

EUTROPHICATION:
RESEARCH AND APPLICATION TO WATER SUPPLY

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DAVID W. SUTCLIFFE AND J. GWYNFRYN JONES

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Changes of water quality and sediment phosphorus of a small productive lake following decreased phosphorus loading

S. I. HEANEY¹, J. E. CORRY AND J. P. LISHMAN

Freshwater Biological Association and Institute of Freshwater Ecology, Far Sawrey, Ambleside, Cumbria, LA22 0LP, England.

Esthwaite Water is the most productive or eutrophic lake in the English Lake District. Since 1945 its water quality has been determined from weekly or biweekly measurements of temperature, oxygen, plant nutrients and phytoplankton abundance. The lake receives phosphorus from its largely lowland-pasture catchment, sewage effluent from the villages of Hawkshead and Near Sawrey, and from a cage-culture fish farm. From 1986 phosphorus has been removed from the sewage effluent of Hawkshead which was considered to contribute between 47% and 67% of the total phosphorus loading to the lake. At the commencement of phosphorus removal regular measurements of phosphorus in the superficial 0–4 cm layer of lake sediment were made from cores collected at random sites. Since 1986 the mean annual concentration of alkali-extractable sediment phosphorus has decreased by 23%. This change is not significant at the 5% level but nearly so. There has been no marked change in water quality over this period. Summer dominance of blue-green algae which arose in the early 1980s after decline of the previous summer forms, *Ceratium* spp., has been maintained. Improvement in water quality is unlikely to be achieved at the present phosphorus loading.

Introduction

There is general agreement that the degree of phosphorus loading to many lakes determines the level of algal production or eutrophication. The relationship between phosphorus loading and algal biomass was assessed by Vollenwider (1968), and the causal effect of phosphorus on algal production was illustrated in the whole-lake experimental studies of Schindler (1974) and in the large-scale mesocosms by Lund (e.g. Lack & Lund 1974). These and other studies have resulted in remedial measures being taken to reduce the external phosphorus inputs to many lakes suffering the severe effects of eutrophication.

Numerous case studies now exist of the reduction of anthropogenic inputs of phosphorus to lakes and reservoirs (e.g. Sas 1989). It is apparent that success in reducing the biomass of algal populations is variable and rarely immediate. A complicating factor is that phosphorus stored in the lake sediment under conditions of anoxia (e.g. Mortimer 1941, 1942; Heaney, Smyly & Talling 1986) and at high pH (e.g. Mackereth, 1953; Drake & Heaney 1987) may be released to the overlying water as an internal load. The release of such phosphorus to the water column at critical times of phosphorus depletion may give rise to algal production in addition to that expected for external loading alone.

The present paper describes the effect of the reduction of sewage-borne phosphorus to a small English lake. Esthwaite Water is generally agreed to be the most productive or eutrophic

¹ Present address: Aquatic Sciences Research Division, Department of Agriculture, Newforge Lane, Belfast, BT9 5PX, Northern Ireland.

lake in the English Lake District (Pearsall 1921; Jones 1972; Gorham *et al.* 1974). The lake is a grade 1 Site of Special Scientific Interest and together with some fenland to its north, including Priest Pot, has been included in the list of Wetlands of International Importance under the Ramsar convention. Despite artificial enrichment by sewage and fish-farm effluents, the lake is one of the best examples in England and Wales of a mesotrophic system. At its northern end is a well-developed and studied hydrosere. The lake contains a nationally rare plant species (*Najas flexilis* (Willd.) Rostk, & Schmidt) and other species of restricted distribution. The lake has been the subject of much study (e.g. Heaney, Smyly & Talling 1986, and references therein). More recently, the phosphorus content of its sediments and their potential for remobilization (Drake & Heaney 1987) and the long-term physical, chemical and biological changes (Heaney *et al.* 1988; Talling & Heaney 1988) have been investigated.

Following a report to the Nature Conservancy Council (Talling & Heaney 1983) which concluded that the sewage treatment plant at Hawkshead was the major point source of phosphorus input to the lake, a decision was taken by the North West Water Authority (NWW) to prevent most of the sewage-borne phosphorus from entering the lake by undertaking chemical precipitation of this nutrient at the sewage works. At the same time it was agreed that the phosphorus removal would be accompanied by a programme of research to determine the effect of decreasing external phosphorus load to the lake on water quality and algal growth. This research had three objectives. The first was to provide a better estimation of the amounts of total phosphorus and soluble reactive phosphorus entering and leaving the lake. This work was undertaken by NWW (Agar *et al.* 1988) and showed that, prior to phosphorus removal, the contribution of phosphate-phosphorus in the effluent from Hawkshead sewage treatment plant was only 47% of the total phosphate-phosphorus loading to the lake, compared to the value of 67% estimated by Talling & Heaney (1983). It was apparent that the earlier study had underestimated the non-sewage contribution from the catchment.

It is the second two objectives that are the subject of the present paper. One was to determine from regular sampling of the lake any changes of water quality before and after phosphorus removal. The other was to determine any changes in the potential internal sediment load of phosphorus, by measuring the sodium hydroxide extractable phosphorus content of the sediment of the lake for a number of years before and after phosphorus removal at the sewage treatment works.

Methods

Water quality

Regular sampling was undertaken at a buoy near the deepest position of the lake (A in Fig. 1). Vertical profiles of water temperature ($\pm 0.1^\circ\text{C}$) and oxygen (resolution $6 \mu\text{mol l}^{-1}$) were measured *in situ* using a combined thermistor and oxygen electrode.

Secchi disc transparency was obtained with a white disc of 30 cm diameter. Water samples for chemical analyses and algal counts were collected using a weighted algal tube (Lund 1949) from the 0–5-m layer. A sub-sample was immediately fixed in the field with Lugol's iodine for subsequent algal counts using the Utermöhl inverted microscope technique (Lund *et al.* 1958).

Total phosphorus, soluble reactive phosphorus, ammonia-nitrogen and dissolved silicon were analysed as described in Mackereth *et al.* (1989) with the $\text{NH}_4\text{-N}$ method adapted for use with a discrete analyser, and ascorbic acid as the reductant for dissolved silicon. Nitrate-nitrogen was analysed as described by Hilton & Rigg (1983).

Chlorophyll-*a* was determined after separation of algae on glass-fibre filters (Whatman GF/C), extraction with boiling methanol, and estimation as the chlorophyll optical equivalent (Talling 1974).

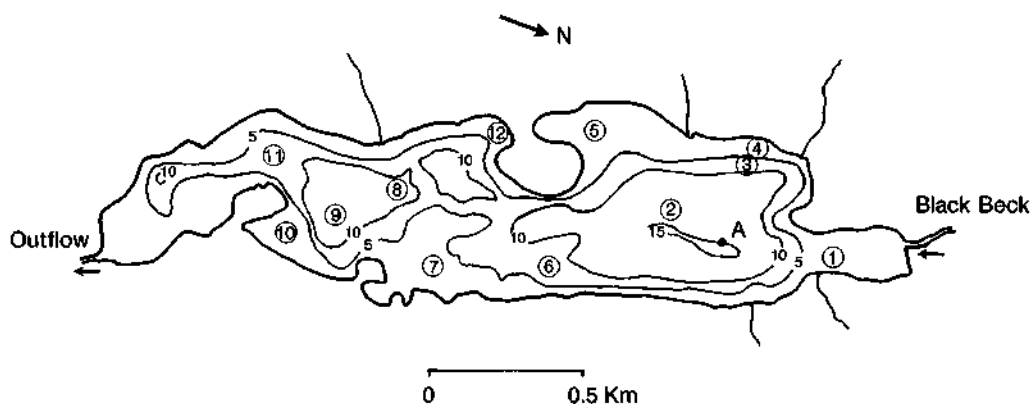


Figure 1. Map of Esthwaite Water showing bathymetric contours (in m), approximate sampling positions for obtaining cores (1-12), and sampling buoy for water quality data (A).

Sediment analysis

Sediment cores were collected according to a randomly stratified sampling design near the positions indicated in Figure 1. Twelve cores were taken where possible for each sample which included three deep (> 5 m water depth) and three shallow (< 5 m water depth) sites in each of the two main basins of the lake. For a period only eleven cores were obtained due to the loss of a corer of imperial dimensions which could not be replaced. Cores were taken using a Jenkin sampler, and the upper 0-4 cm sediment layer was obtained using a hydraulically-operated extruder (Jones & Ohnstad 1982).

The total phosphorus content of the sediment was determined on a sub-sample of 0.1 g dried material from the upper 0-4 cm layer. Each sub-sample was digested to dryness in a teflon beaker after the addition of 2 x 1 ml Aristar nitric acid. Further digests were made in the following order: 2 x 2 ml Analar perchloric acid (70%), 2 x 2 ml Analar hydrofluoric acid (48%) and 1 x 2 ml Analar perchloric acid (70%). The residue was taken up in 50 ml of 0.1 M hydrochloric acid. Suitable aliquots were neutralized and phosphorus measured using the lower sensitivity method for phosphate in Mackereth *et al.* (1989).

Inorganic soil phosphate or alkali-extractable phosphorus was determined on further sub-samples from the 0-4 cm layer after the method of Petersen & Corey (1966). Approximately 2 g wet sediment was accurately weighed into 10 ml plastic tubes. At the same time a separate sub-sample was taken for dry weight determination and dried at 60°C for 1-2 days. 8 ml of 0.125 M NaOH was added to each tube containing the weighed 2 g of wet sediment, stoppered and mixed gently until homogenised. The tubes were fitted onto racks in a Denby RM200 rotator and rotated at *c.* 20 r.p.m. for 17 hours to extract the phosphate. It was determined by experimentation that no phosphate was obtained after further extractions. The suspension was then centrifuged for 15 min at 2400 r.p.m. and the supernatant transferred to another plastic tube. 40 µl of concentrated H₂SO₄ was added to the suspension and swirled gently to flocculate the organic matter, and then centrifuged again for 15 min. at 2400 r.p.m. A suitable aliquote was then taken for analysis of phosphate as described above.

Results

Changes in sediment phosphorus content

Analysis of total phosphorus is very time consuming and two such measurements were only made on samples collected shortly after phosphorus removal from the sewage effluent began

during April 1986, and again during early 1990. The sample for 1986 was collected on 11 June but, as the core from site 1 was not available for analysis, a value for a core obtained on 9 July from this site is used. Table 1 gives the results for both total phosphorus and alkali-extractable phosphorus (AEP) at individual sites. The mean values and standard deviations for total phosphorus and AEP, respectively, for the 1986 sample are 4152 ± 1811 and 2040 ± 1383 , and for the 1990 samples they are 3791 ± 1821 and 1500 ± 1334 $\mu\text{g P}$ per g sediment dry weight. These show decreases in the mean total phosphorus and AEP of 9 and 26% respectively, although the actual concentration decreases are similar at 361 and 507 $\mu\text{g P}$ per g sediment dry weight.

Table 1. Concentrations of total phosphorus (TP) and alkali-extractable phosphorus (AEP) in the top 0–4 cm layer of sediments at 12 sites in Esthwaite Water (see Fig. 1) in 1986 and 1990

Samples were taken on 9 July 1986 (site 1) and 11 June 1986 (sites 2–12), and all sites were sampled again on 7 March 1990. Sites with an asterisk are where the sediments lie below deeper water (5–15m).

Values for TP and AEP are in $\mu\text{g P}$ per g dry sediment; AEP is also given as a percentage of TP at each site.

| Sampling site | 1986 | | | 1990 | | |
|---------------|------|------|-------------|------|------|-------------|
| | TP | AEP | AEP/TP, (%) | TP | AEP | AEP/TP, (%) |
| 1 | 2735 | 1153 | 42 | 2966 | 857 | 29 |
| 2* | 5278 | 3108 | 59 | 4719 | 2771 | 59 |
| 3* | 3385 | 1522 | 45 | 3672 | 1193 | 32 |
| 4 | 3462 | 1430 | 41 | 3678 | 1025 | 28 |
| 5 | 2487 | 943 | 38 | 1647 | 475 | 29 |
| 6* | 4114 | 1707 | 41 | 3518 | 1087 | 29 |
| 7 | 3464 | 1299 | 38 | 2427 | 585 | 24 |
| 8* | 5853 | 4018 | 67 | 5916 | 2990 | 51 |
| 9* | 9001 | 5372 | 60 | 8180 | 4684 | 57 |
| 10 | 3243 | 1241 | 38 | 2659 | 684 | 26 |
| 11 | 3663 | 1321 | 36 | 4113 | 1618 | 39 |
| 12 | 3143 | 1360 | 43 | 1999 | 428 | 21 |

Table 2. Mean concentrations \pm standard deviation of total phosphorus (TP) and alkali-extractable phosphorus (AEP) in sediments of Esthwaite Water, in 1986 and 1990

Results are given for whole lake sediment (12 sites combined), sediment from the north and south basins of the lake (6 sites each), shallow sediment (6 sites lying under 0–5 m depth of water), and deep sediment (6 sites lying under 5–15 m depth of water); N is the number of sediment cores

Values for TP and AEP are in $\mu\text{g P}$ per g dry sediment; AEP is also given as a percentage of TP at each site.

| Year | Site | N | TP | AEP | AEP/TP (%) |
|------|-------------|----|-----------------|-----------------|------------|
| 1986 | Whole lake | 12 | 4152 ± 1819 | 2040 ± 1383 | 49 |
| 1990 | Whole lake | 12 | 3791 ± 1821 | 1533 ± 1299 | 40 |
| 1986 | Shallow | 6 | 3089 ± 339 | 1238 ± 173 | 40 |
| 1990 | Shallow | 6 | 2563 ± 720 | 675 ± 230 | 26 |
| 1986 | Deep | 6 | 5216 ± 2084 | 2843 ± 1625 | 55 |
| 1990 | Deep | 6 | 5020 ± 1776 | 2390 ± 1378 | 48 |
| 1986 | North basin | 6 | 3577 ± 1013 | 1643 ± 767 | 46 |
| 1990 | North basin | 6 | 3367 ± 1016 | 1235 ± 793 | 37 |
| 1986 | South Basin | 6 | 4782 ± 2323 | 2437 ± 1802 | 52 |
| 1990 | South Basin | 6 | 4216 ± 2415 | 1832 ± 1695 | 43 |

Table 2 indicates the changes in AEP from the two major basins and different depth strata in the lake relative to those for total phosphorus. For the whole lake between 1986 and 1990, AEP

as a percentage of total phosphorus declined by 11% from 46 to 35%. The largest change occurred in the 0–5-m depth stratum where there was a decrease of 14% compared to only 6% for the 5–15-m stratum. There was little difference between the north and south basins.

Although total phosphorus was analysed on only two occasions, EAP was measured on samples collected monthly for the first year and thereafter at three-monthly intervals. As there were very large differences in the concentration of sediment phosphorus throughout the lake, indicating a contagious distribution, arithmetic mean values of samples and subsets are given in Figures 2–4 with 95% confidence limits calculated using Taylor's power law transformation (Elliott, 1977). On the few occasions when the analysis for a site was not obtained, a value was assigned by taking the mean of the time-adjacent values for the same site. These interpolated values have been used in all statistical analysis. Comparison of time-series using the analysed values only, and with interpolated values included, show similar overall trends (Fig. 2). The incorporation of interpolated values, however, gives less erratic mean values for samples, especially where analyses for site 9 were missing. This deep-water site always contained at least twice the amount of sediment phosphate found at any other site, and its absence from a sample-collection always decreased the sample mean.

The changes in mean phosphate values for all samples against time are given in Figure 2. The 95% confidence limits are very wide for each date, indicating the large differences in phosphate concentration of the superficial sediment throughout the lake. There was a general decline in the mean phosphate concentration between 1986, when phosphate removal from the sewage effluent began, and 1989. The mean annual phosphate content for 1986–1989 (Table 3) indicates a year-by-year gradual decline in phosphate concentration and a 23% reduction in the phosphate content of the sediment over this time interval.

Table 3. Annual mean concentrations of alkali-extractable phosphorus (AEP) in sediments of Esthwaite Water, in 1986 and 1990

Means for AEP, with 95% confidence limits and variances, are given for sediment cores taken at 12 sampling sites on various occasions in 1986 to 1989; N is the number of sub-samples (sediment cores).

Values are in $\mu\text{g P per g dry weight of sediment}$; the last column shows the cumulative decrease in AEP as a percentage of the mean for 1986.

| Year | N | Mean | 95% CL | Variance | Decrease (%) |
|------|-----|------|-------------|----------|--------------|
| 1986 | 120 | 2081 | 1784 – 2387 | 269501 | — |
| 1987 | 72 | 1927 | 1541 – 2313 | 2685468 | 7 |
| 1988 | 48 | 1723 | 1332 – 2114 | 1800830 | 17 |
| 1989 | 49 | 1611 | 1220 – 2002 | 1782732 | 23 |

To determine if there was a statistically significant decrease in the phosphate content of the sediment between 1986 and 1989 the mean annual concentrations for both years were compared using the "normal deviate (d) test". For both years the number of sub-samples were sufficiently great that the *central-limit theorem* was considered applicable. Thus it was assumed that the probability distribution of the sample means around the sediment mean was normal, with the overall mean equal to the sediment mean (Elliott 1977). The null hypothesis was that both samples came from the same sediment and therefore their means were not significantly different. The calculated value for d was 1.91, indicating that the sample means were not significantly different, although nearly so, at the 5% level. The variances were not significantly different at the 5% level, with the variance ratio, F , = 1.51.

Differences in sediment phosphate between shallow-water (0–5 m) and deep-water (5–15 m) sites is shown in Figure 4. This shows that shallow-water sites contain much less phosphate than deep-water sites.

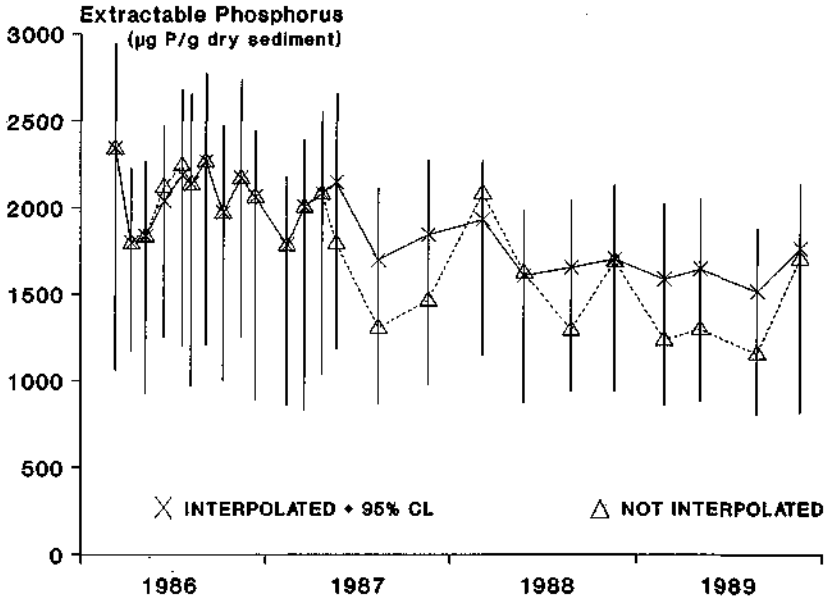


Figure 2. Time-series of the mean concentration of alkali-extractable phosphorus in the 0-4 cm layer of sediment from samples of cores from Esthwaite Water. Values with 95% CL are for samples including interpolated values (x); values for samples not including interpolated values (Δ) are given for comparison (further explanation in the text).

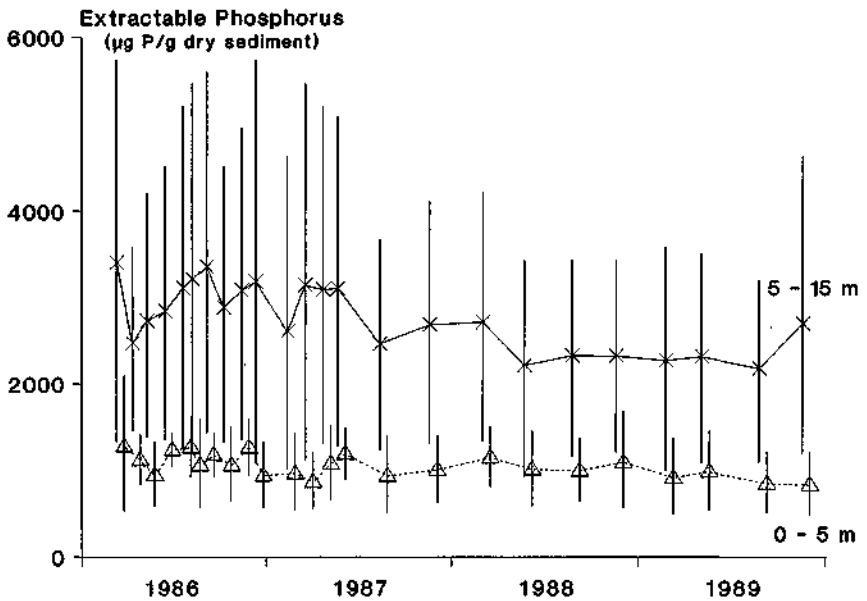


Figure 3. Time-series of the mean concentration and 95% CL of alkali-extractable phosphorus in the upper (0-4 cm) sediment layer of cores from Esthwaite Water, separated into sub-samples underlying shallow (Δ , 0-5 m) and deep (x, 5-15 m) water. Note that values for shallow water samples are offset to the right.

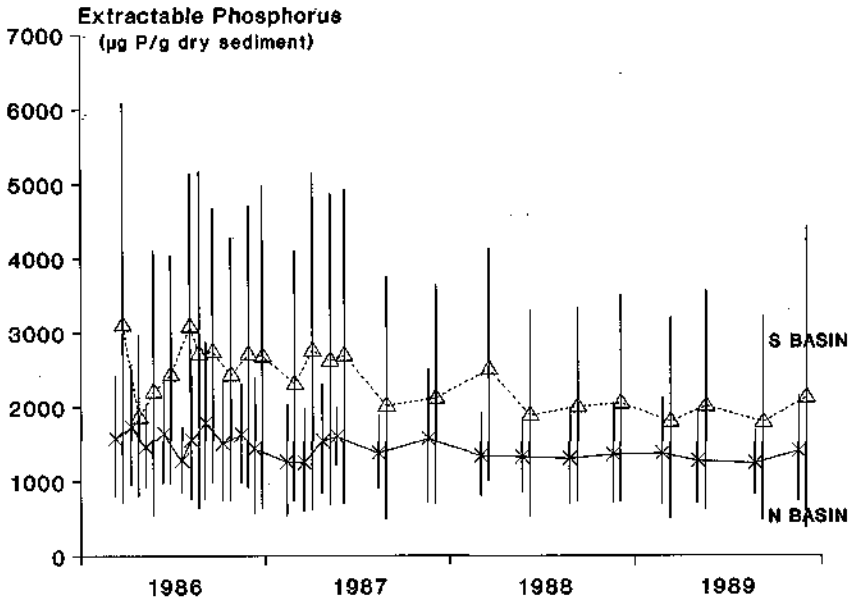


Figure 4. Time-series of the mean concentration and 95% CL of alkali-extractable phosphorus in the upper (0–4 cm) sediment layer of cores from Esthwaite Water, separated into sub-samples from its north (x) and south (Δ) basins. Note that values for the south basin are offset to the right.

Esthwaite Water is divided into two main basins separated by the peninsula of Strickland Ees, with a third and smaller basin at its southern end (Fig. 1). In this study cores were only taken from the north and south basins. The mean annual values of sediment phosphate in the south basin were consistently higher than those in the north basin (Figure 4). Using all the data for 1986–1989 the difference between the sample means from the two basins was highly significant ($d = -5.19$; $P < 0.01$) and the variances were significantly different at the 0.2% level ($F = 8.09$). The consistently very high values of sediment phosphorus at site 9 contributed strongly to the between-basin differences. If values for site 9 are excluded, the sample means for the two basins are not significantly different ($d = -1.75$; $P > 0.05$), although the variances were ($F = 4.25$; $P < 0.002$).

Changes in lake water quality

(a) *Phosphorus*. Seasonal changes in the concentration of total phosphorus in the 0–5-m water layer at the sampling buoy (A, Fig. 1) are shown in Figure 5. There is little pronounced seasonality, although there tends to be a minimum at the end of April associated with the decline and sedimentation of the spring diatom population, and an increase during late summer when blue-green algal blooms occur. Winter concentrations are variable and to some degree influenced by weather-dependent resuspension of sediment.

The mean annual concentrations of total phosphorus (Table 4) surprisingly show that the value for 1986 was lower than for subsequent years. The mean annual value for 1988 was significantly higher, at the 5% level, than for 1986 (Table 5).

As with total phosphorus, the mean annual concentrations of soluble reactive phosphorus show that the value for 1988 was significantly greater than that for 1986 (Table 4). Seasonal changes in the concentration of soluble reactive phosphorus during 1986–1989 (Fig. 6) show

the expected winter maxima and summer minima. The latter, although generally close to the limit of detection of the method ($0.6 \mu\text{g SRP l}^{-1}$), show a number of short-lived increases associated with the decline and mineralisation of the algal populations (c.f. Fig. 7).

Table 4. Annual mean concentrations of total phosphorus (TP), soluble reactive phosphorus (SRP) and chlorophyll-a in lakewater (0–5 m depth) in Esthwaite Water, in 1986 to 1989

Means ($\mu\text{g per litre of lakewater}$) and 95% confidence limits for TP, AEP and Chl a are given for 42–44 samples taken each year at the sampling buoy (see Fig. 1).

Values are also given for Secchi-disc transparency (m depth of water) for 40–44 samples in each year.

| | 1986 | 1987 | 1988 | 1989 |
|---------|------------------|------------------|------------------|------------------|
| TP | 26.7 (24.2–29.2) | 29.7 (26.5–31.9) | 30.1 (28.4–31.8) | 29.3 (27.7–30.9) |
| SRP | 2.5 (1.4–3.6) | 3.7 (2.4–5.0) | 4.6 (3.0–6.2) | 3.9 (2.5–5.3) |
| Chl a | 22.0 (16.2–27.8) | 24.4 (18.8–30.0) | 26.7 (20.6–32.8) | 25.3 (20.4–30.2) |
| Secchi | 2.7 (2.3–3.1) | 2.5 (2.1–2.9) | 1.9 (1.6–2.3) | 2.3 (2.0–2.6) |

Table 5. Results of statistical comparisons between annual means shown in Table 4 for total phosphorus (TP), soluble reactive phosphorus (SRP), chlorophyll-a and Secchi-disc transparency in Esthwaite Water, in the years 1986 to 1989

d is the normal deviate (d test) and F is the variance ratio.

Asterisks denote significance at the 95% level (*), 98% level (**), and 99.8% level (***); all other values are not significant at the 95% level.

| | 1987 | | 1988 | | 1989 | |
|---------|-------|------|---------|--------|--------|---------|
| | d | F | d | F | d | F |
| TP | -1.82 | 1.34 | -2.25* | 2.13** | -1.74 | 2.30*** |
| SRP | -1.44 | 1.30 | -2.18* | 2.03** | -1.59 | 1.51 |
| Chl a | -0.60 | 1.06 | -1.14 | 1.09 | -0.88 | 1.42 |
| Secchi | 0.92 | 1.37 | 3.61*** | 1.91** | 2.00** | 2.31*** |

(b) *Light attenuation and chlorophyll-a.* The depth at which a white Secchi disc of 30 cm diameter disappears from view when lowered into the water gives a useful measure of transparency. For Esthwaite Water there is usually an inverse correlation between Secchi depth and chlorophyll-a concentration, as phytoplankton biomass is a major agent of light attenuation (Talling & Heaney 1983). Changes, with time, of Secchi transparency and the concentration of chlorophyll-a are shown in Figure 7. There is considerable interannual variation, with the usual "clear water phase" present in May–June 1986 and 1987, after the sedimentation of spring diatom growth, but poorly developed in 1988 and 1989 due to the early growth of blue-green algal populations.

The mean values for Secchi transparency and chlorophyll-a concentration are given in Table 4. As with total phosphorus and soluble reactive phosphorus, the mean annual value of chlorophyll-a for 1988 was the highest of the four years, although not significantly higher than in 1986 (Table 5).

(c) *Major algal species.* There was no marked change of algal quality during the period of decreased sewage input. The major algal genera producing the chlorophyll-a maxima are shown in Figure 7. The spring maxima consisted of *Asterionella formosa* Hass. and *Melosira italica* subsp. *subarctica* O. Müller (= *Aulocoseria subarctica* (O. Müller) Haworth). The overall biomass of the spring diatom growth is generally determined in this lake by silicon availability. Summer communities were dominated by the blue-green algae *Aphanizomenon flos-aquae* f. *gracile* Ralphs ex Born. et Flah., *Anabeaena flos-aquae* Bréb. ex Born. et Flah.,

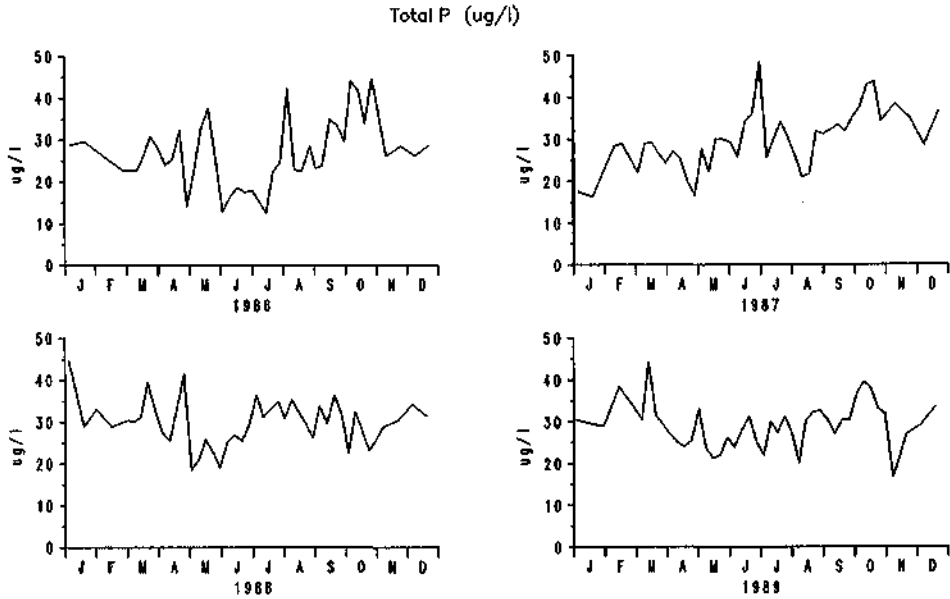


Figure 5. Time-series for concentrations of total phosphorus in the upper (0-5 m) water layer of Esthwaite Water during the years 1986-1989.

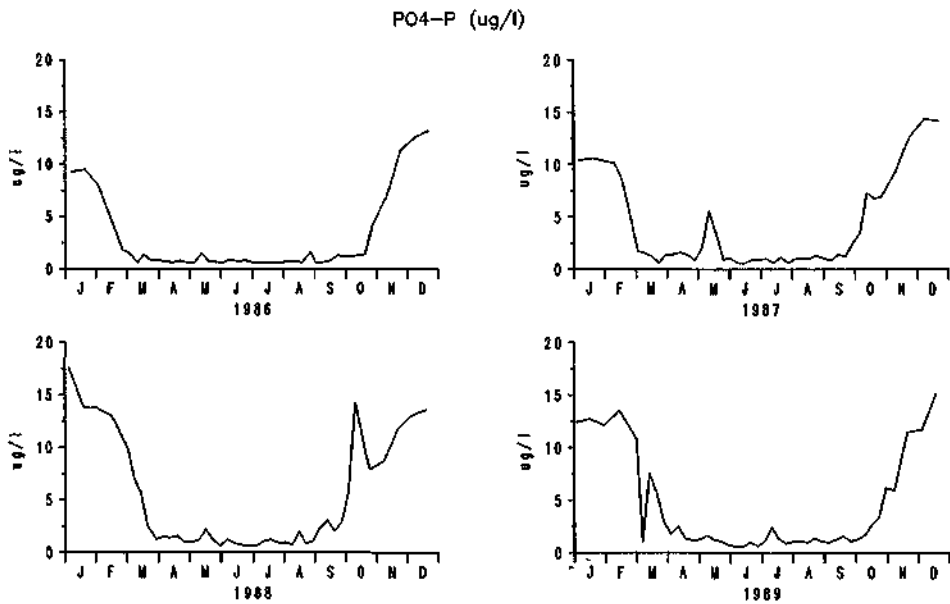


Figure 6. Time-series for concentrations of soluble reactive phosphorus in the upper (0-5 m) water layer of Esthwaite Water during the years 1986-1989.

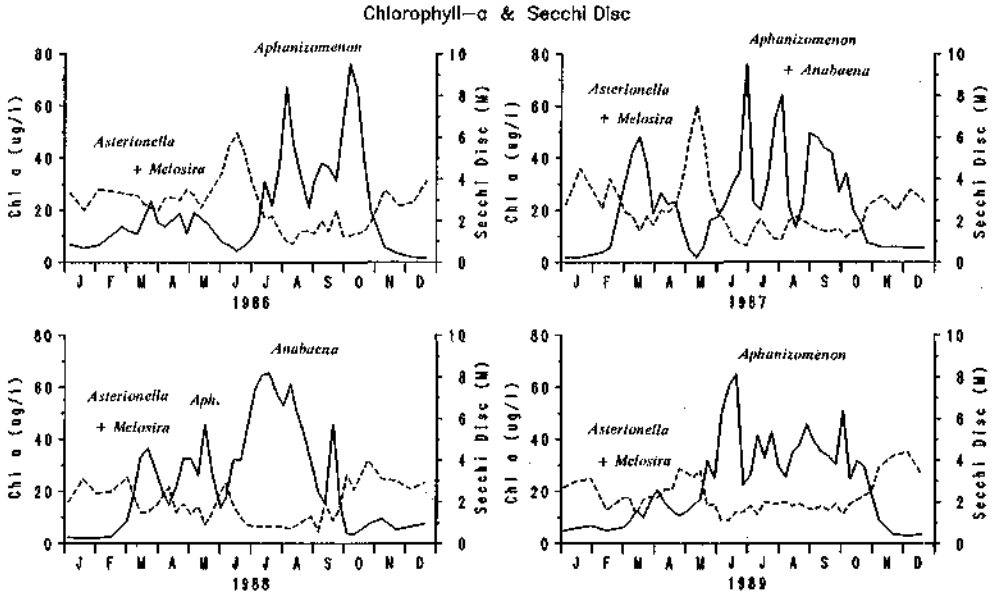


Figure 7. Time-series for concentrations of chlorophyll-a in the 0-5 m layer (—) and for Secchi disc transparency (---) in Esthwaite Water during the years 1986-1989. The dominant algal genera during chlorophyll-a maxima are indicated.

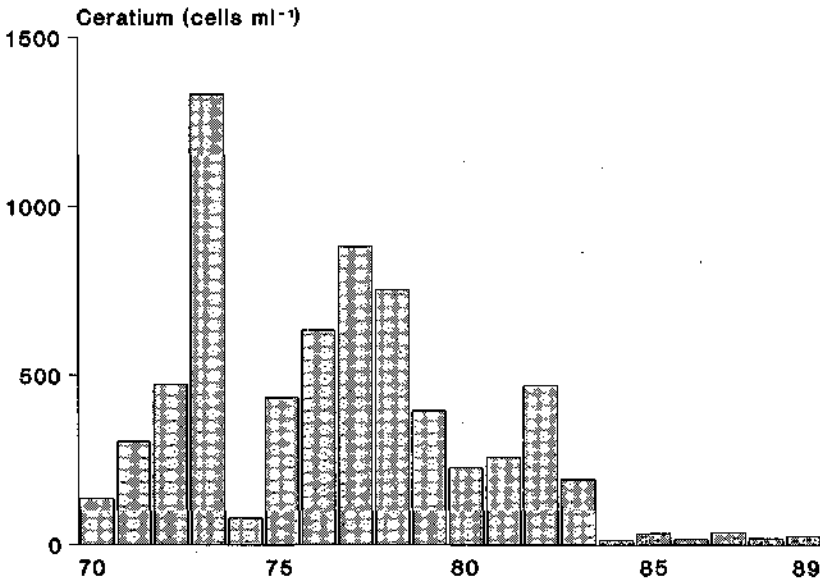


Figure 8. Maximum density of *Ceratium* spp. (cells ml⁻¹) in the upper 0-5 mm layer of Esthwaite Water from 1970 to 1989. The great decrease in maximum annual density between 1983 and 1989 is the result of parasitism caused by the biflagellate fungus *Aphanomyces cryptica*.

and *A. solitaria* Kleb. There was considerable interannual variability in the timing of maxima of certain species of blue-green algae, particularly *Aphanizomenon flos-aquae*. This was caused in part by an unusual grazing episode during summer 1987, by the protozoan ciliate *Nassula*, together with the relatively long hydraulic residence time of the lake, enabling the persistence of slow-growing and poorly-grazed winter populations (Canter, Heaney & Lund 1990). The dinoflagellates *Ceratium* spp. which dominated the summer populations during the 1970s and early 1980s produced relatively small populations (Fig. 8) because of repeated parasitism by the biflagellate fungus *Aphanomyopsis cryptica* Canter (Canter & Heaney 1984; Heaney *et al.* 1988).

Discussion

These results complement the study of Agar *et al.* (1988) which determined the external inputs of soluble reactive phosphorus and total phosphorus to Esthwaite Water between 1985/1986 and 1987/1988, including estimates of the discharge from the cage-culture fish farm into the south basin. This showed that over this period the total phosphorus input had decreased by 1240 kg year⁻¹. The total phosphorus contribution from the fish farm was estimated at c. 620 kg year⁻¹ of which about 60% is likely to be in the soluble phosphorus fraction and directly available for algal growth (Foy & Rosell 1991a). Agar *et al.* (1988) also concluded that, following phosphorus removal at the sewage treatment works, there was a decrease in the estimated annual net uptake of phosphorus by the lake, from 1830 kg for 1985/1986 to 330 kg for 1987/1988.

The results of the present study and that of Agar *et al.* (1988) thus indicate that the reduction in external phosphorus loading has been compensated for by an increase in internal loading from the sediments as the lake approaches a new equilibrium state. It is hardly surprising that until this is reached there have been no appreciable changes in the mean annual concentrations of lakewater total and soluble reactive phosphorus, and chlorophyll-*a*. The new equilibrium state, when it is reached, will depend upon the overall external loading derived from diffuse catchment inputs, sewage-borne phosphorus, now markedly reduced, and inputs from the fish farm. Phosphorus contribution from the latter is currently being determined and is dependent upon the level of fish production and the phosphorus content of the fish food (Foy & Rosell 1991b).

From the sediment core analyses, the mean lake sediment phosphate decreased by 358 µg P per g dry sediment between 1986 and 1988. Using a value of 90% for the water content of the upper 0–4 cm sediment layer, and a conservative estimate of 1 km² for the area of lake sediment, it can be calculated that about 1430 (= $1 \times 10^6 \times 10^4 \times 10^{-1} \times 358 \times 10^{-9}$) kg of phosphorus will have been given up in two years by the lake sediment. Acknowledging the possible errors in such an estimation, it can be compared with a similar reduction in the total external annual input of 1240 kg phosphorus estimated for almost the same period by Agar *et al.* (1988).

The annual external loading of total phosphorus and soluble reactive phosphorus for Esthwaite Water was estimated by Agar *et al.* (1988) to be 1.65 and 0.85 g P m⁻² respectively. These values, following the reduction of sewage-borne phosphorus, would maintain the lake's classification as eutrophic on the basis of the Vollenweider's earliest graphical relationship of phosphorus loading, phytoplankton abundance (as chlorophyll-*a*), and mean lake depth (Vollenweider 1968). However, on the basis of the greater range of variables considered by Vollenweider & Kerekes (1984), the lake would be classified as mesotrophic with regards to nitrogen and phosphorus levels but eutrophic with regard to chlorophyll-*a* concentration and water clarity.

The greatest change in algal quality within the lake took place during 1983 prior to the reduction of phosphorus loading. Before this date the summer phytoplankton was dominated

by populations of *Ceratium* spp. (Heaney *et al.* 1988). Thereafter populations of blue-green algae or cyanobacteria have formed prolonged growths throughout summer and autumn and also occasionally during late spring following a high level of inoculum carry-over from the previous year. As a result there has been a general increase in the annual duration and abundance of phytoplankton growth as shown by chlorophyll-*a* concentration (Talling 1992). This change of quantity and quality of phytoplankton has resulted from a progressive enrichment of the lake (Heaney *et al.* 1988; Talling & Heaney 1988) but more significantly because of the decline of populations of the dinoflagellates *Ceratium* spp. as a result of persistent parasitism by the biflagellate fungus *A. cryptica*. Diseases of phytoplankton and microbial grazing are well documented but the demise of a regular dominant phytoplankton by such means is rarely recorded. The failure of populations of *Ceratium* spp. may in part be due to their relatively slow growth rate, making them unable to recover from fungal attack before nutrient and light resources are used by competing populations of cyanobacteria.

The impact of phosphorus removal for changes of water quality has yet to be fully realized. Indeed it may be reasonably questioned if it ever will with the present phosphorus loading, although the removal of the sewage-borne phosphorus is nevertheless a step in the right direction. Even with continuous year-round removal of phosphorus by the sewage treatment works, at present not undertaken during the winter months, it is likely to be some years yet before an improvement in water quality, i.e. a reduction of summer algal biomass and an increase in water transparency, can be expected. The magnitude of the phosphorus input to the lake from the fish farm may be critical in determining the ultimate success or failure in improving water quality in this lake.

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