deeper water layers. In shallow lakes, sediment removal or sediment dredging often proved useful, an expensive technique further complicated by sludge transport and disposal. Inactivation of nutrients can be achieved by flocculation using alum or ferric chloride. In addition, sediments can be oxidised with calcium nitrate reducing internal loading of nutrients from lake sediments (Dokulil et al., 2011). Reduction of nutrient release from the bottom sediments may also be achieved by sealing. Besides plastic sheeting, several new methods are under development including cover of sediments with a mineral soil of no nutritive value (Quandt et al., 2004, Gupta et al., 2008). Harvesting biomass to remove accumulated phosphorus is an attractive technique when aquatic plants pose a problem. In highly eutrophic tropical lakes harvesting of algae and fish has been tried but proved to be ineffective for nutrient removal (Harper 1992, Mason 2002).

Table 2. Potential effects of cultural eutrophication, caused by excessive inputs of phosphorus and nitrogen to lakes, reservoirs, rivers and coastal oceans (modified from Smith & Schindler 2009)

- Increased biomass of phytoplankton and macrophyte vegetation
- Increased biomass of consumer species
- Shifts to bloom-forming algal species that might be toxic or inedible
- Increases in blooms of gelatinous zooplankton (marine environments)
- Increased biomass of benthic and epiphytic algae
- Changes in species composition of algal assemblages and macrophyte vegetation
- Increased incidence of fish kills
- Reductions in species diversity
- Reductions in harvestable fish and shellfish biomass
- Decreases in water transparency
- Taste, odor and drinking water treatment problems
- Oxygen depletion
- Decreases in perceived aesthetic value of the water body

Management actions without nutrient reduction include artificial aeration supplying air to oxygen depleted layers, artificial destratification to maintain mixed conditions, or biomanipulation altering communities or the food web in a way to keep algal biomass small. Deepening can be an acceptable measure when shallow waters are seriously affected by sediment continuously stirred up from the bottom. Deepening can be achieved b either rising the water level or be dredging the lake bed (Mason 2002, Gupta et al., 2008).

PRESENT SITUATION OF EUTROPHICATION

The current increase of fertilizer consumption worldwide will intensify eutrophication in many countries and regions (FAO 2008). Highest consumption growth rate is expected for nitrogen and phosphorus, 5.7 and 6.1% respectively, in East Europe and Central Asia until 2012. World fertilizer consumption is expected to rise well over 2 percent/year between 2008 and 2012, equivalent to an increment of 19.3 million fertilizer nutrient tonnes. Only Western Europe expects a slight decrease in nutrient consumption of about 1%.

At the same time the demand for surface water for many purposes increases throughout the world, mainly due to population growth. As a consequence of greater industrial and agricultural production and accelerated technological development, the total impact of humans on nature is probably about eight times higher now than about 40-50 years ago. Lakes, reservoirs and rivers reflect the careless way in which society is dealing with its liquid wastes. As populations increase in density these practises become apparent to the public as a deterioration of water quality, most often eutrophication. The present major environmental problems of surface waters are summarized in Table 3.

Eutrophication in many parts of the world has deteriorated recently. Several countries have specific problems due to a large diversity of inland water types (reservoirs, tanks, ponds, relict lakes, billabongs etc.) or specific problems (water hyacinth, cultural siltation in e.g., India, uncontrolled tourist pressure etc.). Water shortage and ephemeral lakes due to eutrophication can often not provide the desired economic benefits to the population dependent on them. Coastal lakes are often seriously affected by imbalance in salinity level.

Table 3. Present major environmental problems of surface waters (modified from http:/www.unep.or.jp/ietc/publications/ short_series/lakereservoirs-3/2.asp

• Over-use of water from lakes, reservoirs and even rivers result in low water level, pronounced deterioration of water quality and adverse changes in the ecosystem
• Overuse or misuse of land and forests in the drainage areas lead to accelerated soil erosion and ultimately to rapid siltation in lakes and reservoirs
• Inefficient control of gas emissions create acid precipitation which acidifies surface waters, resulting in gradual deterioration of ecosystems and finally extinction of fish
• Contamination of water, sediment and organisms with chemicals, often highly toxic, from various sources
• Eutrophication from phosphorus and/or nitrogen loading resulting in deteriorated water quality, (toxic) algal blooms, and a decrease of biodiversity
• In extreme cases, the complete collapse of aquatic ecosystems

Water quality of Chinese lakes has deteriorated considerably in recent decades. Investigations indicate that 41.2% lakes were eutrophic in the late 1970s. The quantity of eutrophied lakes increased to 51.2% until 1992and rapidly rose to 84.5% in 2001-2005 (Liu et al., 2010). The present status of eutrophication in Australia can be found in Davis & Koop (2006). Details of coastal eutrophication worldwide are best depicted in the interactive maps of the World Resource Institute (http://www.wri.org/). Riverine loading is usually the most important pathway of nutrients into estuaries and coastal zones and exceeds the combined contribution from atmospheric deposition, point emission from cities and industries along the coast, and nitrogen fixation by marine organisms (e.g., Kotta et al., 2009).

The process of eutrophication is reasonably well understood in lakes. Currently there is no conceptual understanding of how eutrophication develops in rivers (Hilton et al., 2006). During the 20th century, N and P inputs into rivers around the world have increased dramatically as a result of intensive agriculture and industrial and municipal waste water discharges (Meybeck 1982). Increased concentrations of N and P from point and non-point sources are often accompanied by an increase in algal biomass (Hilton et al., 2002). Point (effluent) sources seem to be of greater importance than diffuse (non-point) agricultural sources for river eutrophication (Jarvie et al., 2006). The effects of eutrophication however, may be moderate because phytoplankton development is limited by short water retention times and/or light limitation due to high suspended particle load in rivers rather than inorganic nutrient concentrations. Few studies have investigated the consequences of eutrophication on Si consumption in the river itself. Using an ecosystem model, Billen et al*.*, (2007) demonstrated that increased anthropogenic N and P inputs resulted in increased retention of Si in the Seine River basin over the last 50 years. Dissolved silica (DSi) concentrations in the Garonne were negatively correlated with chlorophyll-a concentrations, suggesting uptake by algae. DSi depletion was more severe downstream of Toulouse, suggesting that eutrophication may affect DSi consumption in the Garonne River. (Muylaert et al., 2009).

Because many natural streams are net heterotrophic, Dodds (2006) proposed to divide trophy into an autotrophic and a heterotrophic state. Such a division would allow consideration of the influence of external carbon sources as well as nutrients such as nitrogen and phosphorus. Empirical results suggest that phosphorus and nitrogen are the most important nutrients regulating autotrophic state in flowing waters and that benthic algal biomass is positively correlated to gross primary production in streams (Dodds 2006).

SYMPTOMS OF CLIMATE WARMING

Climate warming occurs already. Global temperatures have increased by about 0.8°C since the mid-19th century (IPCC 2007). Several regions of the world have experienced much higher increase. Temperatures in Europe rose by 1.2°C and by 1.6°C in the Alps, twice as large as the global trend (Brunetti 2009). Schindler & Vallentyne (2008) report warming by 1-4°C in many areas in North America. These increasing temperatures tend to intensify eutrophication symptoms. Conceptually, climate-related effects on lake ecosystems can be approached using a model consisting of two components, the *landscape filter* compassing geographical position, catchment characteristics and lake morphology, and the *lake filter* including lake type, lake metabolism and all interactions (Blenckner 2005). As a conceptual framework, the two filters may help to improve lake management options by a more holistic approach considering the mechanisms of both components and their impact on lake ecosystem response.

Even natural eutrophication might be accelerated by global warming. Rising temperature increases nutrient loading by a greater rate of mineralization in catchment soils, changes in stream flow and sediment loading (e.g., Marshall & Randhir 2008, Brookshire et al., 2011). Changes in hydrology and extreme weather events associated with increasing temperature can further increase soil erosion enhancing leaching and delivery of nutrients to surface waters. Erosion and transport processes can be accurately estimated from models (e.g., May & Place 2005, Sorrano et al., 1996). Winter snowpack and glacial water sources are declining resulting in less water availability in spring and summer. This in turn has consequences for replenishment of lakes and reservoirs. Changes in seasonal river flow will change the residence time of many lakes world-wide affecting nutrient loading and ultimately phytoplankton biomass. Regions expecting a decrease in summer flow, may expect substantial changes in phytoplankton biomass (Jones et al., 2011). Increases in algal biomass and changes in composition will be in addition to any other changes owing to warming effects or eutrophication.

Warming has prolonged the ice-free season in several lakes (Livingstone et al., 2010) affecting phyto- and zooplankton under ice (Dokulil & Herzig 2009) and during spring development (Adrian et al., 1999). Reduced light limitation from ice and snow cover in ice-free years together with elevated temperatures result in phytoplankton biomass in spring 3-4 times higher than in ice-covered years (Müller-Navarra et al., 1997). Species composition shifts to diatom domination before and during spring peak (Adrian et al., 1999). Palaeolimnological records substantiate shifts to smaller centric diatoms in non-enriched and non-acidified lakes associated with warming trends (Rühland et al., 2008). As a consequence of temperature increase in winter and spring (e.g., Arvola et al., 2010, Dokulil et al., 2010) and through less rainfall in summer or dry seasons, evaporation and evapotranspiration will increase, removing water from lakes, rivers and aquifers. Climate warming is expected to raise evaporation by 20-25% in the next 30 years compared to the mid-20th century. Dropping lake levels and warmer water temperatures result in increased renewal times, concentration of the nutrients already present and longer periods of stratification. Reduced precipitation and increased evaporation was made responsible for changes in renewal time of lakes in Canada (Schindler & Vallentyne 2008). These changes have profound implications for eutrophication. The percentage of nutrient load flushed by the outflow decreases and the percentage retained in the lake increases. When lake levels drop, littoral sediments will be exposed to mineralization and nutrient release, favouring higher biomass of slow-growing, persistent phytoplankton taxa like cyanobacteria.

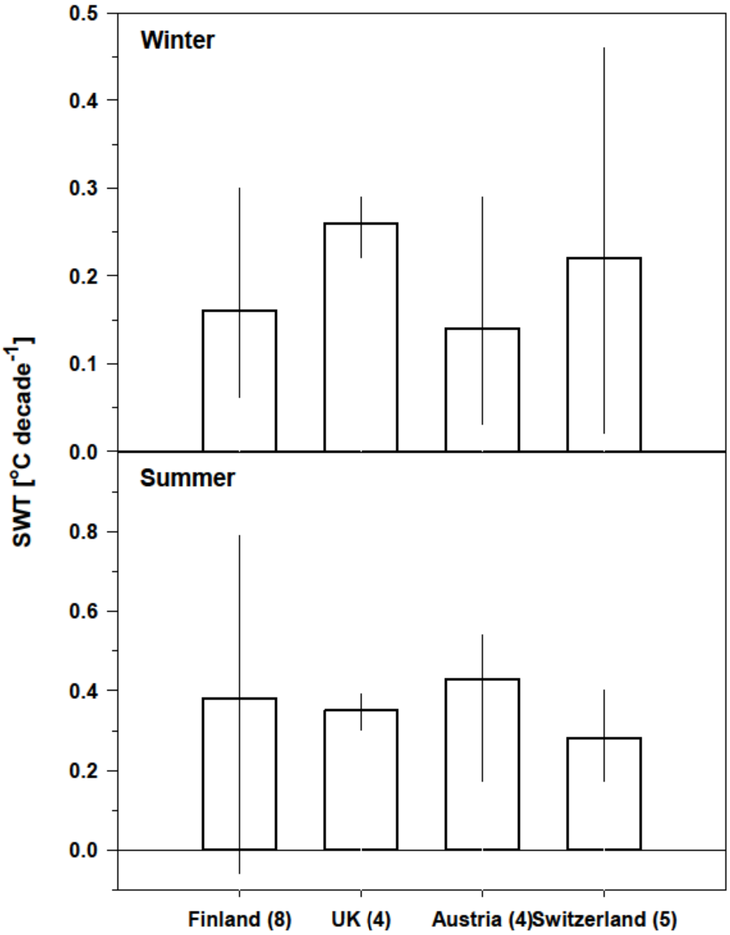

Figure 4. Long-term increases or decreases in surface water temperatures (SWT) in winter and summer in 19 European lakes pooled by country and shown as gradients of the relevant linear regressions in °C decade^{-1}. Indicated are the mean, minimum and maximum gradients in each of the countries. Numbers in brackets aside of country name indicates number of lakes included. Data from in Table 6.2 on page 90 of Arvola et al., (2010).

Annual lake water temperatures tend to increase in many regions of the northern hemisphere (Adrian et al., 2009, Arvola et al., 2010, Dokulil et al., 2010). Long-term temperature data for European lakes, differentiated for winter and summer, are presented in Figure 4. Rise of average winter surface water temperatures (SWT) range from 0.14°C per decade in Austria to 0.26°C per decade in the UK. Greatest difference between minimum and maximum gradient occurs in Switzerland, the smallest range is found in Austrian lakes (Fig, 4, winter). Increments in mean summer SWT range from 0.29°C per decade in Switzerland to 0.43°C per decade in Austria. Interestingly variability in summer is greatest in Finland with even negative increment which indicates no temperature increase or even slight cooling. Smallest variability is again observed in Austrian lakes (Figure 4, summer).

Adrian et al., (2009) report long-term gradients for January in four German lakes and for July in 12 lakes of central and northern Europe, North America and Russia. Gradients for January range from 0.06°C to 0.26°C per decade. Mean July increase is 0.5°C per decade (range -0.05 – 1.57°C). Greatest increase is observed in Stensjön, Sweden while Lake Champlain in Vermont, USA and Blue Chalk Lake, Canada exhibit no increase. Schneider et al., (2009) analysed significant warming trends of summer nighttime temperatures from satellite observations. Rates of increase range from 0.5°C to 1.5°C per decade (average 1.1± 0.2°C per decade) for 6 lakes in North America. These rates of change are an order of magnitude higher than trends observed for global sea surface temperature and about twice as high as regional trends in air temperature.

The long-term data for deep water (hypolimnetic) water temperatures assembled in Arvola et al., (2010) show essentially no change (<0.07°C per decade). A detailed analysis of bottom water temperatures from 12 lakes across Europe however, indicates increase by approx. 0.1-0.2°C per decade (Dokulil et al., 2006). Deep water temperature in Lago Bolsena and Lago Maggiore, lakes in Italy, increased at a rate of 0.3°C per decade (Ambrosetti & Brabanti 1999). In Africa, increase of deep water temperatures was 0.1-0.2°C per decade in Lake Tanganyika (Verburg et al., 2003) and 0.06°C ± 0.02°C per decade in Lake Malawi (Vollmer 2005).

Temperature anomalies calculated as the deviation of SWT 1991-2010 from SWT during the IPCC base period 1961-1990 for 23 Austrian lakes (mean, maximum and minimum) show considerable differentiation between months (Figure 5). Average anomalies are greater than 1°C for the May to August period cumulating at 1.54°C in May. The range between minimum and maximum anomaly is greatest from March to May exceeding 2°C difference with maximum anomalies larger than 2.5°C. Minimum SWT anomalies indicate cooling in January and February in one small lake usually ice-covered. Another medium sized lake tends to have lower SWT now in July and autumn (Figure 5). SWT anomalies summarized for seasons and year

(Figure 6) indicate largest variability in spring, highest anomalies in summer (mean 1.3°C) and smallest warming influence in fall (0.3°C). Average annual warming is 0.83°C for the 23 Austrian lakes as indicated by the mean yearly anomaly in Figure 6. Monthly median SWT anomalies link to the monthly median anomalies of the NAO index (Figure 7). April to September SWT anomalies associate with negative indices (r^2= 0.47, not significant). October to March temperature deviations correlate significantly linear with positive indices (r^2 = 0.84). This indicates strong positive influence of the NAO index on SWT during the winter half year (Figure 7). The overall relation can be described by a significant exponential decay function (r^2 = 0.55).

A conservative estimate of the average increase of summer SWTs from 2005 until 2050 is 2°C (1.2-2.9°C) for Austrian lakes larger than 10 km² surface area accompanied by an average increase of deep water temperature of about 0.8°C until 2050 (Dokulil 2013c). Both surface water and hypolimnetic water temperatures correlate to long-distance climate signals, such as the winter North Atlantic Oscillation (NAO) index (Livingstone & Dokulil 2001, Dokulil et al., 2006).

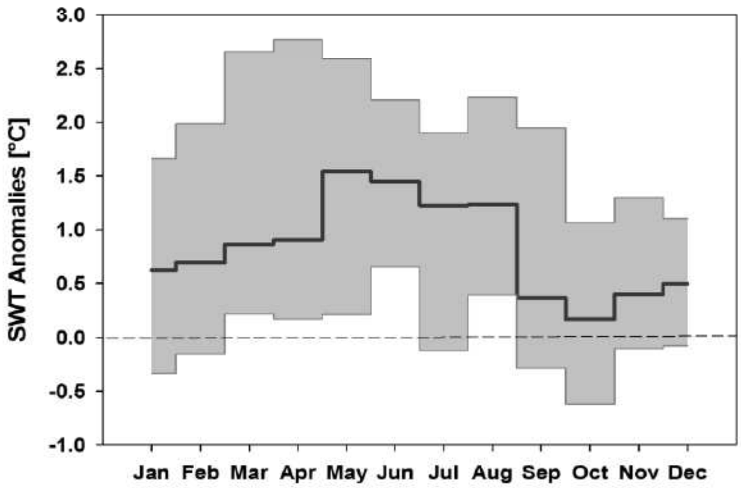

Figure 5. Long-term monthly mean, maximum and minimum anomalies in surface water temperatures (SWT) for 23 Austrian lakes in °C. Temperature anomalies were calculated from daily observations of the years 1991-2010 versus 1961-1990 (IPCC base period). Data originate from the Austrian Hydrographical Yearbooks available at www.lebensministerium. at/wasser/wasser-oesterreich/wasserkreislauf/ hydrographische _daten/jahrbuecher.html.

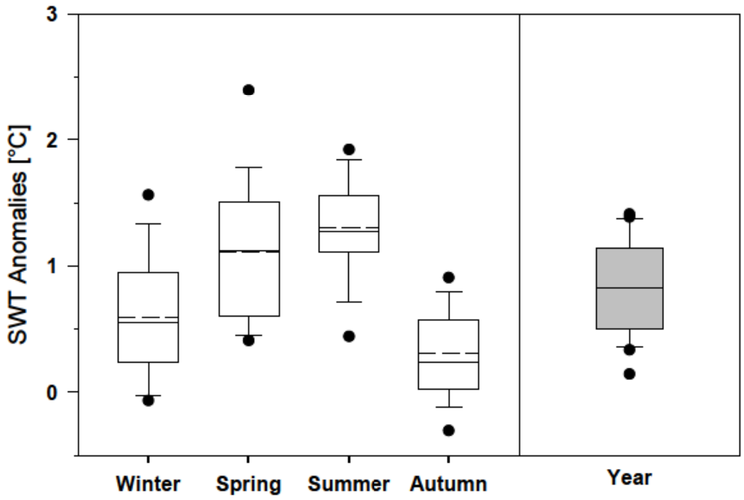

Figure 6. Long-term anomalies in surface water temperatures (SWT) summarized as seasonal and annual box plots. Each plot indicates the median, mean (dotted line), 5 and 95% c.l. and outliers. Anomalies calculated as in Figure 5.

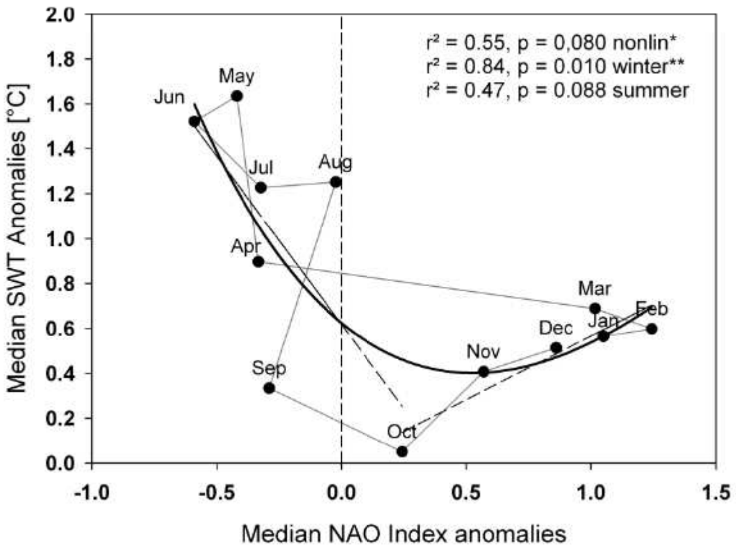

Figure 7. Monthly median long-term surface water temperatures (SWT) plotted versus monthly median anomalies in the North Atlantic Oscillation (NAO) index. Months are indicated and connected. Data are fitted by linear and non-linear regression (variance and significance indicated). Anomalies for both variables calculated as in Figure 5.

Modelling water temperature profiles from Lake Geneva indicate an increase in monthly epilimnetic and hypolimnetic temperatures of 2.3-3.8°C and 2.2.-2.3°C, respectively corresponding to 44-98% of the monthly increase in air temperature. As a consequence, lake stability increases resulting in longer thermal stratification and shorter mixing period (Perroud & Goyette 2010). In Mondsee, Austria the onset of stratification and stability will be earlier in the year while maximum stability and de-stratification occur later. Summer stratification increases by approx. 7 days per decade from an average of 150 days at present to about 176 days until 2050 (Dokulil et al., 2010). The period of thermal stratification is likely to increase from an average of 149 days for the period 1985–2011 to 196 days for 2041–2050 in Ammersee, Germany (Weinberg & Vetter (2012).

Climate warming together with eutrophication affect plants and animals that live in freshwater lakes and rivers, altering their habitat and imposing stress and disease. As temperatures increase, dominance of cyanobacteria (Dokulil & Teubner 2000), predominance of floating plants, or perhaps complete loss of underwater vegetation, will occur at lower nutrient concentrations (Kosten et al., 2009). Harmful cyanobacteria blooms will be favoured in a complex way by eutrophication and climate warming. Both processes will enhance the magnitude and frequency of bloom events (O’Neil et al., 2012, Havens 2008). In addition, neophytic cyanobacteria may invade new regions and new water bodies like the apparence of *Cylindrospermopsis raciborskii* in Europe, a species supposed to be of (sub)tropical distribution (e.g., Dokulil & Mayer 1996). Fish kills may become worse as both nutrients and temperature increase causing greater deoxygenation in the hypolimnion. Finally, lake sediments may become anoxic releasing more nutrients in summer (Moss et al., 2010).

In a warmer climate, exchange between air and water slows down, creating ‘dead zones’ depleted of oxygen. Persistent dead zones can produce toxic algal blooms, foul-smelling drinking water, and massive fish kills. Warming can displace cold-water species in streams and lakes since they may become unsuitable for e.g., cold-water fish but support species that thrive in warmer waters. Some warm-water species are already moving to waters at higher latitudes and altitudes. Earlier snowmelt, rising amounts of precipitation that falls as rain rather than snow, and more severe and frequent flooding may affect the reproduction of aquatic species. Some salmon populations have declined, for example, as more intense spring floods have washed away salmon eggs laid in stream beds. When stream flow peaks earlier in the spring owing to warmer temperatures, low stream flow begins earlier in the summer and lasts longer in the fall. These changes stress aquatic plants and animals that have adapted to specific low-flow conditions. The survival rates of fish such as salmon and trout are known to diminish when water levels in rivers and streams are dangerously low, for example (Ficke et al., 2005). The more intense precipitation that accompanies a warming world makes river flooding more likely. This flooding—combined with sewer system overflows and other problems stemming from inadequate sanitation infrastructure—can lead to disease outbreaks from water-borne bacteria. The interacting effects of climate change on the eutrophication of lakes and estuaries have recently been elegantly summarized by Moss et al., (2010). Shallow lakes and reservoirs in subtropical and tropical regions seem to be particularly sensitive to eutrophication and climate change.

Reservoirs, both large and small, are constantly increasing in numbers world-wide. Downing et al., (2006) estimated the contribution of artificial lakes >1 ha to the world's surface to over half-a-million. Since the 1950s, the volume of water sitting behind dams around the world has probably increased by about ten-fold. These reservoirs are usually created for specific purposes such as power generation or drinking water supply among others. A major problem for reservoirs and their management is the biological outcome of eutrophication, particularly enhanced phytoplankton growth. The resultant population densities cause problems for recreational users, for treatment processes in drinking water supply and directly to consumers. Strong water level fluctuations characterize many reservoirs particularly those designed for storage. The operational procedures associated with this management practice contribute to the increase of internal loading. Under many hydrological regimes, reservoirs act as nutrient sinks such promoting eutrophication. As a result, many reservoirs are affected by intense cyanobacterial blooms impairing water quality and are potential toxic. Consequently, water usage is restricted for many purposes (e.g., Naselli-Flores & Barone 2000, Naselli-Flores 2011). Climate warming is likely to affect reservoirs in several ways. Changes in hydrological conditions in the water shed will change run-off conditions and hence filling of reservoirs. Operation and management will affect water level changes, thermal structure and consequently the level of eutrophication and its consequences.

The impact of climate change and of other anthropogenic pressures on the structure and composition of phytoplankton communities on rivers remains poorly documented. Long-term changes in the phytoplankton community of the middle segment of the river Loire for instance indicate a dramatic reduction in phytoplankton abundance in the mid -1990s. Concomitantly the relative proportion of cyanobacteria increased. The total phytoplankton community however displayed increasing diversity and little change in its size structure. All these changes seem to be related to the reduction in phosphorus concentrations, as well as to changes in climate. Several anthropogenic pressures, acting at different spatial and temporal scales, lead to predictable and unpredictable changes in the phytoplankton community with consequences for the trophic network in this river. (Larroudé et al., 2013).

Climate warming can have cascading effects on ecosystems. Phytoplankton development in spring and summer in Lago di Garda, Italy for instance, is largely controlled by the spring epilimnetic TP concentration which depends on nutrient replenishment governed by the extent of spring mixing depth. Mixing depth depends on spring water temperature largely reflecting winter air temperature which is ultimately regulated by the winter fluctuations of the East Atlantic pattern. In its negative state, this climate signal transports cold air from continental Europe to the Mediterranean, favouring mixing and nutrient replenishment. The organisms which benefit most from the resulting phosphorus increase are cyanobacteria, *Planktothrix rubescens* especially (Salmaso & Cerasino 2012).

How impacts of climate warming on ecosystems will affect the interaction among organisms remains difficult to forecast. Different species can show unique responses to changes in environmental temperatures. Differences in sensitivity to vernal warming can disrupt the trophic linkages between phytoplankton and zooplankton in a deep lake (Winder & Schindler 2004). In this case, a long-term decline of a Daphnia population was associated with an expanding temporal mismatch to the spring diatom peak which has advanced by about 20 days since 1962. In the deep, stratifying lake Mondsee, Austria the phytoplankton spring maximum advanced on average by 30 days (Dokulil et al., 2010), in the shallow Lake Erken, Sweden the spring peak occurs now 30 days earlier than in the past (Weyhenmeyer et al., 1999). In both lakes, ice-out and the timing of the spring peak significantly correlates with the NAO index in late winter. Possible future consequences of climate change on lake function were summarized for Lake Erken in a Swedish case study (Blenckner et al., 2002). The long-term wax and wane of nutrient loading is usually closely resembled by quantitative and qualitative shifts of the phytoplankton community. Reorganization of algal composition and biomass drastically change in response to changes in the trophic status and lake warming. Transitions are nonlinear for phytoplankton and zooplankton indicating a hierarchical response across trophic levels (Hsieh et al., 2010, 2011). Effects of global warming however can be compensated by oligotrophication in deep stratifying lake ecosystems (Stich & Brinker 2010). In Lake Constance for example, oligotrophication is expected to further reduce soluble reactive phosphorus in the lake while current global warming scenarios predict a moderate increase in productivity. Climate induced impacts on lake functioning in summer reveal changes in the phenology of lake plankton. Phytoplankton species composition changed asynchronous to changes in zooplankton. These changes led to disruption in predator-prey relationships (Adrian et al., 2006). The longer summer stratification periods affect dissolved oxygen concentrations and nutrient accumulation in the hypolimnion stimulating the formation of phytoplankton blooms (Wilhelm & Adrian).

Understanding the interactive effects of warming and eutrophication as part of multiple anthropogenic perturbations as well as interactions among adjacent ecosystems are essential to predict changes and future states of ecosystems. Greig et al., (2012) demonstrated in an outdoor experimental study that warming and nutrient additions enhance cross-ecosystem exchange by significantly increasing emergence of aquatic insects and decomposition of terrestrial plant material.

Analysing effects of extreme climate periods such as the warm winter 2006/07 can help to elucidate possible ecosystem functioning in a future warmer world (Straile et al., 2010). Due to the warm winter situation Lake Constance did not mix completely throughout the winter but one warm winter is not sufficient to raise water temperatures in a deep lake up to those expected. Beaver et al., (2013) analysed the effect of three major hurricanes in a large lowland subtropical lake on phytoplankton community structure. The extreme weather events caused sediment resuspension and almost completely eliminated submerged macrophytes, The lower water transparency changed the long-term co-dominance of cyanobacteria and diatoms to a complete domination of meroplanktonic diatoms.

Economic Implications

Empirical investigations on the effects of climate change on resources and economies are necessary and important for the formation of national and regional climate-related policies and programs. Particularly to ensure that estimated policy and program costs conform to estimated benefits. Consistent, accurate and comprehensive assessment of the impacts of climate change on water resources and economies remains challenging. National-scale assessments cannot adequately portray the variation in regional and local water resource conditions. Climate change shall have negative effects on water resources and human health. In the 20th century however, the global average impact was positive. Impacts turn negative in most countries, rich and poor in the 21st century ultimately leading to economic effects. Water resources, decrease in biodiversity and sea level rise are the main negative impacts imposed by climate change (Tol 2010, Hurd & Rouhi-Rad 2013). In parts of the world where runoff decreases, including parts of Europe, the Mediterranean, Central and South America, and large parts of Africa water resource stress will increase as a result of climate warming. In other regions, climate change will increase runoff. Particularly in south and east Asia it might not be beneficial because the increases tend to come during the wet season and the extra water may not be available during the dry season (Arnell 2004). Recent comprehensive over-views on sustainable use of water resources are provided by Schneier-Madanas & Courel (2010) and Filho (2012).

FUTURE CHALLENGES

The loading concept proposed by Vollenweider (1975) proved very useful for mitigating the effects of eutrophication especially in deep stratifying lakes around the world. Many of these lakes have meanwhile been restored particularly in developed countries of the northern hemisphere. Shallow lakes are still much more problematic (e.g., Scheffer 1998). The number of small, shallow water bodies (<0.1 km^2) is estimated to be over 300 Million world-wide, with the majority smaller than 0.01 km^2 and their geographical distribution highly non-uniform (Downing et al., 2006). These small water-bodies however are most often of utmost significance for the populations living in the surrounding as the sole water resource in the area. Eutrophication problems in these types of lakes will become more intensive as hydrology changes. Particularly in arid and semi-arid regions, evaporation will lead to salinization rendering water unsuitable for use as drinking water and even irrigation (e.g., Salameh & Harahsheh 2011). Storage reservoirs and impoundments, as artificial constructions exponentially increasing in number world-wide, are particularly sensitive to nutrient enrichment because of their prolonged residence time, greater evaporation and intensive usage for fish production, irrigation etc.

The degree of eutrophication affects non-infectious disease (Chorus & Bartram 1999) as well as the transmission of diseases by parasites s in aquatic systems. Nutrient enrichment influences the abundance and distribution of hosts, macro- and microparasite production, and affects virulence or toxicity. Eutrophication favors waterborne infectious diseases that involve generalist parasites with simple life cycles, parasites with tolerant intermediate hosts and opportunistic pathogens in combination with stressed hosts. These situations will lead to a greater risk for disease outbreak, epidemics and mortality. In developing countries the risk of waterborne diseases infecting millions is exacerbated by the plentiful permanent surface waters such as rivers, streams, lakes, ponds dams, and irrigation schemes (Fenwick 2006).

Eutrophication not only poses a problem in the country-side, but makes it increasingly difficult to find and utilize new sources of water necessary to satisfy growing water demand for urban populations. A lack of clean water is the result especially for the poor, the residents of peri-urban and squatter areas. Emerging challenges are the delivery of drinking water supply for the growing cities, water for sanitation, recycling of wastewater, irrigation using wastewater, urban agriculture and water to feed depleted aquifers (e.g., Niemczynowicz 1999).

CONCLUSION

The understanding and management of eutrophication has evolved largely since the mid-50th of the last century. Theoretical concepts of eutrophication were developed and applied to natural systems with great success in several parts of the world. Waste water treatment and restoration techniques have been developed to cope with the causes and consequences of anthropogenic eutrophication.

Eutrophication and its control will remain one of the greatest challenges for science and management in the 21st century. The complexity of the problems will increase because of cumulative effects which combinations of stressors impose, in particular climate warming. Changes in climatic conditions, changed in hydrology and extreme events in the catchments will strongly affect eutrophication processes.

The growing world population, their increasing water demand and climatic changes will intensify problems of water quality in many parts of the world, particularly in arid, semi-arid areas as well as in the subtropics and tropics. Solutions for the eutrophication of small, shallow water bodies, reservoirs and wetlands in these regions are urgently needed.

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