

A comparison of sedimentary and diatom-inferred phosphorus profiles: implications for defining pre-disturbance nutrient conditions

N. J. Anderson¹, B. Rippey² & C. E. Gibson³

¹*Geobotany Division, Geological Survey of Denmark, Thoravej 8, DK-2400 Copenhagen NV, Denmark;*

²*Freshwater Laboratory, University of Ulster, Traad Point, Ballyronan BT45 6LR, N. Ireland;* ³*Dept. of Agriculture (NI), Aquatic Sciences Research Division, Newforge Lane, Belfast BT9 5PX, N. Ireland*

Key words: weighted averaging, calibration, phosphorus, diatoms, palaeolimnology, Northern Ireland

Abstract

Interpreting sedimentary phosphorus profiles in terms of changes in the historical P load is difficult due to variable retention and post-depositional diagenesis. An alternative approach is to use diatom assemblages in surface sediments and derive a transfer function for epilimnetic SRP and total P concentrations using weighted average regression and calibration. The obtained relationship can then be applied to down-core changes in sedimentary diatom assemblages and diatom-inferred P (DI-P) used to assess historical changes in epilimnetic P-concentrations. A diatom-phosphorus calibration data set for 43 eutrophic lakes in Northern Ireland has been constructed and applied to two small eutrophic lakes (Lough Mann, White Lough). DI-total P (i.e. predicted) is highly correlated with observed TP ($r^2 = 0.75$) for the surface-sediment training data-set. The resultant changes in DI-P derived from application of the transfer function to down-core changes in diatom assemblages are compared to sedimentary P concentrations. The latter are highly variable, presumably due to redox-derived effects, while DI-P profiles are more readily interpretable, and agree with other stratigraphic records of lake eutrophication. The method offers a good possibility of defining pre-disturbance (i.e. natural) phosphorus concentrations in lakes with associated implications for lake-restoration programmes.

Introduction

Pre-eutrophication phosphorus concentrations in lakes are unknown: many lakes in north-west Europe have been subject to perturbation over a variety of timescales, nearly all of which are longer than current monitoring programmes. However, for both ecological and lake-management strategies it is necessary to have information concerning the natural situation before disturbance, prior to the start of lake restoration (Schindler, 1987).

Realistic estimates of background conditions can often only be made by using the sediment

record. As lake eutrophication primarily results from increased phosphorus loading, it is often assumed that sedimentary P provides a suitable record of enrichment (e.g. Birch *et al.* 1980). Unfortunately, factors controlling phosphorus retention in lake sediments are complex and sedimentary P profiles may have only a vague relationship to the timing of increased P inputs, and even less to the whole-lake P load (Engstrom & Wright, 1985; Carignan & Flett, 1981; Marsden, 1989; Mayer *et al.*, 1982; Bengtsson & Persson, 1978).

Algal communities change as nutrient concentrations increase (e.g. Hörnström 1981; Rosén,

1981). Until recently the diatom record in lake sediments has been interpreted subjectively using such criteria; e.g. phosphorus-rich (eutrophic) lakes are dominated by *Stephanodiscus* and P-poor (oligotrophic) lakes by *Cyclotella*. However, with the development of weighted averaging regression and calibration (ter Braak & van Dam, 1989; ter Braak & Prentice, 1988; Birks *et al.*, 1990; Line & Birks, 1990) it is possible, theoretically, to determine quantitatively the relationship of a species to a given nutrient (e.g. total phosphorus [TP]), and then use these relationships to infer past changes in the chosen parameter from down-core changes in diatom assemblages in lake sediments. This approach has been used with considerable success to infer past changes in lakewater pH and salinity (Birks *et al.*, 1990; Fritz *et al.*, 1991).

We present here results from a diatom calibration or training data-set for eutrophic lakes in Northern Ireland and compare diatom-inferred phosphorus (DI-P) to the geochemical phosphorus record, and assess its implications for determining the pre-enrichment nutrient status of lakes.

Study sites

Training set

The lakes used for the training data-set were selected to provide a phosphorus gradient (Table 1), and all are meso- to eutrophic by OECD guidelines (Forsberg & Ryding, 1980). The lakes are located in Counties Down (16 lakes), Armagh

(25 lakes), and Tyrone (2 lakes), along a broad S.W. – N.E. transect of increasing conductivity and P concentrations. The limnological characteristics of these lakes have been summarized by Gibson (1986, 1988, 1989). The lakes in Co. Down are primarily situated on Silurian greywacke sandstones, siltstones and shales, and a large granitic intrusion, the Newry granodiorite. The geological setting of the Co. Armagh lakes is more complex, but this area too consists of substantial areas of Silurian greywackes (see Gibson, 1989; Wilson 1972 for further details). The two lakes from Co. Tyrone are located on Carboniferous shales and sandstones.

The lakes are small (< 50 ha) with depths ranging from ~ 1 to 20 m (Table 1). Many of them are located within drumlin belts and are, therefore, quite sheltered. Land use at most sites, especially in Co. Armagh, is unimproved grassland, but there is a N.W. trend towards increasing soil fertility and agricultural productivity in Co. Down.

Stratigraphic study sites

To assess the applicability of the diatom-P calibration to down-core changes in diatom assemblages and its comparison with the geochemical record, two eutrophic lakes were chosen (Table 2). Lough Mann has a small, mainly pastoral, catchment with no apparent disturbance history. White Lough was chosen as it was the site of a chemical manipulation exercise (addition of ferric aluminium sulphate) in 1978–1982 (Foy, 1985). Although the effect on epilimnetic P-concentrations was limited, it was hoped that the addition of Fe (~ 3 times the normal load) would be clearly identified in the stratigraphic record, but unfortunately this was not possible.

Methods

For the training set samples surface sediment samples (~ 1 cm depth) were taken using a modified Brinkhurst-Kajak corer from the deepest part of the lakes during April 1990 and May 1991.

Table 1. Summary characteristics for 43 training lakes.

	Range
Lake area:	1–30 ha
Maximum depth	0.6–20 m
Total P	25–800 $\mu\text{g l}^{-1}$ *
SRP*	2–470 $\mu\text{g l}^{-1}$ *

* Mean of all available chemistry.

* Soluble reactive phosphorus.

Table 2. Summary of water chemistry and sediment core chemistry for White Lough and Lough Mann.

Lake	White		Mann	
Area (ha)	6		12	
Max. depth (m)	11.2		8	
Total P* ($\mu\text{g l}^{-1}$)	19–43		51–73	
SRP* ($\mu\text{g l}^{-1}$)	9–16		9–12	
Chl. <i>a</i> * ($\mu\text{g l}^{-1}$)	17–26		18–41	

Sediments	Mean*	Mean		Mean	Mean	
		(mg g ⁻¹)			(mg g ⁻¹)	
Mg	4.15	5.62	2.94	4.51	4.87	4.02
Na	1.96	2.56	1.86	2.02	2.17	1.88
K	11.82	15.1	10.5	2.45	3.10	2.24
Ca	9.53	21.2	5.44	8.18	10.5	6.55
Fe	40.9	60.6	27.9	32.6	61.1	10.8
Mn	1.85	3.51	0.76	1.03	1.65	0.57
P	2.62	4.51	1.16	2.33	4.38	1.36

* Range 1991.

* Mean whole-core.

For the diatom and geochemical stratigraphic analyses short cores (~30 cm sediment recovery) were taken using similar methods (Table 2), sealed in the field, and transported to the laboratory for extrusion. The core from White Lough was extruded at 0.5 cm intervals over the top 11 cm, and at 1 cm intervals below this depth. The Lough Mann core was extruded at 1 cm intervals down to 20 cm, and at 2 cm intervals below this depth.

Water chemistry

The lakes were sampled for water chemistry 2 to 4 times between April 1990 and May 1991, although some lakes were only sampled once in May 1991. Sampling has, however, continued in an effort to characterize better the chemistry of the lakes. Total phosphorus (TP) was determined by persulphate digestion of an unfiltered sample (Eisenreich *et al.*, 1975); and soluble reactive phosphorus (SRP) as phosphorus detectable in an undigested, filtered sample (Millipore

0.45 μm), with final determination by the method of Murphy & Riley (1962).

For the statistical analyses the log-transformation of the arithmetic mean of all samples was used (34 lakes) or the log of the single available sample (8 lakes). A comparative analysis was made using the December 1990 sample ($n = 31$), to assess the possible predictive capacity of winter chemistry prior to depletion by algal growth.

Diatom and geochemical methods

Dry weight and loss-on-ignition measurements were made on the cores using standard methodology. Diatom analyses followed standard methodology (Battarbee, 1986) with ~500 valves counted for each surface sediment sample. For the sediment cores a reduced count size was used (~300 per level for Lough Mann, and ~400 for White Lough). Only species with an occurrence of 1% in any one of either the training or core samples were retained for use in the statistical analyses (112 species from 189 identified). Bulk geochemical analyses for the major ions follow Anderson & Rippey (1988), with total phosphorus determined by colorimetry (phosphomolybdate method: Golterman *et al.*, 1978) on the sediment digest after a twenty- or fifty-fold dilution.

Statistical methods

Preliminary data analysis was undertaken using the canonical correspondence analysis (CCA) routine in the program CANOCO (ter Braak, 1987). Weighted averaging (WA) regression (to estimate taxa P optima) and WA-calibration (to estimate diatom-inferred lake P concentrations) were then done using the program WACALIB version 2.1 (Line & Birks, 1990). Inverse de-shrinking was used during WA calibration as this reduces the predictive error, as measured as Root Mean Standard Error (RMSE; Birks *et al.*, 1990; Fritz *et al.*, 1991). WA calibration with tolerance down weighting [WA(tol)] was also undertaken (Birks *et al.*, 1990). The underlying statistical ra-

tionale and theory, together with further details of the methodology are given elsewhere (ter Braak & van Dam, 1989; ter Braak & Prentice, 1988; Birks *et al.*, 1990).

Apart from routine error screening of both the diatom and chemistry data no species or lakes were removed from the data set prior to calibration. The present data set is not sufficiently large to support screening or splitting, and the ecological basis is not entirely obvious. However, retaining lakes with unusual diatom assemblages at given P concentrations reduces, therefore, the predictive ability of the model (see discussion).

Results

Diatom inferred P using weighted averaging calibration

A CCA (ter Braak, 1986) of the full chemistry data set and diatom data indicated that TP is the first significant chemical variable ($P < 0.05$) in a forward selection of all variables and is highly correlated with CCA axis 1 ($r = 0.86$). SRP is similarly significant and correlated with axis 1 ($r = 0.80$).

That TP is the dominant variable with a larger constrained eigenvalue than any other variable in a series of CCAs with single environmental variables is a requirement 'for the calibration to be useful' (ter Braak, 1987). Even though TP only explains 8% of the variance in the weighted averages of the diatom abundance data, this percentage is similar to that for pH in the Surface Water Acidification Programme (Stevenson *et al.*, 1991) data set (8.1% $\lambda_1 = 0.495$, $\lambda_2 = 0.394$). For the data set represented here, TP is the dominant variable even though $\lambda_1 < \lambda_2$.

A complete analysis of diatom distribution in relation to the full chemistry data set will be published elsewhere (Anderson, Gibson & Rippey, *ms in prep.*). Both WA and WA(tol) diatom-inferred TP have a good relationship with the observed values ($r^2 = 0.75$ and 0.77 respectively; Table 3; Fig. 1a), with no substantial outliers for TP. RMSE and r^2 for TP are marginally better for

Table 3. Summary statistics for Weighted Averaging Phosphorus calibration.

	WA			WA (tol)		
	<i>n</i>	r^2	RMSE	<i>n</i>	r^2	RMSE
Mean TP	43	0.75	0.172	43	0.77	0.164
Mean TP*	41	0.67	0.157	41	0.71	0.156
December TP	31	0.69	0.166	31	0.75	0.150
MeanSRP	43	0.64	0.304	43	0.62	0.309
Mean SRP*	42	0.71	0.267	42	0.73	0.257
December SRP	31	0.70	0.287	31	0.73	0.269

* Excluding Islandderry & Monlough (cf. Fig. 1a).

* Excluding Carrigullian (cf. Fig. 1b).

WA(tol) than for WA (Table 3). Removing the two shallow hypertrophic lakes (Monlough and Islandderry) lowers the r^2 value slightly. The RMSE values for December TP are lower than of those for the mean of all measurements (Table 3), suggesting that winter chemistry may prove to provide a more reliable basis for prediction in a larger data set. Here, although the number of lakes is reduced, r^2 is not substantially lower.

Predicted mean SRP concentrations are also well correlated with observed values (Table 3; Fig. 1b), with only one outlier, but the fit is not as good as for TP (see discussion). Removing the outlier (Carrigullian, a very shallow [z mean: ~ 1.5 m] eutrophic lake; measured TP range $100\text{--}180 \mu\text{g l}^{-1}$; September 1991 $170 \mu\text{g l}^{-1}$ Chl. *a*) improves the relationship (Table 3).

These data suggest that there is a reasonable basis for the use of the relationship for inference of TP from changes in down-core diatom assemblages. Although the summary statistics suggest that WA(tol) is a better predictive tool, WA was used for the down-core reconstructions because the ecological implications of the WA(tol) approach have yet to be assessed.

Core applications

Lough Mann. Cation profiles are stable indicating minimal catchment disturbance, although zinc and copper increase above 3 cm (data not shown),

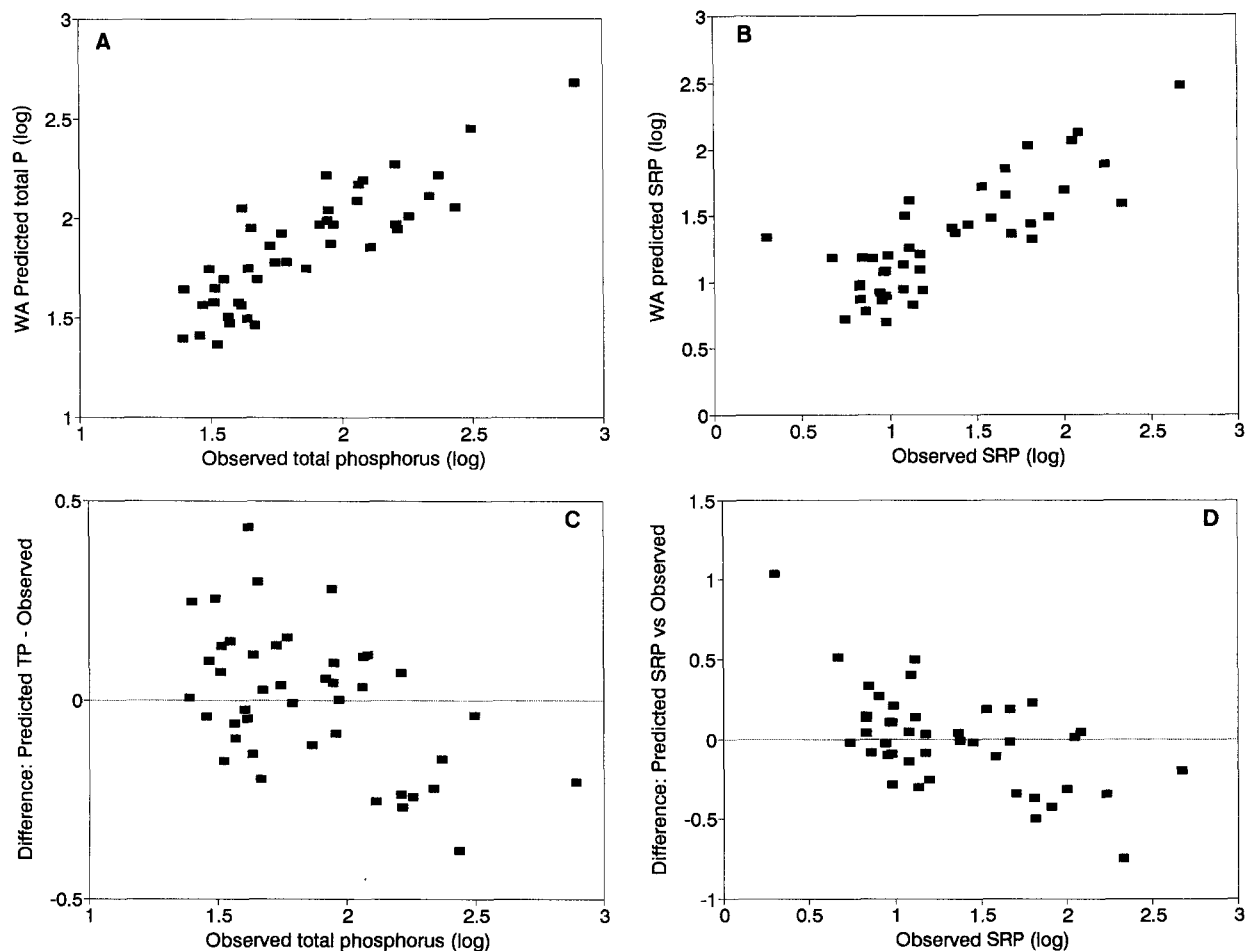


Fig. 1. Parts a & b: Scatter plots for diatom-inferred phosphorus vs observed phosphorus concentrations ($\log_{10} \mu\text{g TP l}^{-1}$); a: TP; b: SRP; both plots use all available chemistry – see text. Parts c & d: Differences between predicted and observed values for diatom-inferred total phosphorus (c) and soluble reactive phosphorus (d), both using Weighted Averaging model (WA).

perhaps due to surface enrichment (Fig. 2; Table 2). The sedimentary P concentration increases a little up-core, but this steady increase is interrupted briefly by a high value (12 mg g^{-1}) at 8.5 cm (Fig. 2). This feature coincides with temporary highs in Fe and particularly Mn concentration (Fig. 2).

Diatom assemblages show few major changes throughout the core. The assemblage is dominated by planktonic species and although there are no species replacements, percentages fluctuate. DI-P values are $\sim 100 \mu\text{g l}^{-1}$ at the base of the core and decline steadily up-core to around $67 \mu\text{g l}^{-1}$ at the surface. The measured mean TP

concentration ($n=2$) for Lough Mann in 1991 was $62 \mu\text{g l}^{-1}$.

White Lough. Again cation and loss-on-ignition profiles indicate stable catchment conditions (i.e. limited shifts in erosional intensity; Fig. 3, Table 2). Mn increases steadily up-core from $< 1 \text{ mg g}^{-1}$ to $> 3 \text{ mg g}^{-1}$ at $\sim 2 \text{ cm}$, apart from a decrease between 10 and 12 cm depth (Fig. 3). Fe concentrations are $\sim 40 \text{ mg g}^{-1}$ to 10 cm, after which there is a variable decline. The Fe:Mn ratio declines from a basal maximum to a pronounced peak at 10–12 cm (Fig. 3), but is then stable to the surface. Minimum sedimentary P concentrations

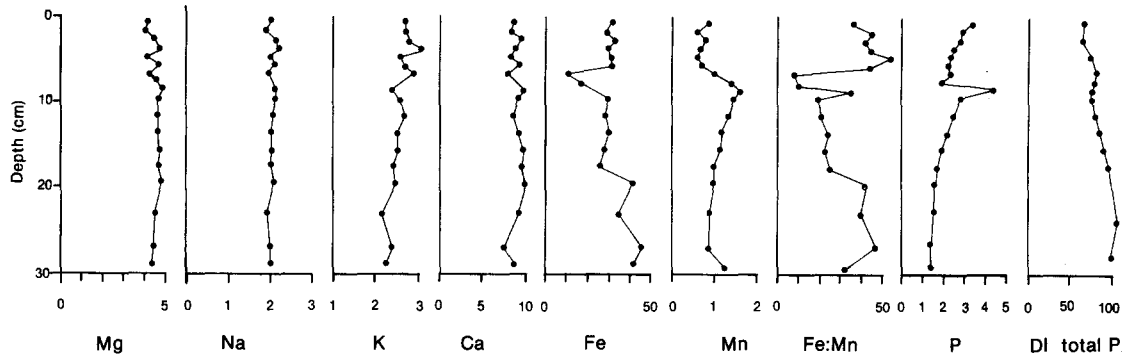


Fig. 2. Lough Mann core stratigraphy showing the differences between sedimentary P and diatom-inferred total phosphorus (DI-P). DI-P concentrations (i.e. for the lake epilimnion) are in $\mu\text{g l}^{-1}$; geochemical values are $\text{mg g}[\text{DW}]^{-1}$; Fe:Mn is a ratio.

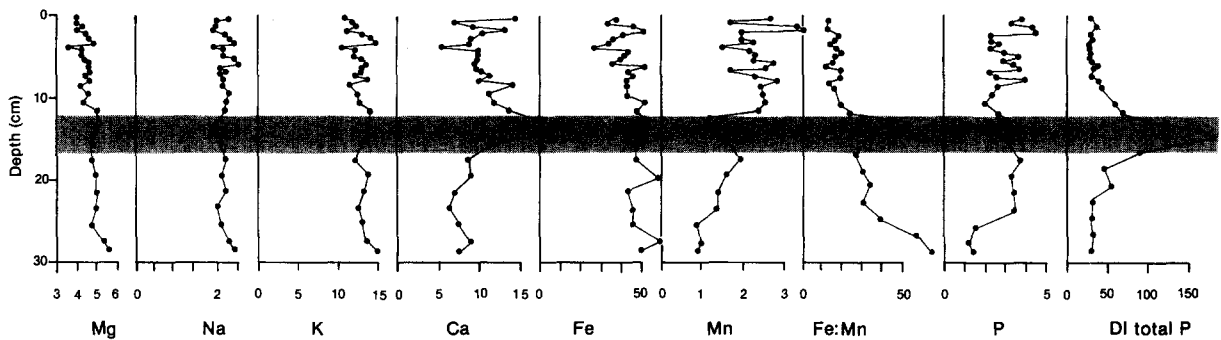


Fig. 3. As for Fig. 2, but for White Lough. The stipple indicates the period of good agreement between the diatom predicted increase in epilimnetic phosphorus concentrations and increases in sedimentary Ca and changes in Mn, Fe:Mn ratios associated with the eutrophication of the lake caused by the increased phosphorus availability.

($\sim 1 \text{ mg g}^{-1}$) occur at the base of the core, after which there is a pronounced increase to $\sim 3 \text{ mg g}^{-1}$. Above 12 cm values oscillate with no major trend. Sedimentary P concentrations are, however, correlated with Mn ($r = 0.60$) indicating the importance of redox cycling on the sedimentary P record at this site.

Despite changes in the diatom assemblages (Anderson, unpubl. data) DI-P is constant throughout the length of the core at $\sim 35 \mu\text{g l}^{-1}$, apart from a major increase to $160 \mu\text{g l}^{-1}$ between 12 and 15 cm (Fig. 3). The surface DI-P value ($35 \mu\text{g l}^{-1}$) agrees well with the measured mean value ($31 \mu\text{g l}^{-1}$). The peak in DI-P at 12–15 cm, apart from coinciding with reduced Mn and the peak in Fe:Mn (Fig. 3; both parameters are indicators of increased anoxia due to, presumably, increased P-loading and eutrophica-

tion), is also associated with a major peak in Ca, due to increased photosynthetic activity and precipitation of CaCO_3 .

Discussion

Increased concern about the quality of inland waters has resulted in the widespread application of a variety of measures (tertiary wastewater treatment, diversion schemes, chemo- and biomanipulation) to reduce nutrient (especially phosphorus) loadings to lakes (Sas, 1989). The possible results of these reductions in nutrient load are normally assessed by either OECD-type regression relationships or site-specific models (Dillon & Rigler, 1974; Forsberg & Ryding, 1980; Kamp-Nielsen, 1986). Few lakes, however, have been

extensively monitored, and prior to lake restoration attempts, monitoring may last only a few years with associated difficulties in determining intra-annual variability and mean nutrient and chlorophyll levels. The specific aim of lake restoration methods is normally not site-specific, but determined by either government legislation or 'subjective' criteria about clean lakes. The approach taken here is an attempt to develop a reliable method of determining pre-enrichment P levels for individual lakes.

Diatom-inferred P: Predictive ability

Diatom inferred-TP is well correlated with observed TP concentrations (Table 3), although there is some scatter associated with the relationship ($\sim 30\text{--}40 \mu\text{g TP l}^{-1}$). This range around the predicted values should, however, be compared both to the range in the data set ($25\text{--}800 \mu\text{g TP l}^{-1}$) and the intra-annual range of P concentrations that occurs in many eutrophic lakes (e.g. White Lough; Foy, 1985).

Errors associated with DI-P and its predictive ability, i.e. reasons for the scatter in the relationship between diatom-inferred and observed phosphorus, relate to a number of factors, in particular, diatom ecology, heterogeneity of diatom assemblages in lake sediments and chemical variability. The errors are, however, primarily due to the fact that the factors controlling algal abundance in eutrophic lakes are multivariate (Reynolds, 1984; Heyman & Lundgren, 1988).

There is a systematic bias – to underestimate at high P concentrations and vice versa – for both predictions of TP and SRP (Fig. 1c & d), and for both WA and WA(tol) models (data not shown). For the moment it is unclear what causes these trends, but at high P concentrations they are probably related to the inability of diatom communities to respond to increasing P concentrations by structural change. Diatom assemblages are quite uniform above $\sim 300 \mu\text{g l}^{-1}$ TP due to the limited number of diatom species physiologically adapted to high P and low Si:P ratios (e.g. *Stephanodiscus*). In the mid-TP range, tolerance weighting probably provides a better fit be-

cause it downweights those species which have a wider tolerance and are found across a range of nutrient concentrations (e.g. *Asterionella formosa* and *Aulacoseira ambigua*).

The range of r^2 values and RSME (nb. RSME values are only comparable for each variable; not between TP and SRP) for the different models (Table 3) suggests that the method is quite robust, in terms of its predictive ability. Interestingly, the December SRP model (WA[tol]), despite a reduced data set is as good as the full data set. On ecological grounds one would expect winter SRP concentrations to be well correlated with diatom assemblages. The overall agreement of the different models (Table 3) suggests that as the data set is expanded to include low P lakes and the chemistry set refined, statistically and ecologically sound predictions of TP will be obtained.

A more stringent test of the predictive ability of DI-P is to test the training-set model against a set of lakes (test set) not included in the training set (cf. Birks *et al.*, 1990). This approach permits a statistically rigorous estimate of the RMSE to be made. The model can also be tested by application to a lake with a known history of epilimnetic P concentrations, reconstructions in a dated sediment core can be compared to monitored data.

A measure of the usefulness of diatom P-calibration models can be gained by comparison with OECD-type regression relationships and the error associated with them (e.g. Dillon & Rigler, 1974). Criticism of the chlorophyll-phosphorus models has been extensive recently because of their poor predictive ability (e.g. Reynolds, 1987). Compared to the predictive ability of these regression relationships, the errors associated with DI-P are as good or better, and indicate that there is an acceptable future for the method in lake management.

Diatom-inferred P: interpretation and error sources

Given that diatoms are deposited in the sediments shortly after death or the collapse of a crop (Jewson *et al.*, 1981; Reynolds *et al.*, 1982) DI-P is a measure of the mean epilimnetic P concentration

at the time of the growth of the diatom assemblage. The DI-P from sediment cores thus provides a record of changing epilimnetic P concentrations.

The problem of mixing of diatom assemblages in lake sediments by bioturbation and resuspension, although of possible importance, is reduced in eutrophic systems because of the higher sediment accumulation rates. Equally, however, some mixing of the surficial sediment assemblage smooths out interannual variability in the diatom assemblages, not all of which may be related to nutrient concentrations. Importantly, the smoothed record is acceptable for management purposes where only major trends are required.

Diatom errors. The assemblages in this training set are dominated by planktonic species, but there are clear relationships between P concentrations and some benthic/epiphytic species (e.g. *Achnanthes minutissima*; Anderson unpubl.). It is possible, however, that some benthic diatoms do not relate directly to epilimnetic chemistry, due to the interactions between motile species and interstitial water nutrient sources and epiphytes with macrophytes (e.g. Jørgensen, 1957). These factors will result in a poorer predictive ability.

Factors controlling diatom assemblages in lake sediments are complex (Reynolds, 1984; Heyman & Lundgren, 1988), and apart from variability associated with depositional processes, lake morphometry influences the diatom assemblage and may result in underrepresentation of certain communities. However, where planktonic diatoms are dominant, it is reasonable to assume that a single sample will be representative (Anderson, 1990). Relative frequency data have the advantage over diatom accumulation rate data in that they are less variable (Anderson, 1992; Whitmore, 1991).

Where diatoms are totally dissolved (e.g. some carbonate systems) the approach cannot, of course, be applied. More importantly, however, are the possible effects of differential dissolution (i.e. loss of lightly silicified species; e.g. *Rhizosolenia*), biasing the assemblage and creating poorer fits between predicted and observed phosphorus

concentrations. Fortunately, severe differential dissolution is rare in freshwater lakes and therefore diatom assemblages in sediments should reflect water chemistry changes (Battarbee, 1986; Whitmore, 1991). This assumption is particularly true for surface sediments, but down-core diagenetic losses in some systems should not be discounted.

Lake water chemistry derived errors. A major problem with diatom training data-sets is the variability in lake-water chemistry. This stems from two sources: first, the limited number of samples that are commonly available when more than 40 lakes are being sampled. Resources do not generally permit a rigorous assessment of inter- and intra-annual variability at all sites. The second problem is the natural inter- and intra-annual variability of TP concentrations in lakes: what mean should be used: winter, spring or annual mean? P concentrations can vary considerably throughout the year, due to internal and external loading variations and biological depletion.

It is often assumed that winter chemistry provides the best predictive basis for mean summer chlorophyll. However, for the SRP model the poor predictive ability may be due to the fact that in many of the lakes in the data set, overwintering populations of *Oscillatoria* (Chl. *a* ~ 20–40 μl^{-1}) deplete SRP prior to the spring diatom crop (cf. Foy & Fitzsimons, 1987). Similarly, some of the diatoms in the data set commonly occur during summer (*Fragilaria crotonensis*, *Aulacoseira granulata*). It is, then, difficult to expect a better fit than that obtained whether the annual mean or winter mean chemistry is used.

For these reasons it may be difficult to reduce the errors associated with DI-P. A reasonable aim would be to get the total predictive error below the expected seasonal variation in measured epilimnetic TP.

Interpretation of sedimentary P

The problems of interpreting sedimentary P profiles in terms of historical P loads have been dis-

cussed extensively elsewhere (Carignan & Flett, 1981; Engstrom & Wright, 1985). Although unambiguous sedimentary P profiles do occur (Shapiro *et al.*, 1971), P mobility associated with redox changes (due to increased organic loadings) means that sedimentary P profiles need to be interpreted with caution (e.g. Bengtsson & Persson 1978). Sedimentary fluxes, as well as concentrations, need to be used to clarify the changes in lake P concentration and loading, and the lack of fluxes for the two sediment cores presented here limits the possible interpretation that can be made.

At both Lough Mann and White Lough it appears that redox changes have influenced the sedimentary P record. Neither sites have P profiles that are readily interpretable in relation to other stratigraphic records related to eutrophication. At White Lough the sedimentary P profile varies considerably, perhaps due to redox control (cf. P relationship with Mn). The DI-P profile, however, agrees well with other indicators of eutrophication (increased Fe:Mn ratio, Ca peak) and suggests a major eutrophication event in the recent past due to increases in epilimnetic TP concentrations to $\sim 160 \mu\text{g l}^{-1}$ (Fig. 3).

Sedimentary P profiles record gross changes in the lake water phosphorus concentrations, particularly when the change is from oligotrophic to eutrophic conditions, e.g. in Lake Washington (Shapiro *et al.*, 1971). However, extracting finer detail is much more difficult and it appears that this is particularly difficult when sediment P concentrations are high. Factors such as redox potential, year-to-year variations in lake conditions, e.g. hydraulic retention time and bioturbation, may have a more pronounced effect on sedimentary P concentrations when these factors are high, compared to when they are low.

Conclusions and management implications

The possibility of inferring epilimnetic phosphorus concentrations from diatom assemblages is very good, and its implications are clear: to provide important background information (i.e. pre-

disturbance) on P concentrations for individual lakes prior to the initiation of restoration schemes.

An important benefit of this approach is that it integrates complex diatom stratigraphic data into a form which is more readily interpretable by water managers (i.e. TP concentrations). The approach is quantitative and gives reliable error estimates, together with enhanced precision in interpreting the sediment record. As part of a multidisciplinary approach (dating, diatoms, pigments and geochemistry) DI-P would increase the possibility of interpreting sediment records unambiguously.

Acknowledgements

NJA acknowledges support from the Swedish Natural Science Research Council (NFR) and the J.C. Kempe Memorial Fund; and Prof. H. John B. Birks for discussions and comments on the manuscript. Prof. Brian Wood willingly made available the facilities at Traad Point. Field assistance by Paul Raven, Steven Smith and Mat Quinn, and the technical support of the FBIU Chemistry Laboratory staff are gratefully appreciated.

References

- Anderson, N. J., 1990. Variability of diatom concentrations and accumulation rates in sediments of a small lake basin. *Limnol. Oceanogr.* 35: 497–508.
- Anderson, N. J., 1992. Inferring diatom paleoproduction and lake trophic status from fossil diatom assemblages. *Proc. Calif. Acad. Sci.*, in press.
- Anderson, N. J. & B. Rippey, 1988. Diagenesis of magnetic minerals in the recent sediments of a eutrophic lake. *Limnol. Oceanogr.* 33: 1476–1492.
- Battarbee, R. W., 1986. Diatom analysis. In B. E. Berglund (ed.), *Handbook of Holocene palaeoecology and palaeohydrology*. Wiley, Chichester: 527–570.
- Bengtsson, L. & T. Persson, 1978. Sediment changes in lake used for sewage reception. *Pol. Arch. Hydrobiol.* 25: 17–33.
- Birch, P. B., R. S. Barnes & D. E. Spyridakis, 1980. Recent sedimentation and its relationship with primary productivity in four western Washington lakes. *Limnol. Oceanogr.* 25: 240–247.
- Birks, H. J. B., J. M. Line, S. Juggins, A. C. Stevenson & C. J. F. ter Braak, 1990. Diatoms and pH reconstruction. *Phil. Trans. R. Soc., Lond. B* 327: 263–278.
- Carignan, R. & R. J. Flett, 1981. Postdepositional mobility of

- phosphorus in lake sediments. *Limnol. oceanogr.* 26: 361–366.
- Dillon, P. J. & F. H. Rigler, 1974. The phosphorus-chlorophyll relationship in lakes. *Limnol. Oceanogr.* 19: 767–773.
- Eisenreich, J. J., R. T. Bannerman & D. E. Armstrong, 1975. A simplified phosphorus analysis technique. *Environmental Letts.* 9: 43–53.
- Engstrom, D. R. & H. E. Wright Jr., 1985. Chemical stratigraphy of lake sediments as a record of environmental change. In J. W. G. Lund & E. Y. Haworth (eds), *Lake sediments and environmental history*. University of Leicester Press, Leicester: 11–67.
- Forsberg, C. & S.-O. Ryding, 1980. Eutrophication parameters and trophic state indices in 30 Swedish lakes. *Arch. Hydrobiol.* 89: 189–207.
- Foy, R. H., 1985. Phosphorus inactivation in a eutrophic lake by the direct addition of ferric aluminium sulphate: impact on iron and phosphorus. *Freshwat. Biol.* 15: 613–629.
- Foy, R. H. & A. G. Fitzsimons, 1987. Phosphorus inactivation in a eutrophic lake by the direct addition of ferric aluminium sulphate: changes in phytoplankton populations. *Freshwat. Biol.* 17: 1–13.
- Fritz, S. C., S. Juggins, R. W. Battarbee & D. R. Engstrom, 1991. Reconstruction of past changes in salinity and climate using a diatom-based transfer function. *Nature* 352: 706–708.
- Gibson, C. E., 1986. A contribution to the regional limnology of Northern Ireland: the lakes of County Down. *Record of Agricultural Research (Dept. Agric. NI)* 34: 75–83.
- Gibson, C. E., 1988. Contributions to the regional limnology of Northern Ireland (ii): the lakes of County Fermanagh. *Record of Agricultural Research (Dept. Agric. NI)* 36: 121–131.
- Gibson, C. E., 1989. Contributions to the regional limnology of Northern Ireland: (3) the lakes of Co. Armagh. *Ir. Nat. J.* 23: 41–47.
- Golterman, H. L., R. S. Clymo & M. A. M. Ohnstad, 1978. *Methods for the physical and chemical examination of freshwaters*. Blackwell Sci. Pub., Oxford.
- Heyman, U. & A. Lundgren, 1988. Phytoplankton biomass and production in relation to phosphorus: some conclusions from field studies. In G. Persson & M. Jansson (eds), *Phosphorus in Freshwater Ecosystems. Developments in Hydrobiology* 48. Kluwer Academic Publishers, Dordrecht: 211–227. Reprinted from *Hydrobiologia* 170.
- Hörnström, E., 1981. Trophic characterization of lakes by means of qualitative phytoplankton analysis. *Limnologia* 13: 249–261.
- Jewson, D. H., B. H. Rippey & W. K. Gilmore, 1981. Loss rates from sedimentation, parasitism, and grazing during the growth, nutrient limitation, and dormancy of a diatom crop. *Limnol. Oceanogr.* 26: 1045–1056.
- Jørgensen, E. G., 1957. Diatom periodicity and silicon assimilation. *Dansk Bot. Arkiv* 18: 1–54.
- Kamp-Neilsen, L., 1986. Modelling the recovery of hypertrophic L. Glumsø (Denmark). *Hydrobiol. Bull.* 20: 245–255.
- Line, J. M. & H. J. B. Birks, 1990. WACALIB version 2.1 – a computer program to reconstruct environmental variables from fossil assemblages by weighted averaging. *J. Paleolimnol.* 3: 170–173.
- Marsden, M. W., 1989. Lake restoration by reducing external phosphorus loading: the influence of sediment phosphorus release. *Freshwat. Biol.* 21: 139–162.
- Mayer, L. M., F. P. Liotta & S. A. Norton, 1982. Hypolimnetic redox and phosphorus cycling in hypertrophic lake Sebasticook, Maine. *Wat. Res.* 16: 1189–1196.
- Murphy, J. & J. P. Riley, 1962. A modified single solution method for the determination of phosphate in natural waters. *Analyt. Chim. Acta.* 27: 31–36.
- Reynolds, C. S., 1984. *The ecology of freshwater phytoplankton*. Cambridge University Press.
- Reynolds, C. S., 1987. The response of phytoplankton communities to changing lake environments. *Schweiz. Z. Hydrol.* 49: 220–236.
- Reynolds, C. S., H. R. Morrison & C. Butterwick, 1982. The sedimentary flux of phytoplankton in the south basin of Windermere. *Limnol. Oceanogr.* 27: 1162–1175.
- Rosén, G., 1981. Phytoplankton indicators and their relations to certain chemical and physical factors. *Limnologica* 13: 263–290.
- Sas, H., (ed.) 1989. *Lake restoration by reduction of nutrient loading: expectations, experiences, extrapolations*. Academia Verlag, St. Augustin.
- Schindler, D. W., 1987. Detecting ecosystem response to anthropogenic stress. *Can. J. Fish. aquat. Sci.* 44, Suppl. 1: 6–25.
- Shapiro, J., W. T. Edmondson & D. E. Allison, 1971. Changes in the chemical composition of sediments of Lake Washington, 1958–1970. *Limnol. Oceanogr.* 16: 437–452.
- Stevenson A. C. [and others] 1991. *The Surface Waters Acidification Project Palaeolimnology Programme: modern diatom/lake-water chemistry data set*. Ensis Ltd., London 86 pp.
- ter Braak, C. J. F., 1987 *CANOCO – A FORTRAN program for Canocial Community Ordination*. Manual. Wageningen., TNO Insitute of Applied Computer Science.
- ter Braak, C. J. F. & I. C. Prentice, 1988. A theory of gradient analysis. *Adv. ecol. Res.* 271–317.
- ter Braak, C. J. F. & H. van. Dam, 1989. Inferring pH from diatoms: a comparison of old and new calibration methods. *Hydrobiologia* 178: 209–223.
- Whitmore, T. J., 1991. Sedimentary diatom concentrations and accumulation rates as predictors of lake trophic status. In J. P. Smith; P. G. Appleby, R. W. Battarbee, J. A. Dearing, R. Flower, E. Y. Haworth, F. Oldfield & P. E. O'Sullivan (eds), *Environmental History and Palaeolimnology. Developments in Hydrobiology* 67. Kluwer Academic Publishers, Dordrecht: 163–169. Reprinted from *Hydrobiologia* 214.
- Wilson, H. E., 1972. *Regional geology of Northern Ireland*. HMSO, Belfast.