Combining limnological and palaeolimnological data to disentangle the effects of nutrient pollution and climate change on lake ecosystems: problems and potential

Battarbee, R.W.[1] Anderson, N.J.[2] Bennion, H.[1] and Simpson, G.L.[1]

[1] Environmental Change Research Centre, University College London, Gower St., London WC1E 6BT, UK.

[2] Department of Geography, Loughborough University, Loughborough LE11 3TU, UK.

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Corresponding author email: r.battarbee@ucl.ac.uk.

SUMMARY

1. As long-term observational lake records continue to lengthen, the historical overlap with lake sediment records grows providing increasing opportunities for placing the contemporary ecological status of lakes in a temporal perspective.

2. Comparisons between long-term data-sets and sediment records, however, require lake sediments to be accurately dated and for sediment accumulation rates to be sufficiently rapid to allow precise matching with observational data.

3. The critical role of the sediment record in this context is its value in tracking the changing impact of human activity on a lake from a pre-disturbance reference through to the present day.

4. Here we use data from a range of lakes across Europe presented as case studies in this Special Section. The seven sites considered all

possess both long-term observational records and high quality sediment records. Our objective is to assess whether recent climate change is having an impact on their trophic status and in particular whether that impact can be disentangled from the changes associated with nutrient pollution.

5. The palaeo-data show clear evidence for the beginning of nutrient pollution varying from the mid 19th century at Loch Leven to the early and middle 20th century at other sites. The monitoring data show different degrees of recovery when judged against the palaeo-reference.

6. The reason for limited recovery is due to continuing high nutrient concentrations related to an increase in diffuse nutrient loading or to internal P re-cycling, but there is some evidence that climate change may be playing a role in offsetting recovery at some sites. If this is the case then lake ecosystems suffering from eutrophication may not necessarily return to their pre-eutrophication reference status despite the measures that have been taken to reduce external nutrient loading.

7. The extent to which future warming might further limit such recovery can only be evaluated by continued monitoring combined with the use of palaeo-records that set the pre-eutrophication reference.

Introduction

There is increasing evidence that climate change is beginning to have a noticeable effect on lake ecosystems (Kernan *et al.*, 2010). The evidence is clearest for lakes in remote arctic and alpine regions where the extent of warming has been most pronounced and where its effects are not heavily masked by the impact of other kinds of human activity; for example, pollution or land-use change (Battarbee *et al.*, 2002; Smol *et al.*, 2005). Identifying the influence of climate change is perhaps most difficult for lakes suffering from eutrophication. Increasing water temperature tends to produce symptoms identical to those created by cultural eutrophication such as increased algal productivity, increased hypolimnetic oxygen stress and accelerated nutrient re-cycling (Moss *et al.*, 2011), and climate-mediated changes in catchment hydrology and soil biogeochemistry can enhance nutrient loading (Moore *et al.*, 2010; Pierson *et al.*, 2010; Schneiderman *et al.*, 2010).

Attempts to disentangle the relative role of climate change from nutrient pollution in driving ecosystem change require the examination of long-term data. Although long-term records exist for some lakes, they are relatively scarce and are rarely sufficiently long to capture the full range of time-scales needed. However the necessary time perspective can be obtained using sediment records. Sediment records do not possess the precision and accuracy of observational data but they occur in almost all lakes and provide an integrated record of lake history. Using palaeo-data together with observational records offers the best opportunity of generating data-sets of sufficient length and quality to address these questions of ecosystem change (cf.Battarbee *et al.*, 2005). In particular, when observational time series and palaeo-records can be accurately matched for the periods of overlap, the different data types can be mutually supportive. Although observational data have their own weaknesses with respect to e.g. the representativity of spot samples within a lake water column, they play an essential role in verifying palaeo-reconstructions, and the palaeo-data can then be used to extend the record back in time, enabling the identification of longer-term trends and the definition of lake status before the impact of human activity (cf.Bennion *et al.*, 2011).

Although key changes in lake hydrochemistry and biology can often be readily identified from such combined or separate time-series it is more difficult to explain change unless matching records of potential drivers of change are also available. This is a special problem for palaeolimnology where independent data on the behaviour of explanatory variables are often missing and where the sediment record itself is often used to derive such data, for example using diatom data to reconstruct nutrient history or chironomid data to reconstruct temperature (cf.Lotter & Anderson, 2012). This inverse approach can lead to problems of circular reasoning and can present the analyst with an equifinality problem in which different drivers can exert pressures that produce identical outcomes, an issue especially relevant to nutrient problems in lakes where both cultural eutrophication and climate change can cause similar effects (Moss *et al*., 2011).

Sediment records themselves sometimes provide the potential for escaping these problems where it is possible to use multiple proxies to reconstruct trends in drivers independently from trends in response variables. The challenge is to generate multi-decadal data-sets of both driver and response variables of sufficient quality and frequency so that the causes of variation can be disentangled with statistical confidence (Lotter & Anderson, 2012).

In this paper we review these issues using data from the studies included in this special “Euro-limpacs” section and other examples from the wider literature. The Euro-limpacs sites were not selected to be directly comparable in terms of size or trophic state; they are sites for which there are both long-term monitoring records and sediment records. With the exception of Mývatn (Hauptfleisch *et al.*, 2012), all have a clear history of cultural eutrophication. Measures have been taken at all sites to reduce nutrient loading, but eutrophication remains a problem at most. They range from very small shallow sites to very large deep lakes situated from Iceland in the north to Italy in the south (Battarbee & Bennion, 2012). Our aim is to assess the extent to which changes that have occurred over recent decades in the study sites are due to climate change either directly or indirectly and whether the influence of climate change can be disentangled from that of other pressures, especially eutrophication.

Establishing sediment chronologies

The most important need in matching observational and sediment records is establishing an accurate sediment chronology. Although some lakes have annually laminated (varved) sediments that allow sediment chronologies to be derived with accuracy and precision enabling perfect sidereal matching with historical observations, most lake sediments are not annually layered. Chronologies in most instances need to be established using indirect techniques, all of which have uncertainties. For recent sediments, i.e. those accumulating over the last 100 years, the most useful technique is ^{210}Pb dating (cf.Appleby, 2001). This is a globally applicable method based on the radioactive decay of ^{210}Pb with a half-life of 22.26 years. Uncertainties include inherent systematic errors associated with the measurement of radioactivity, and lake- or core-specific errors associated with the age modelling assumptions used in the derivation of a chronology. It is important to use as many cross-checking approaches as possible, for example, by comparing ^{210}Pb chronologies with dates derived from measurements of fallout ^{137}Cs caused by nuclear weapon testing in the 1960s and by the Chernobyl nuclear explosion in 1986. Occasionally locally recorded events including tephra from known volcanic eruptions, or clay lenses from dated flood events can provide clear time markers in sediments, and the record of air pollutants (e.g. from fly ash particles and trace metals) can provide valuable time controls at the regional scale (Rose, 2001).

For many eutrophic lakes standard radiometric dating methods perform well although occasionally it can be impossible to establish a good radiometric chronology. Age-depth relationships for the cores used in this study (Table 1) and described in the papers in this Special Section are shown in Fig. 1. All except Mývatn are based on ^{210}Pb dating. For Mývatn historical tephra layers from eruptions of the Veidivötn and Grímsvötn central volcanoes were the only reliable chronological markers available (e.g.Hauptfleisch *et al*., 2012). The diagram (Fig. 1) shows the age range of the method, limiting the dating to approximately the last 120 years, a limit imposed by the half-life of ^{210}Pb. Dates are also subject to uncertainty associated with the random nature of radioactive decay and the age of the sediment (not shown in Fig. 1). Typically the standard error varies from +/- 3 to 5 years for very recent sediments to +/- 20 years for older (70-100 yr) sediments.

Temporal resolution

A second central issue that determines the detail with which observational and sediment data can be compared is the temporal resolution of the sediment samples relative to the sample frequency of observational data. Whilst this depends to some extent on the thickness of the sediment slice used for analysis, the principal controls are the rate of sediment accumulation and the degree to which the sediment is mixed by bioturbation or resuspension.

Oligotrophic lakes tend to have low sediment accumulation rates typically less than 1 mm yr^{-1} (Rose *et al.*, 2011) and therefore provide a relatively low resolution archive. However, lowland lakes, especially those suffering from eutrophication, including those considered here (Table 1), often have relatively high rates of sediment accumulation (Rose *et al*., 2011), usually > 1 mm yr^{-1} and often as high as ca. 1 cm yr^{-1}. When sediment cores from such lakes are sampled at 0.25 cm or 0.5 cm intervals, sediment cores can provide records with an approximately annual resolution (Table 1).

Representativity of palaeo-data

The principal challenge in combining monitoring data and sediment data is representativity. Monitoring data are derived from direct measurements and observation, although they can be severely restricted in space and time, whereas palaeo-data are indirect and either involve the use of proxies or rely on assumptions about the preservation and overall representativity of the biological material found in sediments. One of the reasons diatoms are useful as indicators of past ecological change is that there can be a very direct relationship between diatom communities within lakes and the diatom assemblages preserved in accumulating lake sediments. There is not always a perfect one-to-one relationship; it varies from excellent (e.g.Haworth, 1980) to poor (e.g.Ryves *et al.*, 2006). In the case studies here diatoms were well preserved at all sites where they were analysed, providing not only an integrated record of the plankton but also a record of changes in benthic communities, a community that until recently has rarely been used for lake monitoring purposes.

In Esthwaite Water (Dong *et al*., 2012a) and Loch Leven (Bennion *et al*., 2012) it was possible to compare the sediment diatom record with the diatom component of archived long-term phytoplankton samples. In both cases the results showed excellent agreement indicating that diatoms are very well preserved at these sites and can be used confidently, assuming no change in preservation status through time, to infer changes in environmental conditions in the past, prior to the start of direct monitoring.

In addition to assessing the quality of the fossil record, comparisons between observational data and sediment data enable the integrity of the sediment record to be checked. One test is to examine the extent to which known historical events are faithfully registered by the sediments, an issue of sensitivity on the one hand and resolution on the other. In non-laminated sediments, both can be compromised by bioturbation and sediment resuspension which tend to smooth out the historical record as noted above. Even at White Lough where sediment accumulation rates are >0.3 cm yr^{-1} for much of the 20th century, mixing processes make the identification of short-lived pulses difficult (Rippey *et al.*, 1997).

Although smoothing can be a helpful process in that random noise and high frequency seasonal variability can be smoothed in favour of the underlying trends, it can also lead to the elimination or muting of events in a somewhat arbitrary way. In Mývatn Hauptfleisch *et al*. (2012) show that the amplitude in the cyclical pattern in chironomid egg abundance in the sediment record is four times less than the amplitude of the observed chironomid population, and in Lake Maggiore, Kampf *et al*. (2012) show that not all the recorded floods for the lake are registered in the sediment. So in one case, the signal is muted, possibly as a result of bioturbation and in the second, signals below a certain strength threshold (in terms of magnitude and/or duration) may not be registered.

Extending the temporal range of observational time-series

Once the observational and palaeo data-sets are matched in time and the observational data have confirmed the reliability of the fossil record, the sediment record can be used more confidently to reconstruct changes in ecosystem properties beyond the time-frame of the observational record. In almost all lakes this longer temporal dimension is essential to assess how variable lake ecosystems have been or can be in the absence of human impact and thereby to define the ecological reference status of lakes required to guide their intelligent management (cf.Bennion *et al.*, 2011).

Natural variability and change

Natural ecosystem variability can be generally defined as the behaviour of an ecosystem in the absence of human impact. For lakes such variability may be related both to exogenous factors (such as climate variability) and endogenous ones (e.g. related to stochastic ecological processes or secular change associated with infilling and seral succession). However, separating change caused by natural variability from change caused by human activity can be difficult; it is a central challenge for ecosystem science (Cottingham *et al.*, 2000). Lake sediment records uniquely allow this issue to be addressed (cf.Simpson & Anderson, 2009) by examining palaeoecological data for time periods prior to the first evidence of human impact and by identifying underlying natural patterns in system behaviour that may, without sufficient temporal context for guidance, be mistakenly interpreted as a consequence of human activity.

For most temperate lakes, diatom records show that the pre-disturbance conditions are relatively stable over long periods of time. In Esthwaite Water Dong *et al*.'s analysis of a core spanning the last 1200 years showed that prior to the beginning of eutrophication in the late 19th century the lake was dominated for long periods by *Cyclotella comensis* Grunow and *Aulacoseira subarctica* (O.F. Müller) E.Y.Haworth with relative dominance switching between these taxa at the beginning and end of the Medieval Warm Period (Dong *et al.*, 2012b). The eutrophication period, however, shows continuous species turnover over the last 100 years through to the present day. And in the case of Mýváatn, a remote lake in north-east Iceland, the cyclical record of chironomid egg and Cladoceran abundances in the sediments is thought to reflect a persistent natural feature of the lake (Hauptfleisch *et al*., 2012).

Reference diatom composition

The use of palaeolimnological records to establish the baseline or reference state of a lake has been well established (Bennion & Battarbee, 2007). Although there are an increasing number of examples of lakes set in rich agricultural landscapes that have experienced enrichment as a result of human activity from as far back as the Bronze Age (e.g. Fritz, 1989; Bradshaw *et al.*, 2006; Dressler *et al.*, 2011) and even the early Holocene (Vigliotti *et al.*, 2010), the first point of detectable change for most lakes post-dates ca. 1850 AD, and in many cases is quite recent (Battarbee *et al.*, 2011). However, in no case, has this point of change been captured by contemporary scientific observation or long-term monitoring.

Table 1 shows the respective dates for the first evidence of eutrophication from the sediment record compared with the start of monitoring for the sites included in this Special Section. Except for Esthwaite Water, monitoring started only after eutrophication had been recognised as an environmental problem. Even for Esthwaite where nutrient data have been available since

1945, eutrophication had been occurring for many decades before monitoring began (cf.Dong *et al*., 2012a). The diatom data show that eutrophication began in about 1880, indicated in particular by a decline in *Cyclotella comensis* and an increase in *Asterionella formosa* Hassall but accelerated significantly after the mid-1970s, indicated by the increase in *Stephanodiscus binatus* Håkansson and Kling (Dong *et al*., 2012a). This second stage was coincident with the opening of a new sewage works in 1976 and it is represented clearly by both the core record and the observational record. Although the monitoring data capture the main eutrophication stage, the core data demonstrate that the true reference conditions are those that occurred prior to 1880 when TP (total phosphorus) is estimated to have been between 15 and 20 µg L^{-1} (Dong *et al*., 2012b). The reference flora, prior to 1880, was dominated by *Aulacoseira subarctica* and a number of fragilaroid benthic taxa, indicating much reduced phytoplankton populations in the lake and better light conditions for benthic algae than at present.

For other sites regular monitoring was initiated only after eutrophication was recognized as an environmental problem. In the case of Mjøsa, for example, monitoring began when TP concentrations were at their highest in the early 1970s (Table 1,Hobaek *et al.*, 2012). However, the diatom core data for Mjøsa (Hobaek *et al*., 2012), show very clear evidence for the beginning of eutrophication in the 1950s, marked by a decrease in *Cyclotella* taxa and an increase in *Fragilaria crotonensis* Kitton and small *Stephanodiscus* taxa. The core data show that the pre-1950 diatom flora was characterised by *Aulacoseira islandica* (O. Müller) Simonsen, *Cyclotella pseudostelligera* Hustedt and *Achnanthes minutissima* Kützing, taxa that are now re-appearing and increasing as the lake recovers.

In Loch Leven regular monitoring began in 1968 when the lake was already strongly enriched. The core data (Bennion *et al*., 2012) show a full history of eutrophication beginning in the first part of the 19th century when *A. ambigua* (Grunow) Simonsen, *S.parvus/minutulus, S. hantzschii* Grunow and *C. pseudostelligera* began to replace *Aulacoseira subarctica, Tabellaria flocculosa* (Roth) Kützing and oligotrophic *Cyclotella* taxa that characterised the pre-eutrophication flora. At White Lough the pre-eutrophication plankton flora was characterised by *Cyclotella ocellata* Pantocsek a diatom typical of nutrient-poor water (Anderson *et al.*, 2012). The start of nutrient monitoring for Piburger See in 1974 also post-dates considerably the onset of nutrient enrichment that dates to about 1920, as indicated by the rapid increase in *Cyclotella pseudostelligera* in the sediment record. Prior to this time the lake was characterised by a diatom flora dominated almost completely by benthic taxa, an indication of oligotrophic conditions. The increase in the abundance of planktonic diatoms relative to benthic taxa is commonly observed in the sediment cores of lakes undergoing eutrophication (Battarbee *et al.*, 2001) and this shift is seen in all the sites described in the papers in this Special Section.

Reference TP

Although, as shown above, it is relatively straightforward to establish the composition of reference diatom assemblages from sediment records it is more difficult to reconstruct reference nutrient concentrations. The principal technique developed for this purpose is the use of diatom-total phosphorus transfer functions (e.g. Hall & Smol, 1992; Bennion *et al.*, 1996) on the assumption that TP is a strong controlling variable on diatom assemblage composition and abundance. Despite the promise of this technique the approach has been shown to have limitations, for example when applied to shallow lakes dominated by benthic taxa (e.g.Sayer & Roberts, 2001) or lakes with very high TP (e.g.Bennion, 1994).

There have been several attempts to validate diatom-inferred TP (DI-TP) values by comparing them against measured TP data through time. In most cases to date the comparisons have been reasonably good (e.g.Bennion *et al.*, 1995; Hall *et al*., 1999; Bradshaw & Anderson, 2001). The case studies presented here offer further opportunities to test the method. Fig. 2 shows comparisons between measured TP data and DI-TP for Lake Mjøsa, Loch Leven, Esthwaite Water and Piburger See. Whereas the data for Loch Leven and Piburger See are in substantial agreement, the DI-TP data for Mjøsa systematically over-estimate the measured TP data by approximately 5 µg L^{-1} and for Esthwaite Water the match is good for the 1960s but thereafter there is a progressive disparity with DI-TP showing steadily increasing values whereas the measured data are relatively unchanging.

For Mjøsa, the DI-TP concentrations track the pattern of temporal change recorded by the monitoring programme but are almost twice the measured values. While the model applied to this lake contains relatively large, deep lakes (> 10 m maximum depth) from Northern and Central Europe, the median value for the data-set is 22 µg TP L^{-1} and the current TP of Mjøsa lies at the extreme low end of the range of values covered by the training set. Furthermore, the great depth of the basin and the strong influence of the large alpine part of the catchment distinguishes it from other lakes in the model. It appears, therefore, that the present training set may not be entirely appropriate for Mjøsa and a separate (local) training set may be required to achieve higher accuracy in DI-TP reconstructions (Hobaek *et al*., 2012).

For Esthwaite Water the overestimation by the model for the last four decades arises because of the dominance of *Stephanodiscus* spp. in the main enrichment period, comprising up to 30% relative abundance of the diatom assemblage. In the Northwest European training set these taxa occur in the greatest relative abundances in lakes with high TP concentrations and given that the data-set includes many waters with TP concentrations >40 µg L^{-1}, this results in high TP optima, mostly in excess of 100 µg L^{-1} (Bennion *et al*., 1996). Indeed the diatom model parallels the steeper rise in maximum soluble reactive P (SRP) in Esthwaite Water more closely than the gradual increase in annual mean TP (Bennion *et al*., 2005).

At White Lough, there was very good agreement between DI-TP for the period with monitoring data (1978-1982; see Rippey *et al*., 1997) but there is very little increasing trend in DI-TP values for 1900-1960, a period when there is unequivocal evidence from geochemical and reconstructed P data for an increase in nutrient loading (Anderson *et al*., 2012). An explanation for this may be the abundance of *Aulacoseira subarctica* in this lake which dominates the diatom flora as it is tolerant of the low light levels associated with turbid water and high DOC concentrations typical of Northern Irish lakes (see Anderson *et al*., 2012).

However, these site-specific explanations for the poor performance of Weighted Averaging-based DI-TP ignores the growing evidence for more generic problems with transfer functions (cf.Anderson, 2000) in situations where secondary environmental gradients, such as conductivity or alkalinity need to be accommodated. Phosphorus can be a minor explanatory variable at some sites (e.g., Hall & Smol, 1992) and/or for certain periods in lake history. In lowland lakes, for example the increased availability of phosphorus in the aquatic environment, especially after 1950, has tended to mask the influence of these secondary gradients and consequently there is often good agreement between monitored and DI-TP values during the eutrophication phase as shown by some of the sites in this study (Fig. 2) and by Bennion *et al.*(1995) and Bradshaw *et al.* (2001). However, prior to the intensive use of commercial fertilizers and phosphate detergents, i.e. during the late-19th , early 20th century, there are few or no direct measurements of TP for comparison but DI-TP values are often higher than can be realistically accepted from land-use histories and P-loading models (cf Anderson 1997). These over-estimates result from the inability of transfer functions to deal with dual gradients. DI-TP models applied prior to 1950 may be reconstructing changes in diatom assemblages driven by other variables, such as conductivity or DOC, instead of, or as well as, nutrient concentration.

Environmental and climate drivers

Changes in the biology of lakes recorded by lake sediments or by long-term observations are not always easy to explain in the absence of equivalent high quality data documenting changes in the environmental pressures that lakes face. Such data need to be compiled in detail and with accuracy to enable driver and response variables to be analysed together quantitatively. However, data on trends in driver variables are often limited, especially over the longer time-scales needed for analysis with sediment core data. Potentially they can be generated in two complementary ways, either using data derived from a sediment core itself using an appropriate proxy and/or by using independent data from historical archives or observational time-series, if such exist (Lotter & Anderson, 2012).

A principal problem in using core data is that diatom-TP transfer functions, even if deemed accurate, cannot be used, as the need here is to document the response of lake ecosystems to changes in nutrient concentration and climate change, not the other way round. The environmental data, therefore, need to be generated independently, either through the use of other proxies or other techniques, such as modelling. For TP there have been attempts to use the geochemical P record as a proxy (e.g.Anderson *et al.*, 1993), sediment pigment records (Guilizzoni *et al.*, 2011) and export coefficient modelling (e.g.Bennion *et al.*, 2005) based on known catchment land-use and population change. Each has limitations (cf.Bennion *et al.*, 2005).

In Mjøsa, Hobaek *et al*. (2012) uses the chlorophyll *a* record in the sediment as a proxy for productivitiy, whereas Bennion *et al*. (2012) and Dong *et al*. (2012a) avoid the problem by using direct measurements for nutrients from the respective long-term monitoring records. Their analysis is thereby restricted to the time-frame of the observational data. It also relies upon excellent chronological matching (see above), but the measurements themselves can be assumed to be accurate. For Esthwaite

Water this approach allowed winter SRP as well as TP to be used as an explanatory variable.

In contrast to the general lack of long time-series data on nutrient concentrations for lakes, climate data are potentially much more extensive and easier to obtain. Not only are temperature records for most regions of Europe available for well over 100 years they are also spatially quite coherent (Agusti-Panareda & Thompson, 2002).

The sites in this study are mainly lowland sites or sites with nearby meteorological stations, where data can be assumed to need little or no correction to make them directly applicable to the specific lakes. There is some uncertainty, however, with the relationship between air temperature and water temperature. Although in many cases and many seasons it can be assumed that there is a close relationship between surface water temperature and air temperature the relationship can be complex especially during periods of thermal stratification. For Mjøsa (Hobaek *et al*., 2012), water temperature data for the lake were available since 1972, but before then the temperature record had to be patched together from a range of documentary sources and local weather stations. Combining the data enabled Hobaek *et al*. to generate a temperature record for Mjøsa from 1785 to the present day.

Precipitation data are more difficult to hindcast as precipitation patterns are more variable spatially. However, it is a key variable influencing in particular the timing and amounts of nutrients delivered to lakes from lake catchments (George *et al.*, 2004) and can strongly influence changes in lake level (cf.Kampf *et al.*, 2012). Study lakes need to be located very close to meteorological stations or precipitation data can be modelled at the catchment scale to simulate runoff. For White Lough (Anderson *et al*., 2012), the site is sufficiently close to the Armagh Observatory to enable the long record of precipitation from the Observatory to be used as a direct record of precipitation for the lake.

Analysis of environment – response relationships

Once environmental data are assembled it is often difficult to match them accurately with sediment core records. Instrumental records are precisely and accurately fixed in time whereas sediment chronologies contain uncertainties associated with the use of radiometric dating techniques (cf.Lotter & Anderson, 2012) as outlined above. However, instrumental data are not without problems caused by missing data and do not always accurately register extreme events.

To match instrumental and sediment records it is usually necessary to derive estimated calendar dates for sediments using age-depth modelling (Blaauw & Heegard, 2012) and smooth the instrumental data, for example using LOESS, to a common temporal resolution. The objective is to produce comparable time-series that match each other in chronology and temporal resolution (cf.Battarbee *et al.*, 2002).

Smoothing observed time series to match the resolution of sediment records, however, is also not without its difficulties. When smoothing it is assumed that the observed data come from an underlying process with noise superimposed:

$$y_i = f(x_i) + \varepsilon_i \qquad \varepsilon_i \sim N(0, \sigma^2)$$

where y_i are the observed data, x_i the time points, *f*() a smooth function, and ε_i the residuals representing the unexplained variance. The smoothed time series is then given via:

$$\hat{y}_i = f(x_i)$$

where $\hat{y}_i$ are the fitted values of the smoother. These fitted values exclude the stochastic variation of, and hence have much lower variance than, the unsmoothed data. If some of the signal of interest lies in the stochastic component it will be missing from comparisons of palaeo- and smoothed instrumental data. In addition, the temporal dependence of the observations invalidates many of the usual procedures employed to determine the optimal degree of smoothness required to uncover the signal. This procedure can generate data-sets that can then be subjected to numerical analysis. The statistical techniques for exploring such relationships depend on data type and the length of the time series being analysed. The aim is to assess the relative and varying importance of different environmental variables (in this case climate and nutrients) on species composition or abundance (and in this case the composition of diatom assemblages) through time. There are two key problems, first, diatom datasets are large and complex with many species and often with missing values and second, sediment core analyses are rarely conducted at the high resolution needed to generate the large number of observations required to use techniques such as Additive Modelling (AM) (Simpson and Anderson, 2009). The first difficulty can be addressed by extracting the main trends in species composition using the first few ordination axes or, in the presence of a single gradient, via a principal curve (De'ath, 1999; Simpson & Birks, 2012).

The second problem is dependent on data availability and is often, but not always, related to the intervals at which sediment cores have been sampled. At Esthwaite Water because of the length of the instrumental record (since 1945), the relatively rapid rate of sediment accumulation (> 0.5 mm yr^{-1}) and the high resolution of core sub-sampling, it was possible to generate a diatom time-series of over 80 samples covering the period 1945 to present day, sufficient to enable analysis using AM (Dong *et al*., 2012a). In the Mjøsa study, on the other hand, because of the lower sediment accumulation rate and the shorter TP monitoring time-span (1972 to 2005), only 10 sediment diatom samples were available for analysis. In this and other cases where fewer data points are available, an alternative approach is to use constrained ordination (e.g. Redundancy Analysis; RDA) (cf.Lotter & Birks, 1997). Although this is a powerful method, it can only give a single value for the variance explained across the time series, whereas AM enables the variance explained by different predictor variables to vary through the time series (Simpson & Anderson, 2009; Dong *et al*., 2012a). An additional limitation is that RDA and its relative, canonical correspondence analysis (CCA), model the linear effects of the explanatory variables on the species abundances or compositions. Often the real response relationships are non-linear and are difficult to model using RDA or CCA but can be handled via regression splines in the AM approach.

A hybrid approach was employed by Bennion and co-workers (Bennion *et al*., 2012), who utilised unconstrained ordination to extract the main patterns of variation in the diatom species composition of a core from Loch Leven. The sample scores on the first two principal components were then used, via a multivariate smoother, to fit a response surface in the ordination space that explained variance in the instrumental data matched to the sediment core samples. This approach allowed for non-linear relationships between the main nutrient and climate drivers, although it is limited to situations where the main patterns of change in the species data are related to the measured instrumental variables; where this is not the case, careful consideration of later unconstrained ordination axes will be required.

One of the main constraints on the wider application of more statistically-based approaches to modelling palaeo-data in general or their relationships with instrumental records more specifically, is the temporal dependence of the data. Temporal autocorrelation violates a fundamental assumption of the basic regression model; that the data are independent random variables. A number of methods can be used to relax this assumption (e.g. the correlation structures for model residuals employed by Simpson & Anderson (2009) although the statistical and computational knowledge required to deploy these approaches lies much beyond that of the ordination-based techniques. Expert assistance from statisticians could be usefully employed in the future to help address more comprehensively the issues raised by the data-sets created for the papers in this Special Section.

Disentangling climate impacts

In recent years there have been many attempts to use lake sediment records to provide evidence for recent climate change. However, lakes in populated regions of the world are far from pristine and almost all suffer from the impact of human activity. Identifying the unique influence of climate on lakes using lake sediment records is hence difficult. The confounding impact of pollution cannot be discounted even in remote regions (e.g.Wolfe *et al.*, 2001). In the majority of the case studies in this Special Section the dominant pressure remains nutrient enrichment.

Nutrient concentration and recovery from eutrophication

Following reductions in nutrient loading over recent decades lakes in Europe are or should be recovering from eutrophication, which should be apparent both from monitoring data and from changes in the uppermost sediments (Jeppesen *et al.*, 2005). These reductions in nutrient loading, however, have been paralleled by some of the warmest years on record (IPCC, 2007), and in some cases reductions in point- source nutrient inputs are being offset by increased P-loading from diffuse sources as phosphorus saturation of soils increases (e.g.Foy *et al.*, 2003).

Fig. 3 shows trends in TP and chlorophyll *a* for four of the sites listed in Table 1. In the cases of Mjøsa and Piburger See there have been significant reductions in both TP and chlorophyll *a* following measures taken to reduce external loading from point sources. In the case of Piburger See, monitoring was introduced only after the control measures were taken (Fig. 3) but the success of the measures taken can be judged from the sediment core evidence for its previously enriched state (Fig. 2, Thies *et al*., 2012). In contrast the response of Loch Leven to the reduction in nutrient loading is limited (Carvalho *et al*., 2012) and in Esthwaite Water there has been little response so far (Maberly *et al.*, 2011).

For Mjøsa and Piburger See there have also been significant changes in the diatom assemblages of the sediments towards pre-eutrophication floras of the lakes. However, in Mjøsa not all the pre-eutrophication taxa have re-appeared and the lake remains on the borderline between oligotrophic and mesotrophic, either because nitrate concentrations remain high or because water temperature has increased (Hobaek *et al*., 2012). Also in Piburger See several diatom taxa (i.e. *Asterionella formosa* and some *Cyclotella* species) have appeared in the lake that have not occurred previously, again possibly due to climate change (Thies *et al*., 2012). For Loch Leven and Esthwaite Water diatom evidence for recovery is also less apparent in accord with the less pronounced reduction in TP concentrations at these sites. Both are relatively shallow sites and are more naturally eutrophic than Mjøsa and Piburger See, and may also be more susceptible to internal P recycling.

Climate change

The principal difficulty in identifying the role of climate change separately from that of nutrient loading on lake ecosystems is not simply the confounding influence of eutrophication as a stressor but that climate change itself promotes symptoms of eutrophication both directly through extending the growing season and indirectly through its effect on nutrient cycling and on food-web structure (cf.Moss *et al*., 2011). High nutrient concentrations and increased algal growth can be caused by either or both pressures. From a management perspective, climate warming thereby will tend to counter measures taken to restore lakes from the effects of eutrophication.

A further complication is the different ways in which climate change is expressed, for example through changes in temperature or precipitation, through changes in seasonality or through changes in the magnitude and frequency of extreme events. Indeed given also that the natural climate system possesses inherent variability on different time-scales, e.g. associated with modes of variability such as the NAO (North Atlantic Oscillation), it is perhaps not surprising that unambiguous impacts of anthropogenic climate change are not yet being detected either by observational records or by lake sediments. Nevertheless, it is not too soon to examine long-term records and lake sediment records for evidence of climate change, related to both natural climate variability and to anthropogenically forced climate change.

Modes of variability

Following the pioneering work of George & Harris (1985) and George & Taylor (1995) on the relationship between zooplankton populations in Windermere and decadal-scale cycling in water temperature, there has been a wealth of data accumulated from long-term records across Europe of the importance of the NAO in driving variability in lake systems, especially in winter water temperature, SRP and silicate concentration, the spring phytoplankton bloom and spring daphnid populations, all of which show strong regional coherence (Blenckner *et al.*, 2007).

Identifying the record of NAO variability and other modes of variability such as El Nino Southern Oscillation (ENSO) in lake sediments is therefore an important challenge for palaeolimnology. In the Americas, there has been some success not only in identifying ENSO-like variability, but also in demonstrating changes in its intensity over longer time periods (e.g.Conroy *et al.*, 2008). In Europe, attempts have not been so successful. Here both Hauptfleisch *et al*. (2012) and Anderson *et al*. (2012) have reported evidence for decadal scale cyclicity in the sediments of Mývatn and White Lough, respectively, and Kämpf *et al*. (2012) have shown evidence for repeated flooding in Maggiore, the most severe of which have been linked to the NAO (Guilizzoni, personal communication). In Mývatn the monitored cycles of chironomid and cladoceran populations seem unrelated to the NAO and the authors prefer to explain the cyclicity in terms of in-lake food web dynamics. Nevertheless the sediment record shows evidence for cycles prior to the start of lake monitoring in 1975 indicating the potential value of further work at this site in exploring the longer history of the variability and its possible link to climate variability, should the sediment chronology be sufficiently improved in future. In White Lough, the sediment is well dated and Anderson *et al*. (2012) argue that the cycles in biogenic silica recorded in the sediments can be linked to the NAO through the correlation between the summer precipitation record from the nearby Armagh observatory and the NAO (Butler *et al.*, 2007). However, from the perspective of the sediment record at White Lough, the in-lake response is probably mediated by catchment soil processes and the competition between diatoms and cyanobacteria for nutrients. This is a conclusion that could not have been reached without detailed monitoring records (in this case of nitrate and flow in the River Blackwater) and an understanding of phytoplankton dynamics in the lake (Foy & Fitzsimons, 1987).

In Esthwaite there is no NAO signal in the sediment recorded by the diatom assemblages, although such a signal has been identified from the long-term zooplankton data (George *et al*., 2004). Dating accuracy and sediment accumulation rates at this site are sufficiently high for the NAO record to be potentially recognised. The lack of a signal may be either due to smoothing by bioturbation or to weather patterns related to the NAO having a greater influence on water column productivity rather than on algal compositional change. As the zooplankton populations appear to be especially sensitive to NAO effects (George *et al*., 2004) a further measure for the future would be to examine changes in the abundance of cladoceran remains in the Esthwaite sediments.

Secular trends in temperature

In addition to cyclical or quasi-cyclical natural variability in the climate systems, meteorological records show that there has been a significant underlying increase in annual mean air temperature across Europe over the last few decades (IPCC, 2007), with especially large increases recorded in high latitudes and in alpine regions. For example, in Iceland temperatures have risen by 0.7-1.6^{o}C over the time period of 1871-2002 (Hanna *et al.*, 2004). Sea ice was frequently observed at the coast of Iceland during past centuries (Ogilvie and Jónsson, 2001), but has vanished during the last 30 years (Björnsson *et al.*, 2008). Moreover the Greater Alpine Region (the area between 4-19°E and 43-49°N) has warmed twice as much as the global or Northern Hemispheric mean since the late 19th century at all elevations (Auer *et al.*, 2006).

The increase in air temperature has caused increases in surface water temperatures both of rivers and lakes (Fang & Stefan, 1999; Hari *et al.*, 2006), hypolimnetic warming in large lakes (Dokulil *et al.*, 2006), and decreasing ice-cover in northern and high altitude lakes (Magnuson *et al.*, 2000). There is evidence to show that such physical changes in surface waters have been the cause of changes in the seasonality of phytoplankton (Catalan *et al.*, 2002), an extension to growing seasons and increases in lake productivity (Blenckner *et al*., 2007), changes in the geographical range of taxa e.g. Odonata (Hickling *et al.*, 2005) and threats to cold stenothermal taxa (e.g.Griffiths, 2007).

The sediment records of the lakes in this study provide only limited evidence for warming, as it is clear that nutrient enrichment remains a more important driver of trophic state. Indeed, this is in keeping with other palaeoecological studies that show that climate variables explain only a small percentage of the variance when used as a predictor of ecological turnover in sediment records of eutrophication (Hall *et al*., 1999).

For Mjøsa, Hobaek *et al*. (2012) used RDA to show that temperature exerted a significant influence on changes in diatom composition in the sediment record, but also pointed out that temperature is correlated with depth and time so that other age-related factors might be important. However, the data from Mjøsa show that although TP has now been reduced to reference levels (4 µg L^{-1}) primary production remains relatively high, as indicated by the trends in chlorophyll *a* in comparison with TP (Fig. 3), and the diatom flora has not yet returned to its composition during the reference period (pre-1950). The authors argue that the recovery is prevented either because N concentration remains high and/or because the lake is now warmer than in the past.

Multivariate analysis of the Loch Leven data (Bennion *et al*., 2012) also shows that some of the variability in the diatom assemblage data through the core can be explained by changes in winter rainfall and temperature, although the key driver on a decadal time-scale is still nutrient concentration. The authors argue that climate is important in explaining inter-annual variability associated with diatom seasonality, not with the secular trend.

Esthwaite data (Dong *et al*., 2012a) show some evidence for the recent increase in *Aulacoseira granulata* (Ehrenberg) Simonsen being related to increasing temperature, but, as for Loch Leven, the dominant control on diatom assemblages remains nutrient concentration. The impact of warming on phytoplankton phenology identified by George *et al*. (2004) at this site is not apparent from the sediment record where small shifts in seasonality are unlikely to be registered.

Perhaps the clearest evidence for a climate impact related to recent warming comes from Piburger See (Thies *et al*., 2012), where there is good evidence that the change in diatom composition recorded both by contemporary monitoring and by the uppermost sediments appears to be related to an increase in productivity linked to climate change rather than to nutrient loading, as nutrient loads have now been reduced to baseline levels and internal P recycling, unlike Loch Leven and Esthwaite Water, may be more limited. Both TP and chlorophyll *a* levels are now increasing despite the reduction in external loading (Fig. 3).

The complementarity of palaeo-data and observational data

As long-term lake time series continue to lengthen, the temporal overlap with lake sediment records increases providing increased opportunities for comparing the two. Comparisons allow the faithfulness of the sediment record and the robustness of palaeoecological techniques used to be evaluated providing added confidence in the use of sediments in extending the lake record back through time. In European lakes this is especially important as even the longest observational records, such as those from the English Lake District that began in the 1940s, are insufficiently long to capture conditions prior to the onset of nutrient enrichment from human activity (Bennion *et al*., 2011).

The ultimate value of combining information from long-term observations and sediment records lies in the potential of building data-sets sufficiently long and robust to enable the potential causes of ecological change to be examined. From the case studies presented in this Special Section, there is clear palaeo-evidence for the beginning of nutrient pollution varying from the mid 19th century at Loch Leven (Bennion *et al*., 2012) to the early and middle 20th century at other sites (Table 1) in agreement with the range of dates for the onset of nutrient enrichment reported for other European lakes (Battarbee *et al*., 2011). Recovery or, in some cases, lack of recovery from eutrophication is best recorded by monitoring data. Point-source nutrient loading has been significantly reduced at all the case study sites, but the response (e.g. as indicated by chlorophyll *a* values) has been varied (Fig. 3) with little evidence of recovery at some sites, e.g. Esthwaite (Dong *et al*., 2012a) and Loch Leven (Bennion *et al*., 2012), and only partial recovery at others, e.g. Piburger See (Thies *et al*., 2012). Statistical analysis suggests that the reason for limited recovery is mainly due to continuing high nutrient concentrations in the lakes, possibly related to an increase in diffuse nutrient loading or to internal P re-cycling. Higher water temperatures may also be playing an increasingly important role (cf. Moss *et al*., 2011).

Combining the palaeo- and observational evidence for ecosystem change enables the past, present and potential future trajectories of European lake systems experiencing pressures from nutrient pollution and climate change to be constructed, as indicated conceptually in Fig. 4. From a starting point (a) that represents relatively oligotrophic conditions, the palaeo-record provides evidence for the progressive increase in nutrient loading that has taken place over previous decades (and in some cases centuries) giving rise to increasing primary productivity. Concern for deteriorating water quality that became manifest in the 1960s and 70s and that culminated in the passing of the Urban Waste Water Directive in 1991 led to measures being taken to reduce nutrient loading, represented by point (b). The response of many lakes, including the ones considered here, has been followed by monitoring programmes purposely designed to track the expected recovery process. In some cases, e.g. Mjøsa, there has been an immediate decrease in production (as measured for example by chlorophyll *a* values) with the recovery trajectory (arrow d) heading directly towards the origin or reference (point a). In other cases, especially shallow lakes such as Loch Leven and Esthwaite Water, the response has been limited with little reduction in production despite significant decrease in external loading. At such sites, recovery is delayed by internal P recycling despite the reduction in loading, but may be expected eventually to head back towards the reference (arrow e).

Overall the evidence indicates that nutrient enrichment remains the dominant influence on the trophic status of the study lakes and arrows (d) and (e) represent the dominant trajectories expected. But, although it is difficult to distentangle fully its relative importance there is also evidence that climate change may be offsetting some of the recovery expected from a reduction in nutrient loading at some sites. If this is indeed the case then algal growth may not be declining simply in line with nutrient loading, but remaining relatively high, tending to follow a deflected trajectory away from the reference (arrows f and g) and towards a new endpoint (c). Piburger See appears to be following the trajectory represented by arrow (g).

Whether these trajectories will be followed as nutrient loading continues to decrease and climate change pressures continue to increase is unknown. But it is becoming clear that attempts to restore lakes suffering from the effects of eutrophication will increasingly need to take into consideration the probability that baselines are shifting and that future targets for restoration may need to be different from past references. Evaluating the extent of the difference will need data from monitoring programmes into the future in comparison with benchmark data for the past, provided by the palaeolimnological record.

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Table 1 Length of long-term record compared with core data for the seven study sites included in the Special Section. Start year is start date of nutrient monitoring; length (yr) is length of nutrient monitoring record to 2012; core date is the year the dated core was obtained; SAR (mm yr^{-1}) is mean (or range) of sediment accumulation rate of last 25 years; overlap is sediment depth (cm) of start date of nutrient monitoring; sample interval is thickness of sediment sample slices (cm); sample resolution is number of years represented by a sediment sample; point of change is approximate date of first diatom evidence for nutrient enrichment; P control date is date of first measures to reduce P loading. Note that data for Loch Leven in Fig. 2 are taken from core LEVE11 as well as LEVE12 and the "point of change" for White Lough is derived from diatom analysis of WHI-12 (Anderson *et al*., 2012).

Site	**Myvatn**	**Mjösa**	**Leven**	**White**	**Esthwaite**	**Piburger See**	**Maggiore**
Start year	1975	1972	1968	1978	1945	1974	1966
Length (yr)	37	40	44	6	67	38	46
Core code	KB-1	MjösaA	LEVE12	2007Kajak	ESTH7	PIB04	LM06
Core date	2006	2006	2005	2007	2006	2004	2006
SAR (mm/yr)	8-23	1.1-2.1	10-30	3-14	5.8	2	4-8
Core overlap (cm)	35-36	5	62	17	21	5.5	15-20
Sample interval (cm)	0.5	0.5	0.25	0.25-0.5	0.25	0.5	0.5-1.0
Sample resolution (yr)	< 1	2-10	< 1	<1	1-2	2-3	1-4
Point of change	na	1950	1850	1880	1880	1920	1960-5
P control date	na	1976	1985	na	1986	1970	1980

Figure captions

Fig. 1. A comparison of age-depth relationships for cores from the case study lakes described in this Special Section. For Mýyvatn, dates are based on a combination of tephra and the known dates of chironomid population maxima. For all other sites, data are from ^{210}Pb dating. For clarity, standard errors are not shown (see text).

Fig. 2. Comparisons between diatom-inferred total phosphorus (DI-TP) concentrations and measured mean annual total phosphorus (TP) concentrations for Loch Leven, Esthwaite Water, Mjösa and Piburger See through time. DI-TP data for Loch Leven are based on core LEVE11 for the pre-1945 period and LEVE12 thereafter.

Fig. 3. Comparisons between trends in mean measured total phosphorus (TP) and mean annual chlorophyll *a* for Loch Leven, Esthwaite Water, Mjösa and Piburger See showing the start date of measures taken to reduce nutrient loading at each site (→).

Fig. 4 Conceptual diagram using a combination of palaeolimnological data (dashed line), contemporary long-term monitoring data (solid lines) and future conjecture (dotted lines) to show idealised changes in the past, present and future relationship between nutrient loading and productivity for European lakes recovering from eutrophication. Point (a) indicates the reference state and the target endpoint following restoration; point (b) indicates the point of intervention to reduce nutrient loading; and (c) indicates a more probable potential endpoint in cases where recovery to the past reference (a) is prevented by the enriching effects of climate change. Arrows (d,f,e and g) are explained in the text.

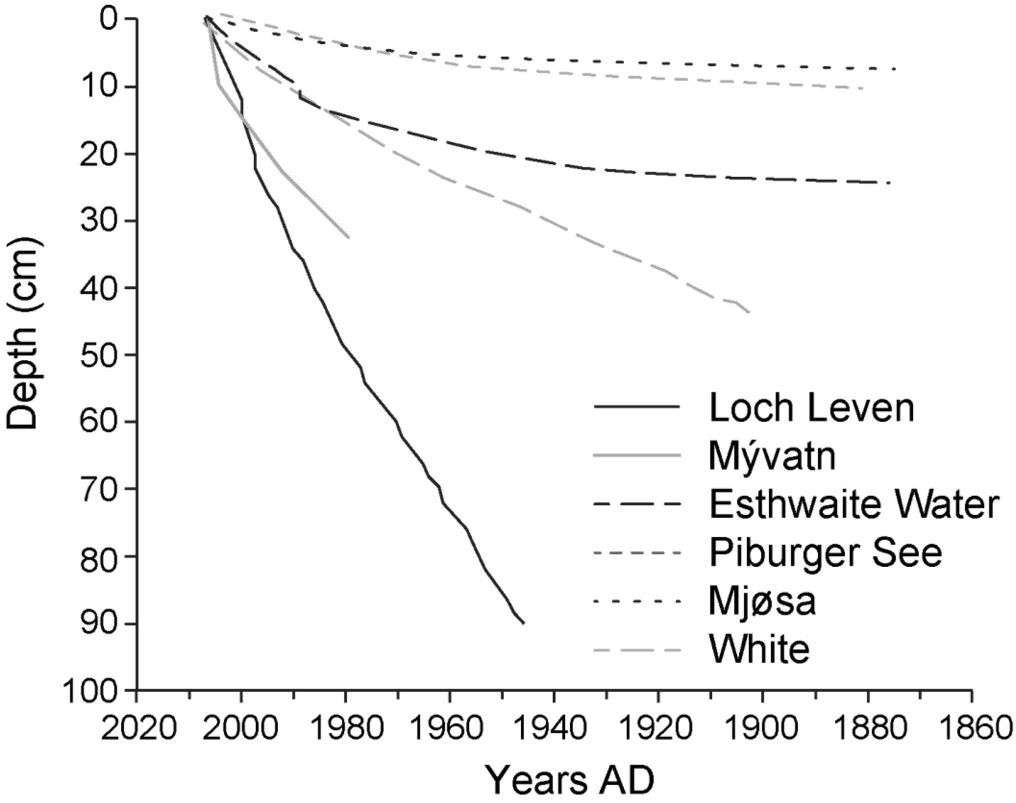
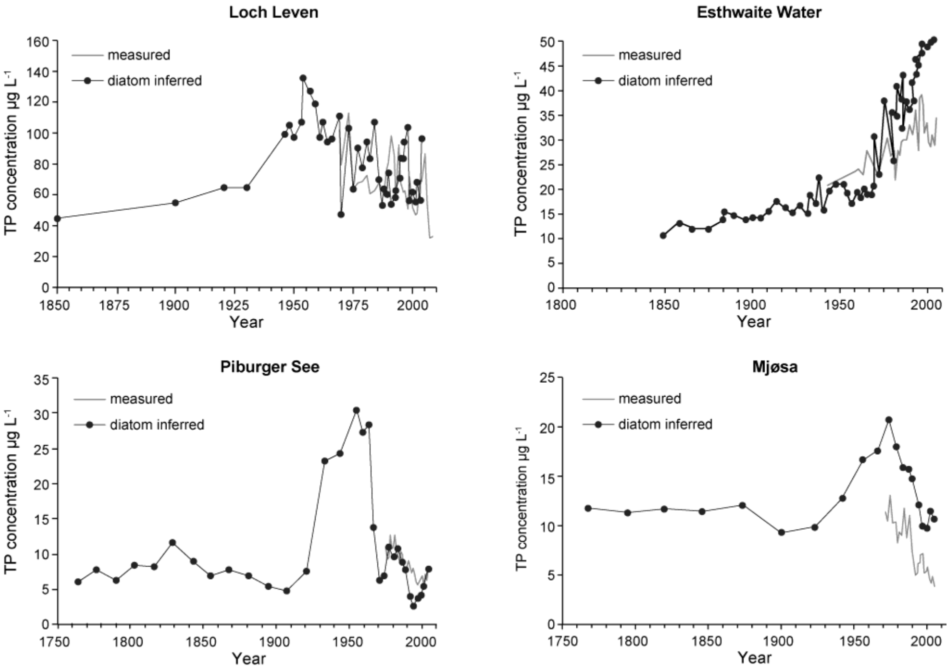

Depth (cm)
0
10
20
30
40
50
60
70
80
90
100
2020
2000
1980
1960
1940
1920
1900
1880
1860
Years AD
Loch Leven
Mývatn
Esthwaite Water
Piburger See
Mjøsa
White

Fig. 1

Fig 2



Loch Leven
Esthwaite Water
Piburger See
Mjøsa
measured
diatom inferred
TP concentration µg L⁻¹
Year
0
20
40
60
80
100
120
140
160
1850
1875
1900
1925
1950
1975
2000
0
5
10
15
20
25
30
35
40
45
50
1800
1850
1900
1950
2000
0
5
10
15
20
25
30
35
1750
1800
1850
1900
1950
2000
0
5
10
15
20
25
1750
1800
1850
1900
1950
2000

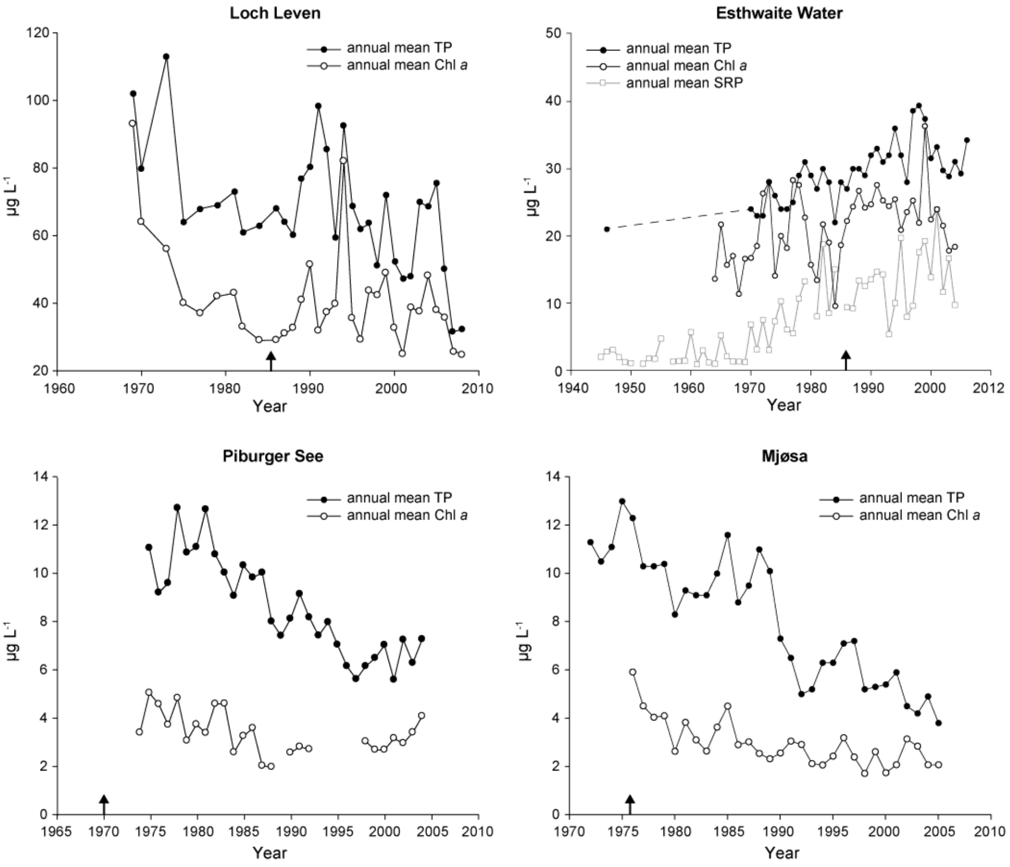

Loch Leven
annual mean TP
annual mean Chl *a*
µg L⁻¹
Year
120
100
80
60
40
20
1960
1970
1980
1990
2000
2010
Esthwaite Water
annual mean TP
annual mean Chl *a*
annual mean SRP
µg L⁻¹
Year
50
40
30
20
10
0
1940
1950
1960
1970
1980
1990
2000
2012
Piburger See
annual mean TP
annual mean Chl *a*
µg L⁻¹
Year
14
12
10
8
6
4
2
0
1965
1970
1975
1980
1985
1990
1995
2000
2005
2010
Mjøsa
annual mean TP
annual mean Chl *a*
µg L⁻¹
Year
14
12
10
8
6
4
2
0
1970
1975
1980
1985
1990
1995
2000
2005
2010

Fig. 3

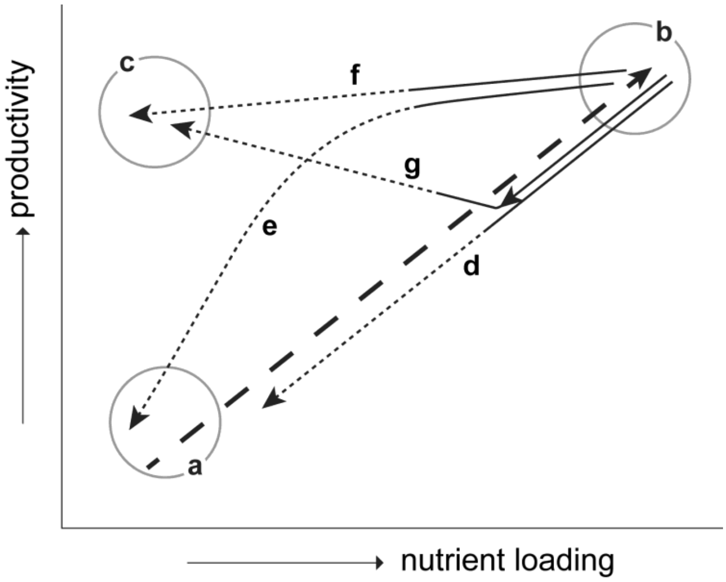


Fig. 4