Institute of Freshwater Ecology

BASSENTHWAITE LAKE: AN ASSESSMENT OF THE EFFECTS OF PHOSPHORUS REDUCTION AT THE KESWICK STW ON THE SEASONAL CHANGES IN NUTRIENTS AND PHYTOPLANKTON, USING A DYNAMIC MODEL

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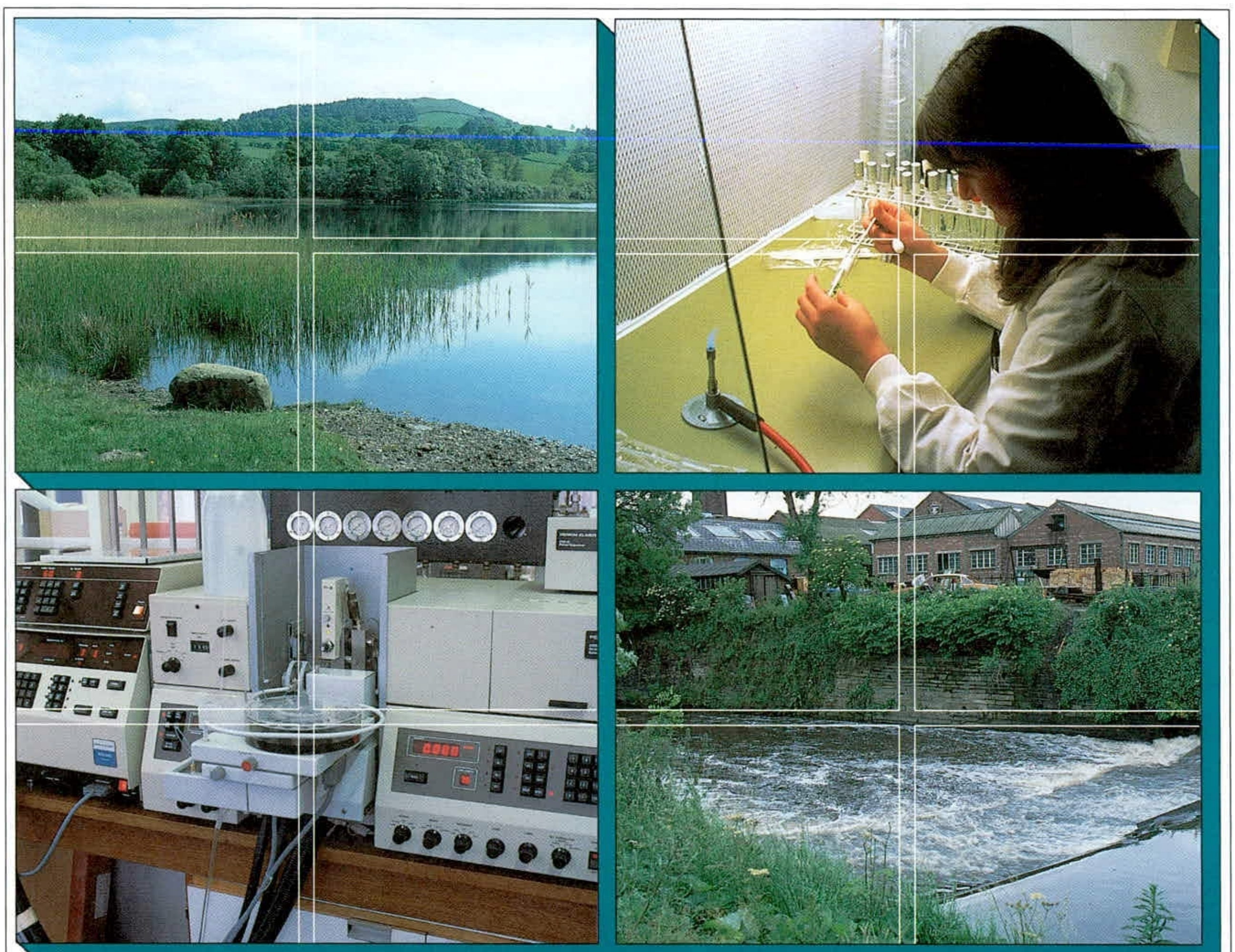
Report to National Rivers Authority, NW Region - November 1993



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Contract Completion Date: 30 November 1993
TFS Project No.: T11055y1
IFE Report No.: 1

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Summary

There is a public perception that increased enrichment of Bassenthwaite lake has resulted in deleterious algal blooms occurring from time to time. This view is, to some extent, supported by scientific evidence which shows a progressive enrichment of this water body and increased oxygen deficiency in the bottom waters in recent years. The latter is a particularly worrying aspect because it threatens the survival of the resident Vendace population, which is one of the only two remaining populations in the UK and is protected under the Wildlife and Countryside act, 1981.

Analysis of chemical data for stream and rivers flowing into Bassenthwaite show that at least 78% of the phosphorus entering the lake in the form most readily used by algae (orthophosphate, OP), comes from sewage treatment works, while only 22% is derived from agricultural runoff. Effluent from the Keswick works, in particular, constituted 72% of the total OP input.

A dynamic model incorporating data on water and nutrient inputs, lake morphometry and initial populations of phytoplankton, produced good simulations of the changes observed in nutrients and algae in Bassenthwaite Lake for the period 1 January to 31 August 1993. The analyses suggest that, especially during early 1993, the phytoplankton biomass was reduced, compared to a dry year, by the effect of large volumes of water passing through the lake. However, the availability of OP controlled the size of the algal populations throughout the year, and particularly so from June onwards. Hence reductions in the P input are likely to reduce algal biomass.

The model suggests that a planned reduction of 80% of the phosphate in the Keswick STW effluent (equivalent to *ca* 55% of the total input of this nutrient to the lake in 1993) would result in some improvement in water quality in the short term. However, the major benefits would accrue over a 10- to 20-year timescale and would result in significant reductions of algal biomass compared to the present day.

Contents

1. INTRODUCTION

1.1 Background

Bassenthwaite is one of the larger water bodies in the English Lake District. It is classified as a Grade I SSSI on account of its Vendace population which is 1 of only 2 remaining in the UK (Maitland and Lyle 1991) and is protected under the Wildlife and Countryside Act, 1981. As the lake is extremely well-flushed (water throughput rate of *ca* 15 lake volumes y^{-1}), phytoplankton populations are fairly moderate, but the sedimentation of cells over the long-term may well explain gradual increases in the deficit of oxygen in the restricted hypolimnetic zone. of This is particularly worrying because it may threaten the survival of the Vendace which is a species that tends to congregate in the deeper, cool zones of the lake (Hilton and McEvoy 1993), and is particularly sensitive to hypolimnetic de-oxygenation. So any strategies resulting in a reversal of eutrophication trends are of interest. Nutrient inputs are likely to have increased during the last 40 years due to many factors including increased population in the catchment, and elevated use of phosphorus (P) -based detergents, but no data on nutrient loads were available until very recently.

1.2 Aims of the study

The main aims of this project are as follows:

(i) to identify the sources of nutrients within the catchment, with particular emphasis on phosphorus,

(ii) to simulate, using a dynamic model, the observed dynamics of nutrients and phytoplankton

(iii) to quantify the main components driving algal growth in Bassenthwaite and to identify the major factors limiting the accumulation of algal biomass.

(iv) to model nutrient and algal changes assuming 80% removal of the ortho-phosphate (OP) from the Keswick STW effluent (see also Bailey-Watts *et al* 1992a).

1.3 Approaches adopted

Nutrient loadings were estimated from NRA data on the levels of dissolved ortho-phosphate (OP), nitrate-nitrogen (NO_3-N), silicate-silica (SiO_2) and the flows for the feeder waters and outflow of Bassenthwaite Lake. IFE provided the information on the dynamics of in-lake nutrients, physical conditions and phytoplankton populations. Reynolds' PROTEC1 model version 1 was used to simulate observed changes in nutrients and phytoplankton over the period January 1 to August 31 (1993), and to predict the patterns assuming an 80% reduction in the OP derived from the Keswick STW. IFE information on thermal stratification was also taken into account in assessing the likelihood of phosphate release from the sediments.

2. THE DATABASES

2.1 NRA NW Region

Of the data supplied by NRA, the following were used as inputs to the model: estimated flow rates and the concentrations of OP, NO_3-N and SiO_2 in 11 feeder streams and rivers. Sites above and below the outfalls of treated sewage effluent from Keswick STW were included. **Figure 1** shows the sampling points in relation to the lake, while **Table 1** gives site names and locations, and indicates the relative importance of each feeder water on the basis of the size of the area drained relative to that of the only continuously flow-gauged site, Portinscale.

2.2 IFE Windermere

The main body of data considered in conjunction with the model and supplied by IFE concerns the observed changes in surface water temperature of the lake, and the concentrations of nutrients and phytoplankton (as chlorophyll$_a$ and the densities of cells, colonies and filaments of the various species). For the purposes of the present project an 8-month run of information (1 January to 31 August 1993) was used. Water samples were collected from open water at fortnightly intervals, although extremely rough weather on 1 March prevented access onto the lake.

Table 1. Sampling sites for inflow water chemistry: * numbered clock-wise from Derwent inflow (= 1); ** ratio of catchment size to that of Portinscale (continuous flow gauging only at Portinscale).

Inflow*	Name	Grid ref.	Catchment ratio**
1a	River Derwent at Portinscale (above Keswick STW)	NY 251 268	1.000
1b	River Derwent at Lowstock Bridge (below Keswick STW)	NY 251 242	1.050
2	Newlands Beck below Chapel Beck	NY 230 262	0.136
3	Un-named tributary downstream of Thornthwaite STW	NY 255 263	0.002
4	Beckstones Gill	NY 224 265	0.002
5	Beck Wythop	NY 214 283	0.004
6	Dubwath Beck	NY 201 312	0.018
7	Un-named tributary	NY 208 320	0.002
8	Chapel/Halls/Dash Becks	NY 215 310	0.063
9	Pooley Beck	NY 218 308	0.002
10	Tributary near Bowness Bay	NY 224 298	0.005
11	Skill Beck	NY 227 287	0.009

3. THE MODEL

The model used is a functional one which applies limnological knowledge to the problems of enhanced algal growth, and biomass accumulation in particular. It has many advantages over steady-state models, which cannot deal with dynamic situations, and statistical models, which require extensive calibration.

A forerunner of this model was developed jointly by IFE and Welsh Water under contract to Messrs. Wallace Evans and Partners, for use in the Cardiff Bay Barrage study (Reynolds 1989a). Reynolds (1984b, 1989b) describe the mathematical bases of the present model, while Hilton, Irish and Reynolds (1992) describe the model itself. The essentials are as follows. Daily hydraulic flow rates, metabolisable phosphorus concentrations (approximated as OP levels here), dissolved silica (SiO_2) and nitrogen concentrations (taken as nitrate), are entered into the model for each feeder stream and sewage discharge. The model then calculates the OP, NO_3-N and SiO_2 in the lake assuming no utilisation by algae.

The light climate perceived by circulating algal cells is estimated from input data on incoming insolation, the mixed layer volume (taken here as the whole volume of Bassenthwaite), mixed and mean depths. Assuming no limitation by nutrients, 'inocula' of up to 8 algal species/species groups, characteristic of different types found in the waterbody in question, are allowed to grow at their own individual maximum rates, adjusted according to insolation and water temperature.

Algae are allowed to grow at their maximum rates (for a given temperature and light input) unless nutrient concentrations fall below a growth rate-saturating level. At these times the growth rate is reduced as a function of nutrient concentration. The model checks whether there are sufficient nutrients to sustain the calculated amount of growth (each day), by comparing the nutrient ratios with the 'ideal' Redfield ratio (Redfield 1934). The model provides for a more efficient use of nutrients - especially OP - when external concentrations fall below certain low thresholds. If no limitation is suggested, the originally estimated (end-of-day) biomass is retained and the appropriate amounts of OP, NO_3-N and SiO_2 are removed from the nutrient pool. Otherwise, the biomass is increased by an amount allowed by the limiting nutrient. The rate-limiting nutrient is reduced to zero and the other nutrients are reduced accordingly.

By classifying the different algae according to size, the growth responses of those susceptible to grazing by zooplankton (rates defined by input information on temperature and zooplankton population densities), can be made to differ from those not eaten by these animals. Equally, N-fixing cyanobacteria are 'treated' differently from other algae in respect of the effects of NO_3-N depletion. As a result of zooplankton grazing, OP and NO_3-N are assumed to be released instantaneously and re-introduced into the available nutrient pools. Nutrients in solution, and algal cells are washed out of the system at a rate proportional to the rate of water throughput (flushing). As with all other terms, the model program cycles to calculate these values daily.

4. INVESTIGATIVE METHODS, DATA HANDLING AND ANALYSIS

4.1 Feeder water hydraulic and nutrient loads

As the model requires daily data for the period of interest, discharges (m^3 s^{-1}) and nutrient concentrations (mg m^{-3}) for the 'non-sampled' days for each feeder water were generated in one of two ways. Missing OP values were calculated by linear interpolation between the measured values as most sites had been sampled at 2- or 7-day intervals. In contrast, for NO_3-N and SiO_2, which had been measured much less frequently, missing values were obtained by different means. For each nutrient a number of mathematical relationships were examined between (a) the measured nutrient concentrations and the estimated flows at the instants of sampling, and (b) the product of these concentrations and flows (i.e. instantaneous loads, mg s^{-1}) and the estimated flow at the time of sampling. The relationship described by the regression equation with the highest coefficient of determination (r^2 value) was then used to generate daily nutrient concentrations from the estimated daily flows. The equations derived from these regressions were used to predict 'missing' NO_3-N and SiO_2 concentrations are shown in **Table 2**, while **Figure 2** shows two examples of the plots obtained. This approach was not possible for NO_3-N at Chapel/Halls Beck, so linear interpolation was used instead. Values prior to the first measured value and after the last measured value were assumed to be the same as the first and last measured figures respectively, for every site and nutrient. An NO_3-N outlier for an un-named Beck at Broadness Farm was removed prior to calculating the regression for this site between flow and nutrient levels. In-lake phosphate, nitrate and silica levels were determined according to the procedures described below.

4.2 Estimating the OP load in sewage works effluent

4.2.1 Keswick STW

Effluent from the outfall pipe and storm overflow from Keswick STW enter the River Derwent between Portinscale and Lowstock Bridge. Flow and water chemistry data were available for both sites, i.e. upstream and downstream of the STW outfall. The amount of OP entering the Derwent from this works was estimated as the difference in loading between Portinscale and Lowstock Bridge.

Table 2. Equations used to predict missing nitrate (NO_3-N, n) and silica (SiO_2, s) concentrations in the inflow streams from their estimated rates of flow (f, Ml d^{-1}). Coefficients of variation (r^2 values) for the regressions from which the equations were derived are also given.

Site	NO_3-N concentration (mg m^{-3})	r^2	SiO_2 concentration (mg m^{-3})	r^2
R.Derwent at Lowstock Bridge	$n = 10^{-0.0004079f + 2.9057}$	0.7848	$s = 1950.51 - (75989.76/f)$	0.9344
Newlands Beck below Chapel Beck	$n = 10^{-0.002557f + 2.8550}$	0.5841	$s = 2078 + (9732/f)$	0.9742
Un-named tributary downstream of Thornthwaite STW	$n = 10^{1.3087\log(f) + 3.0687}/f$	0.7498	$s = 3221 + (198/f)$	0.9907
Beckstones Gill	$n = 621.82 - (128.51/f)$	0.8840	$s = 2327 + (468/f)$	0.9576
Beck Wythop	$n = 266 + (178/f)$	0.8200	$s = 2344 + (545/f)$	0.9870
Dubwath Beck	$n = 10^{0.7279\log(f) + 3.2901}/f$	0.7965	$s = 3792 + (1725/f)$	0.9368
Un-named tributary	$n = 1203 + (231/f)$	0.9105	$s = 4400 + (289/f)$	0.9677
Chapel/Halls/Dash Becks	---	---	$s = 2312 + (17323/f)$	0.9737
Pooley Beck	$n = 10^{1.4994\log(f) + 2.7189}/f$	0.7949	$s = 3150 + (1157/f)$	0.9126
Tributary near Bowness Bay	$n = (2429\log(f) + 502.17)/f$	0.7763	$s = 3700 + (97/f)$	0.9186
Skill Beck	$n = 547 + (349/f)$	0.8355	$s = 3617 - (212/f)$	0.9929

4.2.2 Bassenthwaite STW

The Bassenthwaite STW discharges into Halls Beck. No flow data were available for this site, so the average discharge consent of 0.175 Ml day^{-1} was used to calculate the OP load from this source. This was multiplied by the mean OP concentration of the effluent (2.95mg l^{-1}) which was calculated from values recorded at weekly intervals for the period February to May, 1993. This concentration was very similar to the mean of the average OP concentrations for 1990, 1991, 1992 and 1993 i.e. 2.90mg l^{-1}.

4.2.3 Thornthwaite STW

Neither discharge flows nor discharge consent were available for this works. However, as it is of similar size to the Bassenthwaite STW, it was originally agreed with NRA that a value of 0.175Ml day^{-1} could be used to estimate the likely average daily discharge at this site. The corresponding mean OP concentration in the effluent for the period February to May 1994 was 2.26mg l^{-1} which is similar to the average value for 1990 to 1993 of 2.31mg l^{-1}. However, the total OP load estimated by this method for the 8 months was found to be 97.2kg which is 61% higher than that (60.4kg) of the feeder stream into which it drained. These approximations were clearly not valid. Instead, it was assumed that the un-named beck which received the effluent from the Thornthwaite STW had the same flow (and diffuse, background OP concentration) as Beckstone Gill which was nearby, drained a similar sized area and received no sewage effluent. This suggested that the discharge from Thornthwaite STW was, in fact, only about 0.1Ml day^{-1}. The OP load from this works was obtained by subtracting the background load from Beckstone Gill from the measured load in the un-named tributary.

4.2.4 Armathwaite STW

The Armathwaite discharge is not consented and there are no data on flows for this site. It discharges directly into the lake close to the outflow and takes sewage only from a small private hotel. It is, thus, unlikely to make a significant contribution to the overall OP loading to Bassenthwaite. As such, it has been ignored throughout this study.

4.2.5 Embleton STW

This works, which serves 250 people year-round and an addditional 280 during the months June to August, discharges into Dubwath Beck, resulting in a relatively high OP loading for this stream (see below). The actual contribution of the STW has not been measured, but if an OP output of 0.5kg y^{-1} *per capita* of is assumed, a value of 133kg over the 8-month period is obtained.

4.3 Bassenthwaite surface water temperatures

Daily surface water temperatures (which the model requires for determining algal growth rate potentials) were obtained by linear interpolation into the series of fortnightly values measured.

4.4 Nutrients in the lake

Nutrient levels in the lake were determined on Whatman GF/C-filtered water using the following procedures. Partially automated spectrophotometric methods of Mackereth, Heron and Talling (1978), were used to determine the concentrations of OP, nitrate and silica.

4.5 The phytoplankton

Samples for phytoplankton analysis were taken from open water in the lake at fortnightly intervals with a weighted plastic tube to secure an integrated sample over the uppermost 5m of the water column. The tube was lowered vertically and slowly into the water. The upper end was then sealed with a rubber bung, the tube was recovered, and the water was transferred to previously rinsed 5-l plastic bottles. Samples were taken in duplicate. Total biomass was estimated as chlorophyll$_a$ concentration by concentrating the algae from a known volume of water onto Whatman GF/C pads. These were steeped in 90% methanol overnight and in the dark at *ca* 4°C to extract the pigment. A Shimadzu UV-150 spectrophotometer was used to read the absorbance of the centrifuge-cleared extracts at 665nm (with a reading at 750nm for turbidity correction). The equation of Talling and Driver (1963) was used to convert the spectrophotometric absorbances to concentrations of chlorophyll$_a$. Water for phytoplankton species determination, cell/colony/filament counts and size measurements was fixed with Lugol's Iodine. The relatively large (and often sparse) species were concentrated in sedimentation chambers prior to counting with an inverted microscope. Depending on general abundance level, the algae from volumes of water ranging from 1ml to 100ml were enumerated. The relatively small species were enumerated in a Lund nanoplankton counting chamber.

The chlorophyll$_a$ values and the abundance estimates of all but some 15 species that were small *and* sparse over the January to August 1993 period were transferred from IFE laboratory notebooks to an EXCEL spreadsheet. A further 'weeding out' of rare, unimportant species resulted in a core database with information on the 42 species indicated in **Table 3** and classified into the 8 morphological-physiological types which the model distinguishes and assigns specific growth characteristics:

(i) non-silica utilising, non-motile nanoplankton (e.g. small Chlorococcales)

(ii) small, grazeable silica-utilising algae (diatoms such as some

Stephanodiscus species and certain chrysophytes)

(iii) non-silica utilising motile nanoplankton (e.g. chlamydomonads)

(iv) nitrogen-fixing blue-green algae (e.g. *Anabaena*)

(v) as (ii) but large non-grazed forms (e.g. *Fragilaria*)

(vi) large, motile green algae (e.g. *Gemellicystis*, *Pseudosphaerocystis*)

(vii) large, non-motile green algae (e.g. desmids)

(viii) non-N fixing blue-green algae

Each algal group was also assigned three 'representative' morphometric values for volume, maximum dimension and surface area, respectively (**Table 4**).

The biomass of each algal group was next calculated in terms of chlorophyll$_a$ concentration. This involved calculating the biovolume (e.g. mm^3 l^{-1}) of each group and converting this to a chlorophyll$_a$ equivalent value according to literature values on μg chlorophyll$_a$ mm^{-3} cell volume.

Table 3. Bassenthwaite algal genera contributing significantly to the 8 morphological-functional categories of phytoplankton incorporated into the model.

(i) small, non-motile algae, not using silica	*Aphanocapsa* *Chlorella* *Cladomonas* *Coccomyxa* *Kirchneriella* *Monoraphidium* *Nephrochlamys* *Stichococcus* *Synechococcus*
(ii) grazed diatoms and other silica users	*Chrysococcus* *Chrysochromulina* *Cyclotella* *Kephyrion* *Monochrysis* *Pseudokephyrion* *Stephanodiscus*
(iii) small, motile algae, not using silica	*Chlamydomonas* *Cryptomonas* *Rhodomonas*
(iv) nitrogen-fixers	*Anabaena*
(v) ungrazed diatoms and other silica users	*Asterionella* *Aulacoseira* *Dinobryon* *Fragilaria* *Synedra S* *Synedra L* *Urosolenia*
(vi) large, motile algae, not using silica	*Eudorina* *Gemellicystis* *Gonium* *Paulschulzia* *Peridinium*
(vii) large, non-motile algae, not using silica	*Botryoccus* *Dictyosphaerium* *Koliella* *Micractinium* *Sphaerocystis* *Spondylosium* *Staurastrum*
(viii) large, non-N fixing blue-green algae	*Microcystis* *Oscillatoria limnetica* *Oscillatoria bourrellyi*

Table 4. Size values and other key descriptors representative of the 8 algal groups described in the text.

Algal group	volume (μm^3)	max. dimension (μm)	surface area (μm^2)	diatom?	grazed?	N fixer?
(i) small, non-motile forms (exc. diatoms)	33	4	50	N	Y	N
(ii) small diatoms	500	12	350	Y	Y	N
(iii) small, motile algae	33	4	50	N	Y	N
(iv) nitrogen fixing blue-green algae	9000	100	6000	N	N	Y
(v) large diatoms	40000	250	52000	Y	N	N
(vi) large motile green algae	2500	160	4300	N	N	N
(vii) large non-motile green algae	65000	50	7850	N	N	N
(viii) large non-N fixing blue-green algae	6800	350	5500	N	N	N

4.6 Other in-lake data

A temperature logger installed in Bassenthwaite at the end of July was run continuously until the end of the present study. This recorded values at the surface, at 2-m intervals down to 14m, then at 15m, 17m and 18.5m in the deep pot of the lake, thus giving information on thermal stratification.

5. OBSERVED HYDROLOGICAL, TEMPERATURE, NUTRIENT AND PHYTOPLANKTON FEATURES - January to August 1993

5.1 Hydraulic loads

Flow rates derived by assuming a *pro rata* discharge according to the ratio of each of these subcatchment areas to that of Portinscale are shown in **Table 5.** It is evident that some 95% of the hydraulic loading to Bassenthwaite is accounted for by only 3 of the 11 inflows i.e. 1, River Derwent (80.12%), 2, Newlands Beck (10.43%) and 8, Halls Beck (4.83%).

The total input of water over this period, i.e. the sum of the values for all inflows was 2.95 x 10^8 m^3. (The Coal Beck was ignored as it effectively flows directly into the outflow of the lake). If matched by outflow losses, this corresponds to a flushing rate of 10.6 lake volumes or a water residence time of 22.9 days over the 8-month period. However, a regression of the estimated inflow values ('estimated outflow' in **Figure 3**) on the values obtained from the NRA gauge at the outflow, shows that the estimated figures are consistently some 30% greater than the gauged values; the reasons for this inconsistency are being investigated. Temporal variability in the rate of water input is considerable (**Figure 4**). Daily mean inputs vary from <2 m^3 s^{-1} to 70 m^3 s^{-1} and even monthly mean values range over an order of magnitude (inset **Figure 4**). In these features, Bassenthwaite is like a number of other shallow systems in northern Britain e.g. Loch Leven (Bailey-Watts *et al* 1990) and Loch Dee (Bailey-Watts and Kirika 1993).

5.2 Phosphorus inputs

Dissolved ortho-phosphate (OP) is the main focus here, as it is the form of phosphorus which is most available to algae and is potentially the main growth-limiting nutrient. The River Derwent (inflow 1) accounted for almost 90% of the estimated total loading of 5223kg OP to Bassenthwaite over the 8-month period reviewed here (**Table 6**). The treated effluent from the Keswick STW which flows into the Derwent constituted 72% of the total OP income to the lake. The rest of the STW contributed only a further 6%. Of the 5 inflows (other than the Derwent) contributing more than 1% of the total OP load, 3 also carry sewage (Inflows 3, 6 and 8) while 2 do not (Inflows 2 and 7). The loading from inflow 2 (Newlands Beck) is large because this stream drains a large catchment.

Table 5. Hydraulic loadings to Bassenthwaite Lake for the 8-month period January to September 1993.

Inflow	Total water (m^3)	% water in
River Derwent at Lowstock Bridge	2.39×10^8	80.12
Newlands Beck below Chapel Beck	3.11×10^7	10.43
Un-named tributary downstream of Thornthwaite STW	4.57×10^5	0.16
Beckstones Gill	4.57×10^5	0.16
Beck Wythop	9.14×10^5	0.31
Dubwath Beck	4.11×10^6	1.38
Un-named tributary	4.57×10^5	0.16
Chapel/Halls/Dash Becks	1.44×10^7	4.83
Pooley Beck	4.11×10^6	1.38
Tributary near Bowness Bay	1.14×10^6	0.38
Skill Beck	2.06×10^6	0.69

A preliminary digest of NRA data on total P (TP) i.e. the sum of all P fractions, suggests that its loading is approximately 1.7 times that of OP. If it is also assumed that the OP load for the calendar year is 1.5 times that given for the 8-month study period in **Table 6** (i.e. 1.5 x 5223kg), the total yearly P input is 13,318kg or 2.52g TP m^{-2} lake surface. For a water, such as Bassenthwaite, of *ca* 5m mean depth, this specific areal loading is some 38 times the rate considered 'permissible' and approximately 20 times the 'dangerous' rate, as suggested by OECD (1982); these categories refer to the likelihood of algal blooms and other biological manifestations of nutrient enrichment.

5.3 Lake surface water temperature

The temperature of the lake at *ca* 0.5m depth increased at a more or less regular rate from 2°C at the start of the year to 12°C by the end of May (**Figure 5**). This was followed by a brief period of somewhat accelerated warming to 17°C in early June, then a subsequent cooling to *ca* 13°C and finally, a very gradual rise to 15°C by the end of August.

5.4 Nutrients in the lake

Although nutrient values in the open water of the lake represent what is unused (e.g. by phytoplankton) at the time of sampling, the low values for all three of the nutrients of major interest, even in winter, in Bassenthwaite are typical of a mesotrophic system (**Figure 6**). OP concentrations rarely exceeded 10mg m^{-3}, although the early August peak of *ca* 24mg m^{-3}, which is significantly higher than the early year (winter) values, is notable (see later). The average OP level calculated for the 8-month study period is 6.67mg m^{-3}. Nitrate-N levels also low, at no more than 500mg m^{-3} in winter, but they exhibited a more marked trend than OP. The concentrations fell from *ca* 450mg m^{-3} in January to virtually nil at the end of May. For the remainder of the study period levels of around 100mg m^{-3} prevailed. The mean nitrate-N concentration for this period is 253mg m^{-3}.

In winter, therefore, the nitrate-N to phosphate-P weight ratio was around 40:1, while ratios of *ca* 5:1 and 10:1 were more commonly found later on. On this basis, the lake would be viewed as progressing from a P-limiting to possibly an N-limiting state over the period of study.

The pattern of silica was much the same as that shown for nitrate but with winter and summer levels of approximately 2,500 and 500mg SiO_2 m^{-3}, respectively. The mean value for the 8-month period is 980mg m^{-3}.

5.5 Phytoplankton

Total phytoplankton abundance, as indicated by chlorophyll$_a$ concentrations, remained at very low levels throughout the first two months of 1993 (**Figure 7**) - almost certainly as a consequence of the very high rainfall over that period. For example, the total input of water in January has been estimated from the various flow gaugings and data manipulations outlined above, at 8.72×10^7 m^3. As this is 3.13 times the total volume of the lake (2.79×10^7 m^3), a mean hydraulic retention period of 9.9 days is calculated for this month. With such high rates of throughput of water, few algae could sustain a marked build-up of biomass however richly supplied with nutrients (Bailey-Watts *et al* 1992b; Bailey-Watts and Kirika 1993). Also, in winter and at this latitude, low insolation (and low temperatures to a lesser extent) does not favour rapid photosynthetic production (Talling 1971; Bailey-Watts 1988).

Only during March does algal biomass increase appreciably and in this case the growth continued into April when the maximum chlorophyll$_a$ concentration recorded over the study period was achieved (i.e. 36mg m^{-3}). Thereafter, pigment levels fluctuated between troughs of *ca* 12mg m^{-3} in late April, early July and late August, and the two remaining peaks in abundance - of around 30mg m^{-3} in early June and mid-August. For the period as a whole, the mean chlorophyll$_a$ concentration is 14.9mg m^{-3}.

As is commonly the case in phytoplankton studies, chlorophyll$_a$ values calculated from the sum of the products of algal species biovolumes and assumed chlorophyll$_a$ contents per unit cell volume, rarely account fully for the measured chlorophyll$_a$ levels. This is almost certainly due to the fact that some of the pigment measured as 'good' chlorophyll$_a$ is actually pheophytin - a breakdown product of the healthy pigment but with a similar absorbance spectrum. Thus, the mean, measured value of 14.9mg m^{-3} is indicative of a eutrophic system, while the actual levels of algae would place this water in a less rich category. Nevertheless, the 3 major peaks in pigment concentration coincide with maxima of large diatoms, while small (grazed) diatoms also contribute significantly to the March pulse (**Figure 8**). Otherwise, although small non-motile algae are relatively prominent in May, no algal group attains much by way of biomass over this particular study period. Note the differing scales on the y axes of the graphs in **Figure 8**. Hereafter, where the observed algal data are compared to various model simulations a single scale is used, as this illustrates better the relative importance of each algal group.

A comparison of the algal data with the information on nutrients, water inflow rate and temperature suggests that no algae could build up biomass during the very wet early part of the study period. Also, in spite of warmer weather as the year progressed, occasional episodes of high water throughput rate prevented any major bloom formations after the spring. Indeed, coupled with the fact that

Table 6. Orthophosphate loadings (kg P) from the 11 inflows to Bassenthwaite Lake over the 8-month study period; the total loading is 5223kg.

Inflow	OP load	% of total OP load
River Derwent at Portinscale (above Keswick STW)	881.8	16.88
River Derwent at Lowstock Bridge (below Keswick STW)*	4638.8	88.81
Newlands Beck below Chapel Beck	119.9	2.30
Un-named tributary downstream of Thornthwaite STW**	60.4	1.16
Beckstones Gill	1.8	0.04
Beck Wythop	3.3	0.06
Dubwath Beck***	131.6	2.52
Un-named tributary	53.9	1.03
Chapel/Halls/Dash Becks****	161.2	3.09
Pooley Beck	9.4	0.18
Tributary near Bowness Bay	15.8	0.30
Skill Beck	26.6	0.51

* takes Keswick STW effluent, 3757kg; ** takes Thornthwaite STW effluent, 58.6kg; *** takes Embleton STW effluent, 133kg; **** takes Bassenthwaite STW effluent, 125.4kg.

nitrate and silica were gradually depleted, it is not surprising that diatoms and a number of other forms relying on a fixed nitrogen source exhibited low biomass. In this connection, it is of note that the N-fixing *Anabaena* is relatively prominent at the end of the study period, although *Oscillatoria* (which does not fix atmospheric nitrogen) also reached its peak biomass at this time.

5.6 Making sense of the main features - hydraulic and phosphorus balances, and the relationship between P loading and the in-lake concentration

While by no means as refined as the model used later in this study, existing eutrophication models (e.g. Dillon and Rigler 1974, 1975; Kirchner and Dillon 1975; Vollenweider 1975; OECD 1982) based on correlative relationships between mean annual in-lake TP concentrations, TP loadings and various physical features of a waterbody, can be used to check whether the data such as those available for Bassenthwaite make sense. This approach has been found to be very useful in a wide variety of situations (Bailey-Watts *et al* 1992b; Bailey-Watts and Kirika 1993; Bailey-Watts in press). If, for Bassenthwaite, the relationships referred to above between OP inputs measured over 8 months and an annual TP load are assumed, a mean annual value of 22.2mg TP m^{-3} is obtained. If, also for reasons outlined above, this is taken to represent 1.7 times the OP figure, a mean annual concentration of *ca* 13mg OP m^{-3} is predicted. This is approximately double the mean value of 6.67mg m^{-3} observed. However, the observed value refers to the 8-month study period - not the year as a whole. It thus takes no account of the autumn-winter period during which higher than average OP levels can be expected. In addition, much of the OP entering a lake is changed into particulate form, so the fact that the actual OP value is less than the predicted figure is, again, to be expected. If these considerations are taken into account, this analysis suggests that the present data are in accord with the basic tenets of these models.

6. MODEL SIMULATIONS OF THE OBSERVED NUTRIENT AND PHYTOPLANKTON DYNAMICS

In the first instance the simulation was run using initial values specific to Bassenthwaite Lake (**Tables 4 and 7**) where these were available, and standard default settings where they were not. Daily values for lake water temperature, rate of flow and nutrient concentrations in the feeder waters, and rate of flow in the outflow, were supplied as inputs to the model. At this stage it was assumed that all the expected links between nutrient availability, flushing rate or zooplankton grazing would operate to control the observed chlorophyll$_a$ levels and that there was no recycling of nutrients from the sediments.

Table 7. Initial values used for model simulations of Bassenthwaite Lake phytoplankton and nutrients.

Number of inflows: 11
Number of outflows: 1
Surface area: 5.284×10^6 m^2
Volume: 2.79×10^7 m^3

Initial in-lake nutrient concentrations: 10.4mg OP m^{-3}
412mg NO_3-N m^{-3}
2410mg SiO_2 m^{-3}

Proportion of total phytoplankton chlorophyll$_a$ in each of 8 species categories (see Table 4):
(i) 0.003 (ii) 0.001 (iii) 0.03 (iv) 0.001
(v) 0.95 (vi) 0.01 (vii) 0.004 (viii) 0.001

The results (**Figure 9**) are encouraging. Temporal shifts in the predicted and measured concentrations of NO_3-N and SiO_2 are very similar. The early part of the chlorophyll$_a$ curve was well-represented but the predicted chlorophyll$_a$ levels were lower than those observed from the middle of March onwards - although only markedly so around the time of the 3 measured maxima in early April, early June and mid-August. There was little correspondence at all between the 2 curves for OP, with observed levels being much higher than predicted.

Ungrazed (i.e. large) diatoms dominated the Bassenthwaite phytoplankton especially during the first 4 months of this study period, in both the measured data and the values predicted according to the first run of the model (**Figure 10**). In common with the situation shown for total chlorophyll$_a$, the populations of these algae were only markedly under-estimated for the period associated with the early spring peak. In contrast, the levels of grazed (i.e. small) diatoms were consistently under-estimated by this run of the model. The predicted curves for the abundances of the other groups of algae varied in their similarity to the observed fluctuations, but these organisms represented such a minor component of total phytoplankton biomass, that they can be ignored.

The model predicts the maximum biomass of algae that the lake can sustain at a specific time given its temperature, light climate, flushing characteristics and nutrient inputs (Reynolds 1992). It is not possible for the lake to sustain algal concentrations higher than predicted if the factors such as nutrient supplies controlling algal growth have been correctly represented in the model. The predicted biomass ceiling can only be raised by manipulating the limiting factor; such as to promote algal biomass accumulation. In order to identify, for the present situation, whether OP, flushing or both of these factors are limiting, it is possible within the model to turn off their effects and look at the chlorophyll$_a$ values which are predicted as a result. If the predicted levels increase when a particular effect is 'turned off', this factor is likely to have been limiting the accumulation of algal biomass at that time. **Figure 11** shows that, for Bassenthwaite Lake, chlorophyll$_a$ levels would increase, especially in the spring, if flushing was less (and losses of algal cells down the outflow were reduced) or if phosphorus levels were increased. However, the model suggests that by far the most important factor limiting pigment levels in June, July and August, was the availability of OP, while hydraulic flushing had no significant effect over this period. The results of similar manipulations of NO_3-N are not plotted since they showed that this factor is having little effect.

There are four possible reasons why the initial predictions could have been inadequate:

(i) There was an un-accounted source of P from the catchment, permitting greater algal production.

(ii) A high inoculum of algae entered the lake from Derwentwater, allowing higher algal biomass to accumulate in a given time.

(iii) Algal cells were re-suspended from the sediments - acting as an inoculum permitting higher algal biomass to develop in a given time.

(iv) There was an internal source of P, most probably from the sediments, enhancing algal production.

An un-accounted external P source was considered unlikely as the catchment sampling strategy had been thorough in both spatial and temporal terms. Similarly, although some algae from Derwentwater undoubtedly reach Bassenthwaite, simulations showed that even if there were no cells were lost during passage from one lake to the other, the inoculum would contribute very little to the observed growth in spring. There are, however, grounds for believing that all three major chlorophyll$_a$ peaks were due in part to pulses of P from the sediment. The August maximum at least, followed an overturn event after a short period of thermal stratification. This is evident from **Figure 12** where the lines showing temperature at different depths in the lake over time diverge, indicating stratification, and then converge due to thorough mixing. Because of relatively high water temperatures (≥ 15°C) at the start of such short-term stratification periods, the sediments and the overlying water de-oxygenate very quickly, and iron compounds in the sediments are reduced and dissolved P is released into the sediment (pore) waters and the water immediately above the sediment. A sudden mixing event then circulates this P so that it is available for algal growth. The likely influence of algae re-suspended from the sediments will be discussed later.

Several pulses of P from the lake deposits were simulated in the next model run. The best fits to the observed data (**Figures 13** and **14**) resulted from the following pulses:

(i) for a 13-day period 17 March to 29 March inclusive, with daily inputs of 200kg OP on the first three days, and 300kg on each of the other days

(ii) an input of 50kg d^{-1} 13 May to 20 May, 100kg d^{-1} from 21 May to 3 June, 200kg d^{-1} on 4 and 5 June, 30kg d^{-1} on 6 and 7 June, 200kg on 8 June, and 100kg d^{-1} on 9 and 10 June

(iii) 50kg d^{-1} for the 6-day period 15 to 21 July, and

(iv) 100kg d^{-1} 2 to 11 August, 200kg on 12 August and 300kg on 13 August.

The possibility that wind-induced resuspension of algal cells from the sediments could give rise to 'extra' chlorophyll$_a$ should not be ignored. Moreover, the filamentous diatom *Melosira* (now *Aulacoseira*) which dominated the phytoplankton maximum recorded in March, is just the type of organism that can appear suddenly in greatly enhanced numbers due to this mechanism. The over-estimation of the uptake of nitrate and silica in March in the best-fit simulations which include sediment P suggest that the March peak, at least, contained some algae from this source. However, it is impossible to quantify the extent of this contribution to the algal numbers at that time. On the one hand, it might be assumed that no re-suspension took place (a worst case scenario as far as the simulation exercise is concerned), while at the other extreme, the assumption would be that all the algae were derived from the sediments, and no P was released. A proper mass balance for total P in Bassenthwaite (*cf* the exercise carried out as described in section 5.6) could help to establish the magnitude of the overall contribution of internal recycling of this nutrient from the sediments.

These simulations (**Figures 13, 14**) are considerably better than those produced by the first run as regards chlorophyll*a* from mid-March onwards. The predicted NO_3-N and SiO_2 values are still fairly close to the observed concentrations but there are only slight improvements in the fit of the predicted OP concentrations and the abundances of large diatoms, to their measured values. One possible explanation for the discrepancy between the observed and measured OP levels is that the model assumes that OP is taken up instantaneously by growing algae. Experience suggests that this is an over-simplification, and that OP actually remains in the water column for some days after release (from senescing cells or from sediments) before being used.

The new simulation of changes in algal species with time (**Figure 14**) highlight well the dominance of diatoms at the March peak. However, from mid-March onwards, the model suggests that large motile green algae become a significant proportion of the total biomass, while this was not the case in nature. This discrepancy is due to a slight over-estimation of the uptake of silica by diatoms, which resulted in its complete depletion in late March (**Figure 13**). As a result, large green algae developed populations which were dense enough in mid-April even though silica levels had recovered, to compete effectively with diatoms. In reality, however, silica was not completely depleted, (although low levels were reached) so green algae never attained very significant population densities.

It is reasonable to conclude that, with the inclusion of pulsed inputs of OP from the lake sediments, the model predicts very well the general course of events in Bassenthwaite. Thus, the model must have included the dominant controlling factors within its assumptions. However, further simulations were carried out - still with pulsed releases of sediment P, but with the links between biomass accumulation and limiting factors turned 'off'. These produced almost exactly the same curves as shown for the initial run of the model (**Figure 11**). This suggested that chlorophyll$_a$ levels were still, for the most part, OP-limited.

7. LIMITATIONS TO THE GROWTH OF ALGAE IN BASSENTHWAITE AND THE PROBABLE EFFECTS OF AN 80% REDUCTION OF ORTHO-PHOSPHATE AT THE KESWICK STW

Having produced an acceptable simulation of the general course of events in Bassenthwaite Lake for 1993, it was possible within the model to reduce the estimated OP output from the Keswick STW by 80% and look at the effects of this strategy on chlorophyll$_a$ levels in the lake. As mentioned above, OP appeared to be the major factor limiting algal productivity. It was thus expected, that a reduction in the amount of OP in the inflow would lead to reduced chlorophyll$_a$ levels in the lake.

These expectations were realised (**Figure 15**). Reduced chlorophyll$_a$ values were predicted, with the main algal peak of *ca* 32mg m^{-3} (in March) dropping to 24mg m^{-3}. However, the overall reduction in pigment levels is not large. The probable reason for this is that a significant proportion of the OP which is available to the algae comes from the sediments, that is, an extra supply to the external loading. Indeed, while an 80% reduction in the loading of OP from the Keswick works amounts to a 55% decrease in the total loading of OP from the catchment, it is equivalent to only 22% of the combined external and sediment-derived OP load. It follows, that if 80% of the OP were removed from the Keswick STW effluent, the magnitude of the change in chlorophyll$_a$ would depend upon the amount of OP recycled from the sediments. **Figure 16** includes 2 chlorophyll$_a$ curves that are predicted for two extremes after reduction in the external OP supply; these are (i) with no release of OP from the sediments to offset the reduction in external P loading, and (ii) with recycling to the (appreciable) extent indicated for 1993. Situation (i), which is simulated by the lower curve in **Figure 16** is unattainable since there are sources of P in the catchment apart from the Keswick STW, and these would continue to contribute

P to the sediments, and thus extend the potential for release of OP. Hence, the actual seasonal curve of pigment concentration would lie somewhere between the lines corresponding to situations (i) and (ii) in the Figure. It would, however, take some considerable time for this shallow lake to achieve this change.

8. DISCUSSION

The model predicts that reductions in chlorophyll$_a$ may not be large immediately after removal of OP from Keswick STW. In part this is a result of rapid flushing which allows a lake such as Bassenthwaite to sustain high P loadings without manifesting dense algal blooms. In contrast, Loch Leven (which is similar in mean depth, and receives P at only one-third of the rate) produces phytoplankton crops with chlorophyll$_a$ values of >100mg m^{-3}. This is because it has a very much lower flushing rate than Bassenthwaite (i.e. *ca* 2 lake volumes y^{-1} *cf* 16 lake volumes y^{-1})). It is for this reason that Bassenthwaite appears to be relatively insensitive, in the short term, to OP removal from Keswick sewage works.

Nevertheless, Keswick sewage treatment works contributes 72% of the external load of OP to Bassenthwaite. A significant proportion of this, and other inputs, is (and has been) trapped in the lake sediments. Some of this stored P is then recycled allowing algal crops to attain much higher biomasses than would otherwise occur. No research to date has been carried out on the state of the sediment store. If it is not saturated, continued input of P from external sources would increase the size of that store, allowing algal biomass to rise in proportion to the increase in recycling of P. A reduction in P input to the lake would reduce the amount of P being stored in the sediment for later recycling. As a result of washout of recycled P, the sediment P content would slowly decline as a new dynamic equilibrium developed. At the present time, we have no way of estimating the length of time required for the lake to exhibit these changes, but informed opinion suggests a period of 10-20 years. If the P removal option is pursued, more work should be done to improve the estimates of the time required to reach the new equilibrium.

There is a good possibility that the removal of OP from Keswick STW would have a larger short-term effect than the model suggests. In each year the reduction will affect mainly the first ('spring') phytoplankton maximum of the year. Such peaks consist largely of diatoms (Bailey-Watts 1988), which settle rapidly, taking a relatively large proportion of the production of these 'heavyweights' of the algal world to the bottom deposits (Reynolds 1984a; Bailey-Watts 1976; Bailey-Watts, Smith and Kirika 1989). Lowering this peak would thus be expected to reduce the subsequent long-term build-up of sedimentary P and, more important, reduce the amount of recycling within the same year due to breakdown of the deposited algae (Marsden 1988; Bailey-Watts, May and Kirika 1991; Bailey-Watts, Gunn and Kirika 1993). It should be noted here that it is the magnitude of the P release later in the year in Bassenthwaite which reduces the immediate impact of P removal (see above). The 'Keswick strategy' would also be expected to lead to a reduction in the extent and duration of de-oxygenation in the bottom water.

The weather in Cumbria is very variable and a shallow system like Bassenthwaite will respond rapidly to seasonal, weekly and even daily changes in the weather (*cf* Loch Leven, Bailey-Watts *et al* 1990). In a dry summer the reduced washout rates would allow the phytoplankton more time to capitalise on the nutrient resources *and accumulate (troublesome) biomass* - even if actual production rates were unaltered (Brook and Woodward 1956; Bailey-Watts *et al* 1990). Under drier and warmer conditions too, there would be an even greater likelihood of thermal stratification and potential for P release from the sediments and blue-green algal blooms. Calm conditions favour the rapid growth of blue-green algae. It is likely that the poor summer in 1993 played a large part in mediating against the accumulation of high algal biomass and blue-green algal formation during 1993 (not least through its effect on flushing rate). It would be very instructive to run the model using data for a year of less inclement weather, that is, with longer, warmer periods of low flushing than was experienced in 1993. In the absence of appropriate data at this time, an approximation to this situation was made by switching off the effect of flushing and algal washout from the lake (**Figure 17**). Under these circumstances, the spring algal biomass approaches 50mg m^{-3}, an increase of about 30% over the levels observed in 1993. Removal of 80% of the Keswick STW phosphorus again appears to reduce chlorophyll$_a$ levels overall by about 25% - in spite of the June maximum being somewhat higher after OP removal than before this occurs. Under drier conditions, however, the inputs of diffuse land-derived runoff P would be less than estimated for 1993, but the point sources of this nutrient would be unaffected. Hence, the Keswick STW would contribute proportionally more than the estimated 72% of the total external load of OP in 1993. The proportional reduction in chlorophyll$_a$ would be thus be correspondingly greater than the model suggests on the basis of conditions prevailing in 1993.

All of these factors suggest that P reduction at the Keswick STW would produce benefits, even during a poor summer, although they would be greater in a dry year. Nevertheless, removal of 80% of the OP from the Keswick STW is unlikely to provide a 'quick fix' and an immediate reversal of eutrophication trends in Bassenthwaite. But in the longer term, benefits would accrue giving significant reductions in algal biomass compared to the present.

9. CONCLUSIONS

The following conclusions are strictly valid for the period January to August 1993, only; they may also hold for other years but this should not be assumed:

(i) predictions of water quality, accumulated algal biomass and algal types are well represented by the model.

(ii) the recycling of ortho-phosphate from the sediments plays an important role in promoting algal growth during the summer months.

(iii) chlorophyll$_a$ levels were severely limited by hydraulic flushing from January to early May, and by phosphorus from early March to mid-August.

(iv) removal of 80% of the OP from the Keswick STW effluent would reduce peak chlorophyll$_a$ levels by about 10mg m^{-3} in spring, but appears to have less effect later on in the year.

(v) high rainfall and relatively low temperatures in 1993 probably reduced the effect of P removal on algal biomass.

(vi) resuspension of algal cells from the lake sediments may well contribute to peak phytoplankton numbers.

10. RECOMMENDATIONS

(i) The model should be re-run for a warm, dry year to assess the effect of P removal on algal biomass accumulation.

(ii) A total P balance should be carried out to assess the magnitude of sediment P releases.

(iii) An estimate should be made of the likely time it would take for a new equilibrium to be reached after P removal from the STW.

(iv) Although apparently modest, the reductions in OP and chlorophyll$_a$ achievable by removing 80% of the Keswick effluent OP, would lead to less accumulation of P-rich sediment and, thus, reduce the potential for P releases; it is therefore suggested that the Keswick option be maintained.

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12. ACKNOWLEDGEMENTS

We are especially grateful to Mr George Jaworski, Mrs Julie Parker, Mr Eric Rigg, Mrs Jose Roscoe for providing, and advising on the Bassenthwaite phytoplankton and nutrient data. Drs Colin Reynolds and Tony Irish helped us apply the model, and Miss Nicola Wiltshire is thanked for help with analysing the algal data.

13. FIGURES

Figure 1. Part of the catchment of Bassenthwaite Lake showing sampling sites 1-11 on the inflows, and the approximate location of the main sources of sewage effluent entering the lake.

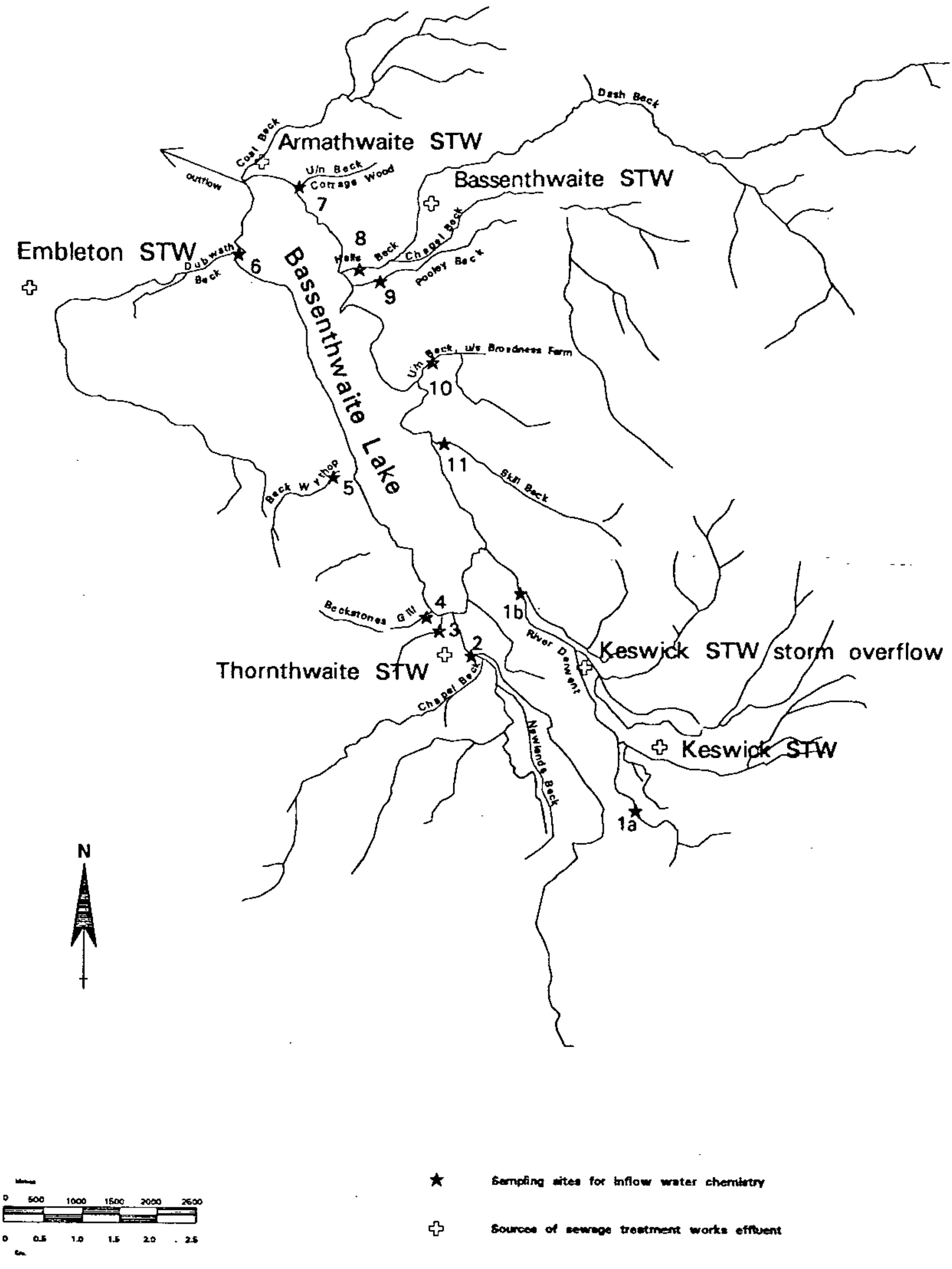


Figure 2. Example of the regressions used to estimate missing daily values for NO_3-N and SiO_2 concentrations in the inflow streams from their flows: River Derwent at Lowstock Bridge.

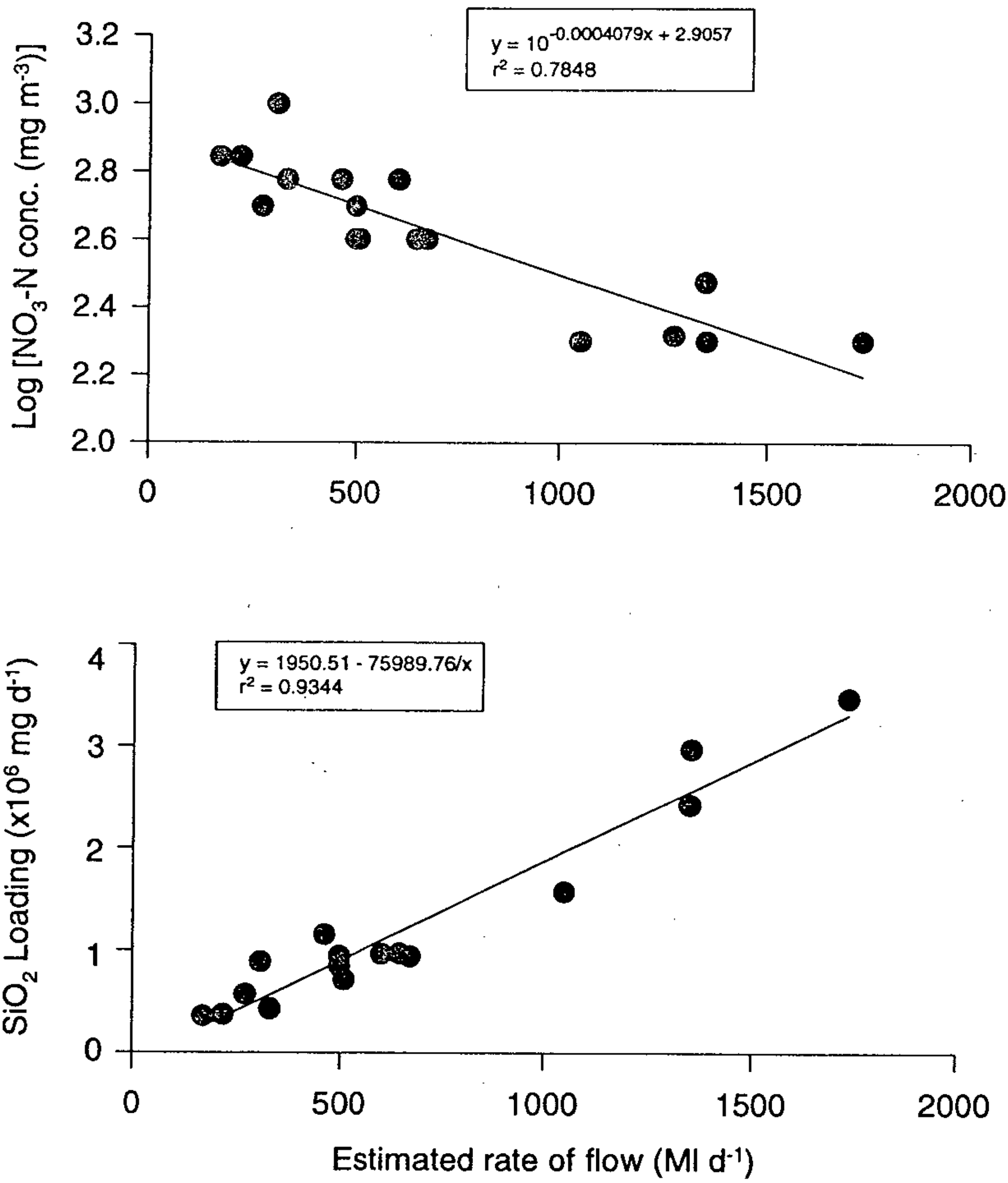


Figure 3. The relationship between rate of outflow estimated from NRA gauging data at the outflow, and rate of outflow estimated as the sum of the inflows.

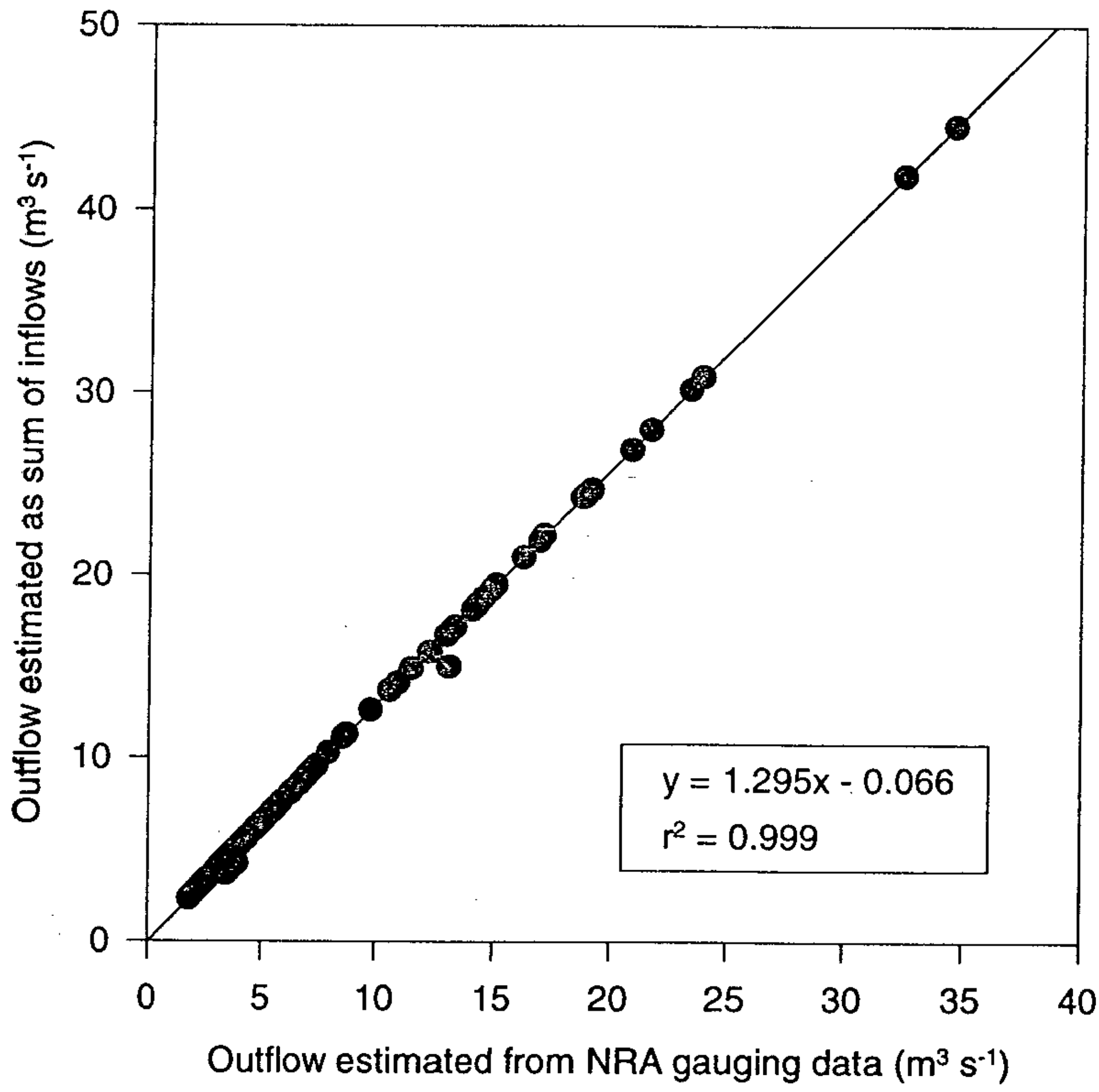


Figure 4. Total rate of inflow of water to Bassenthwaite Lake from the 11 main feeder streams.

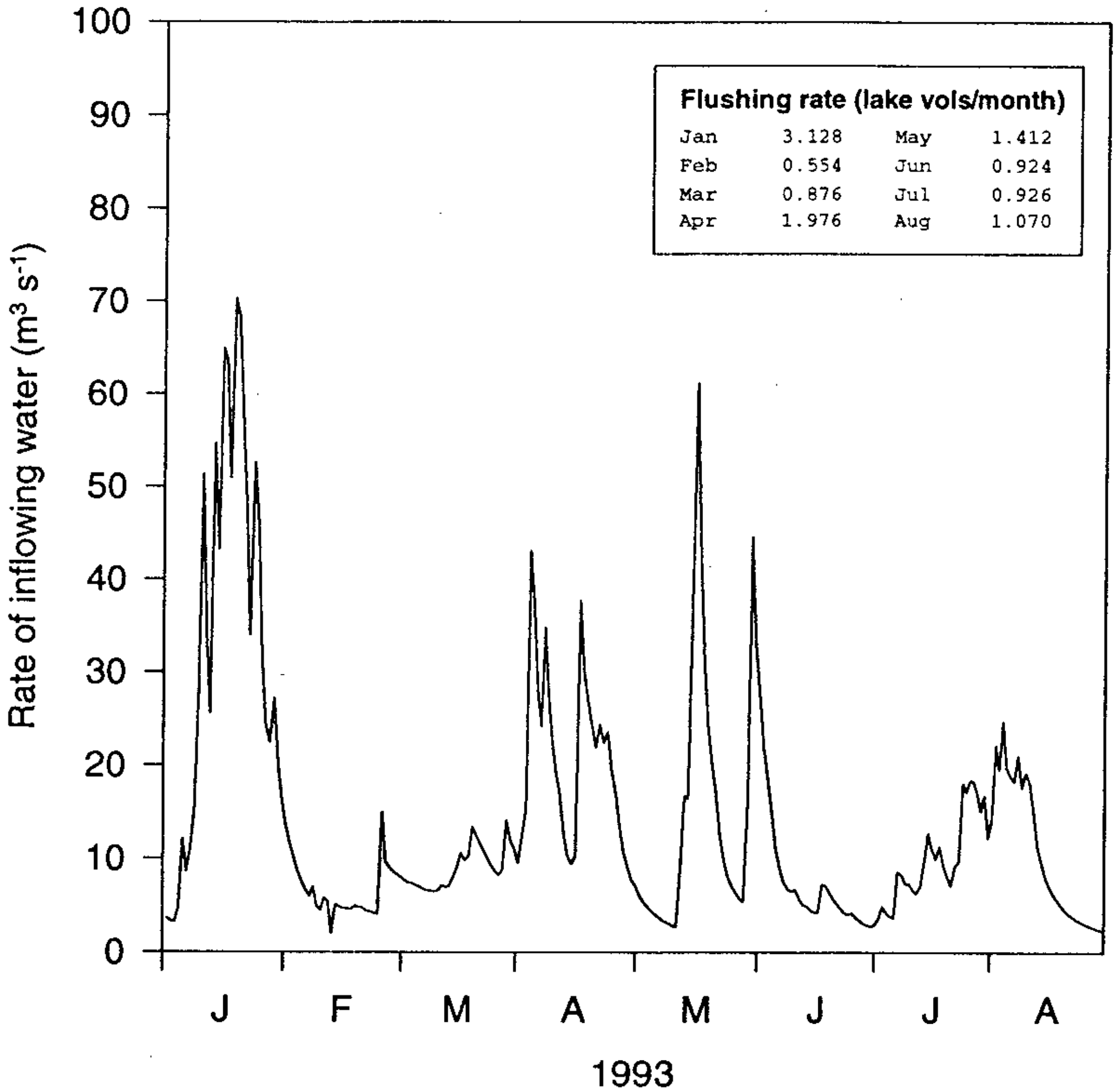


Figure 5. Measured open water temperature of Bassenthwaite Lake.

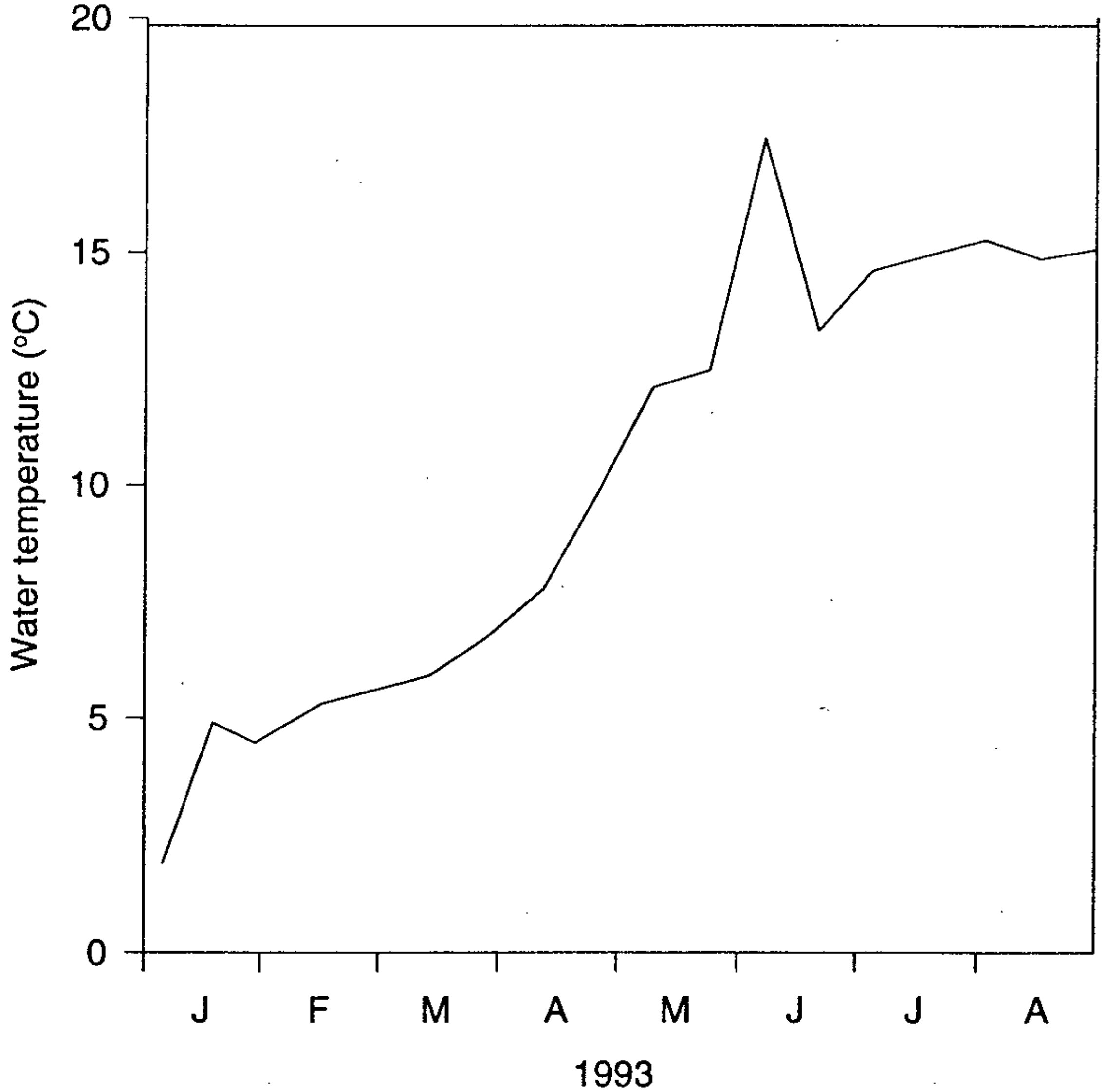


Figure 6. Measured in-lake concentrations of OP, NO_3-N and SiO_2.

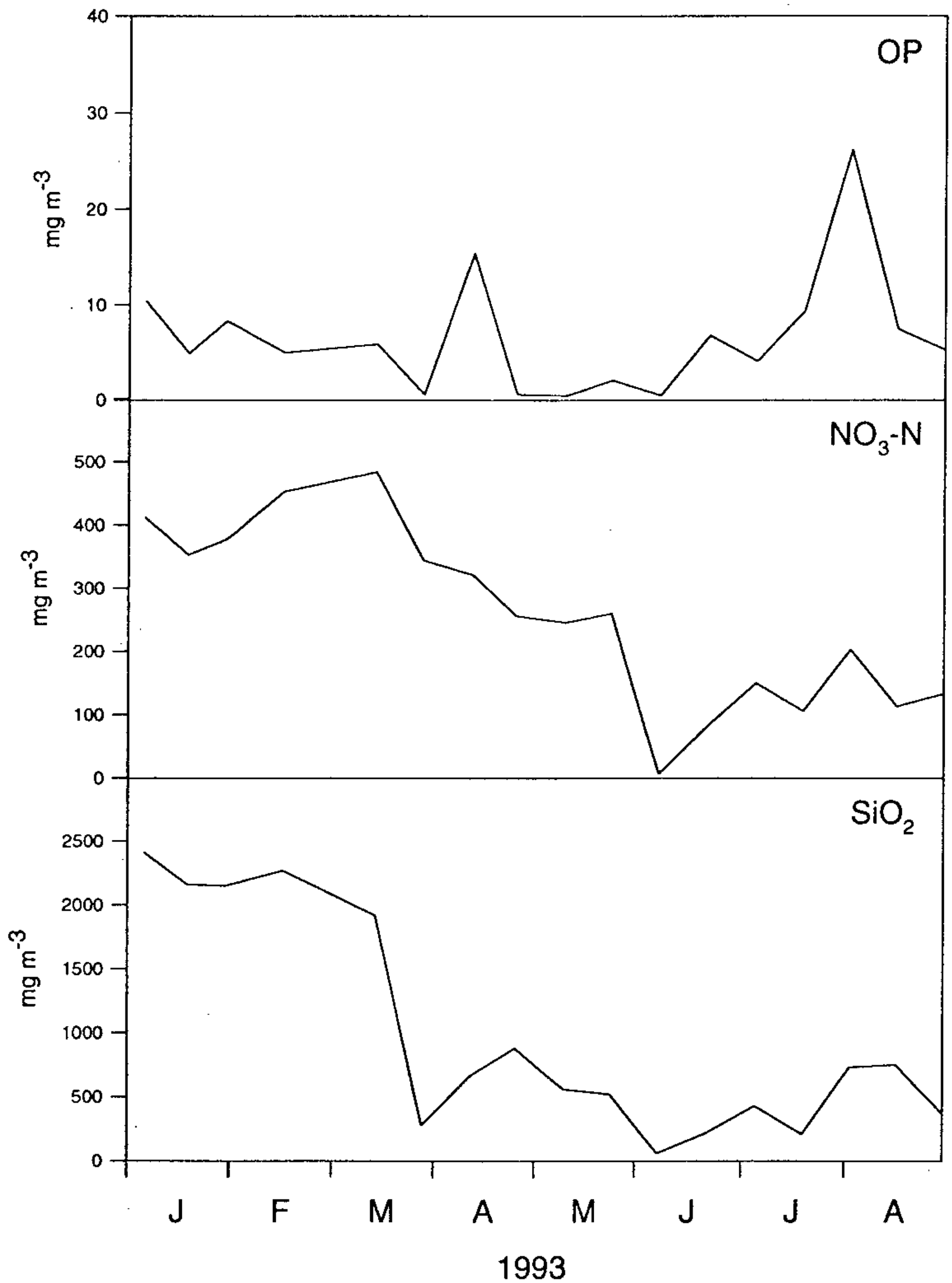


Figure 7. Measured total in-lake chlorophyll$_a$ concentration.

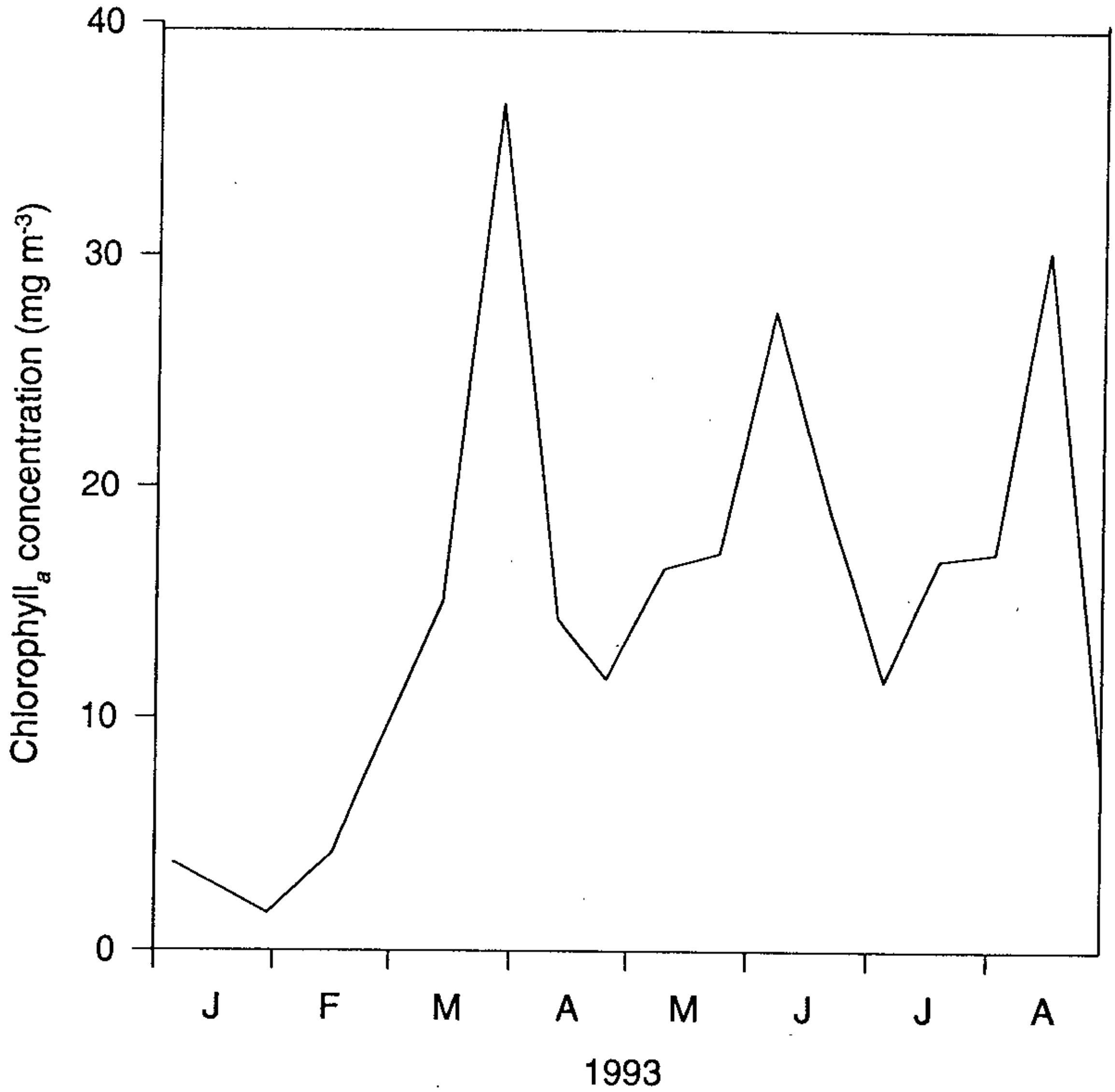


Figure 8. Measured open water chlorophyll$_a$ values for each algal group.

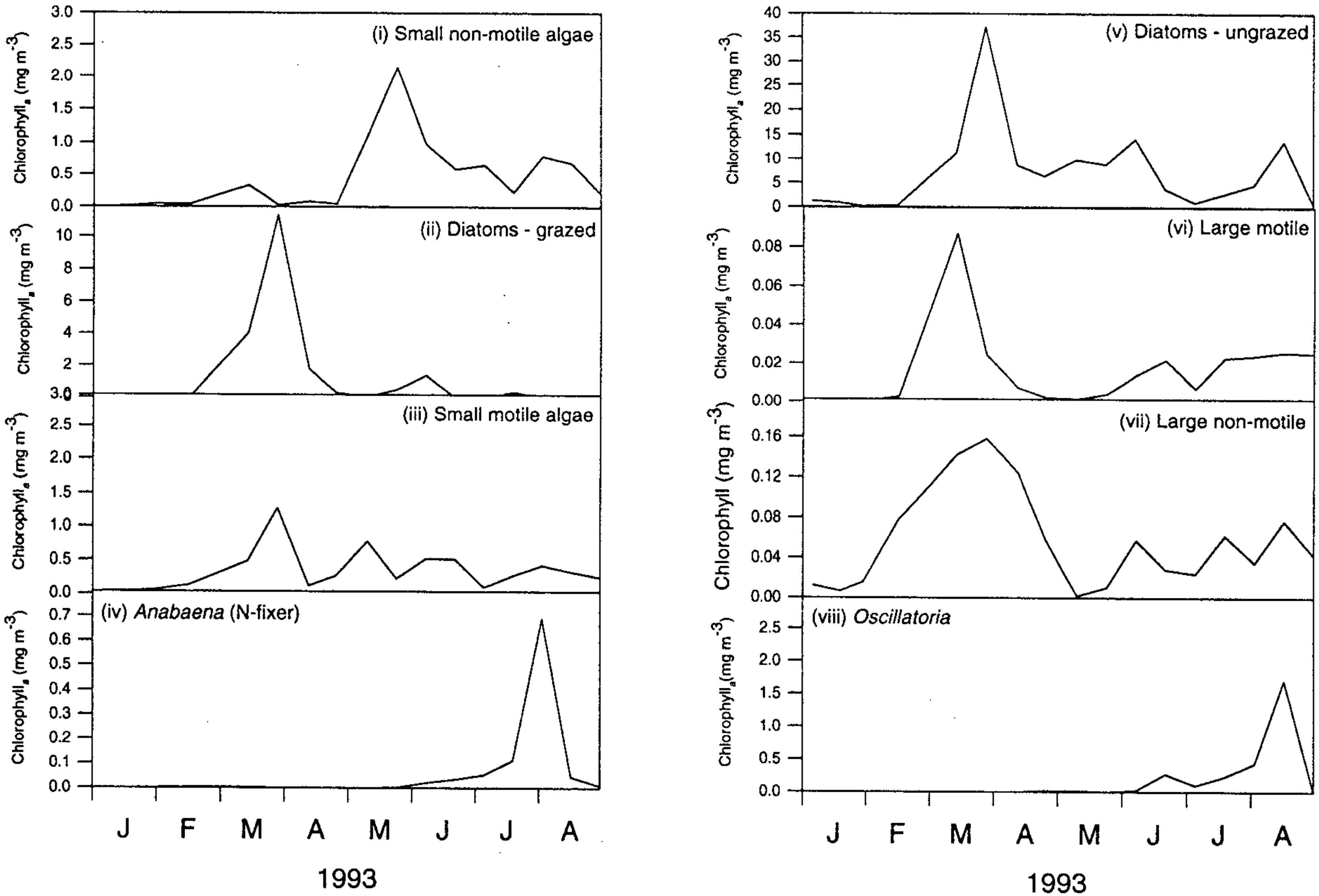


Figure 9. Initial output from the model in relation to the measured in-lake values for chlorophyll$_a$ and nutrient concentrations.

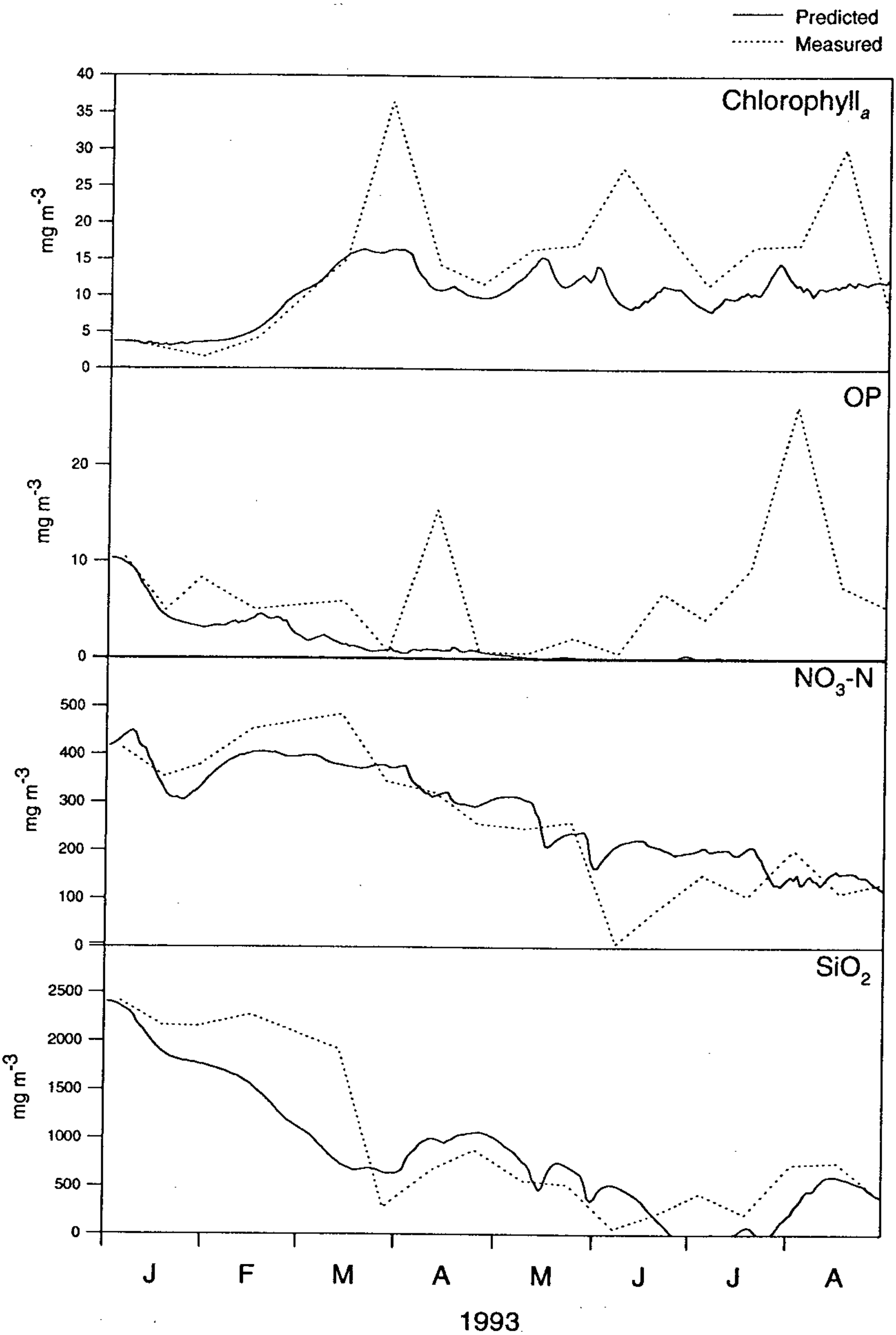


Figure 10. Measured open water chlorophyll$_a$ values for each algal group and values predicted from the initial run of the model.

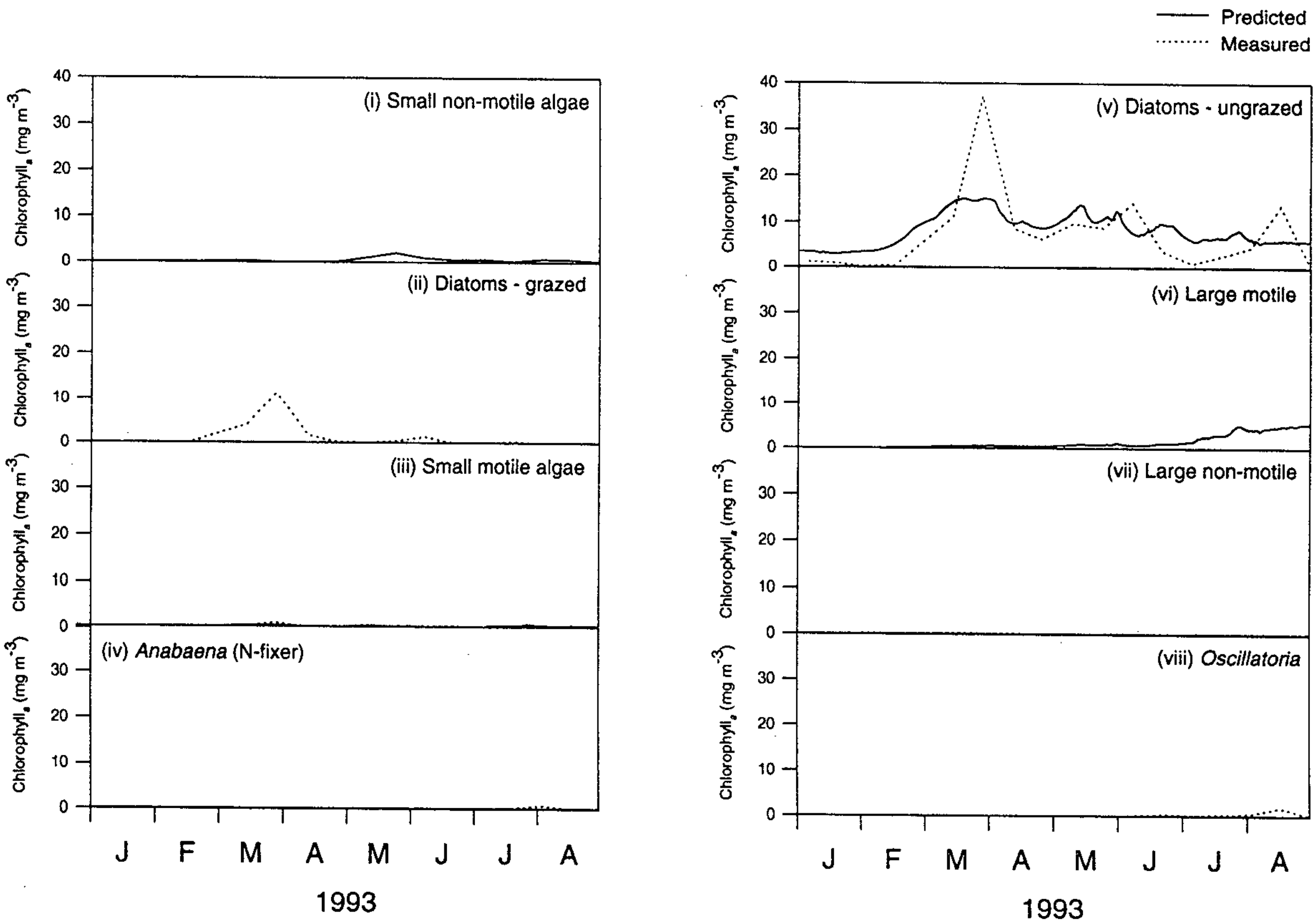


Figure 11. Comparison of predicted chlorophyll$_a$ levels with different limiting factors.

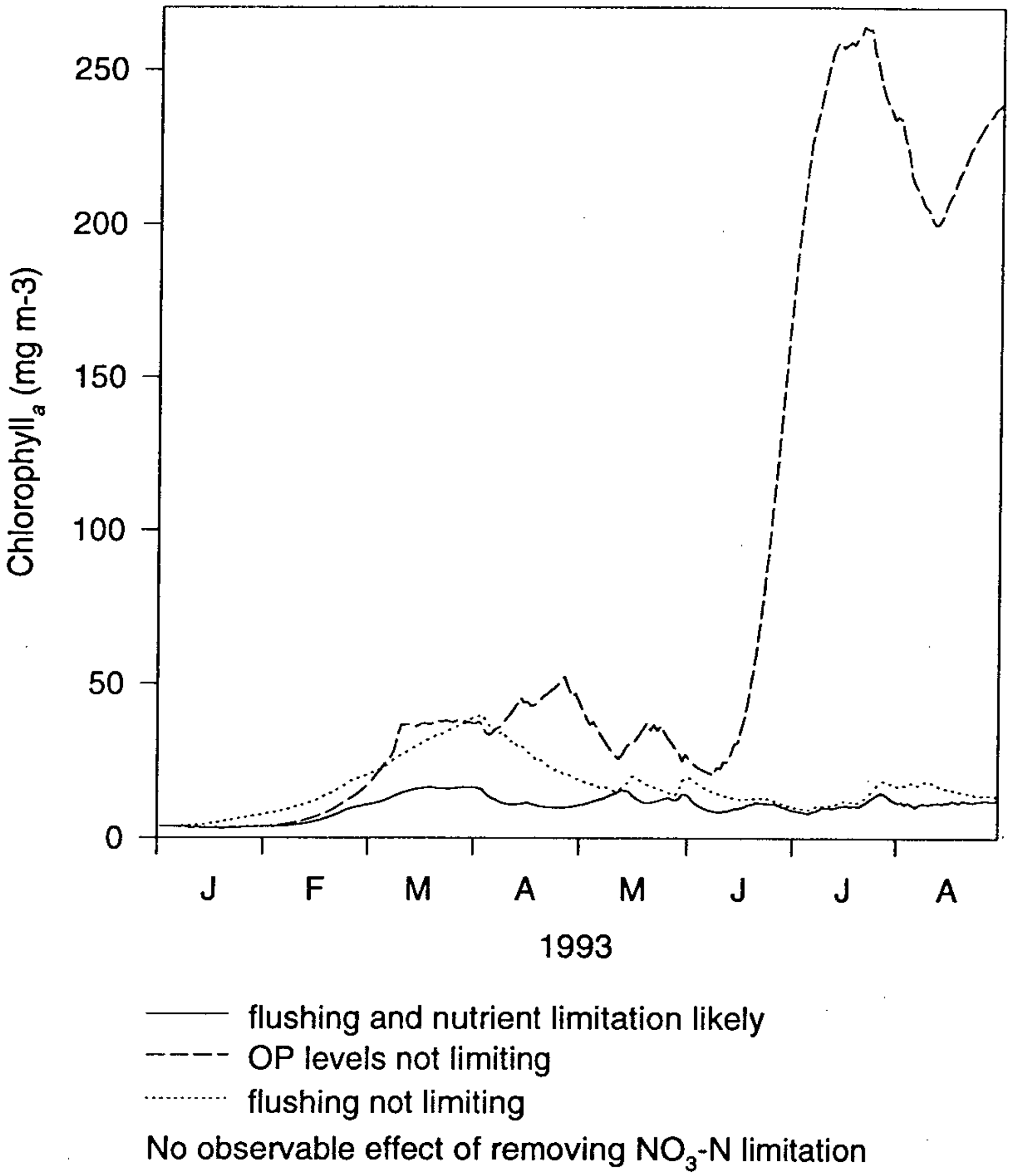


Figure 12. Temperature profile for Bassenthwaite Lake, 27 July - 10 October, 1993.

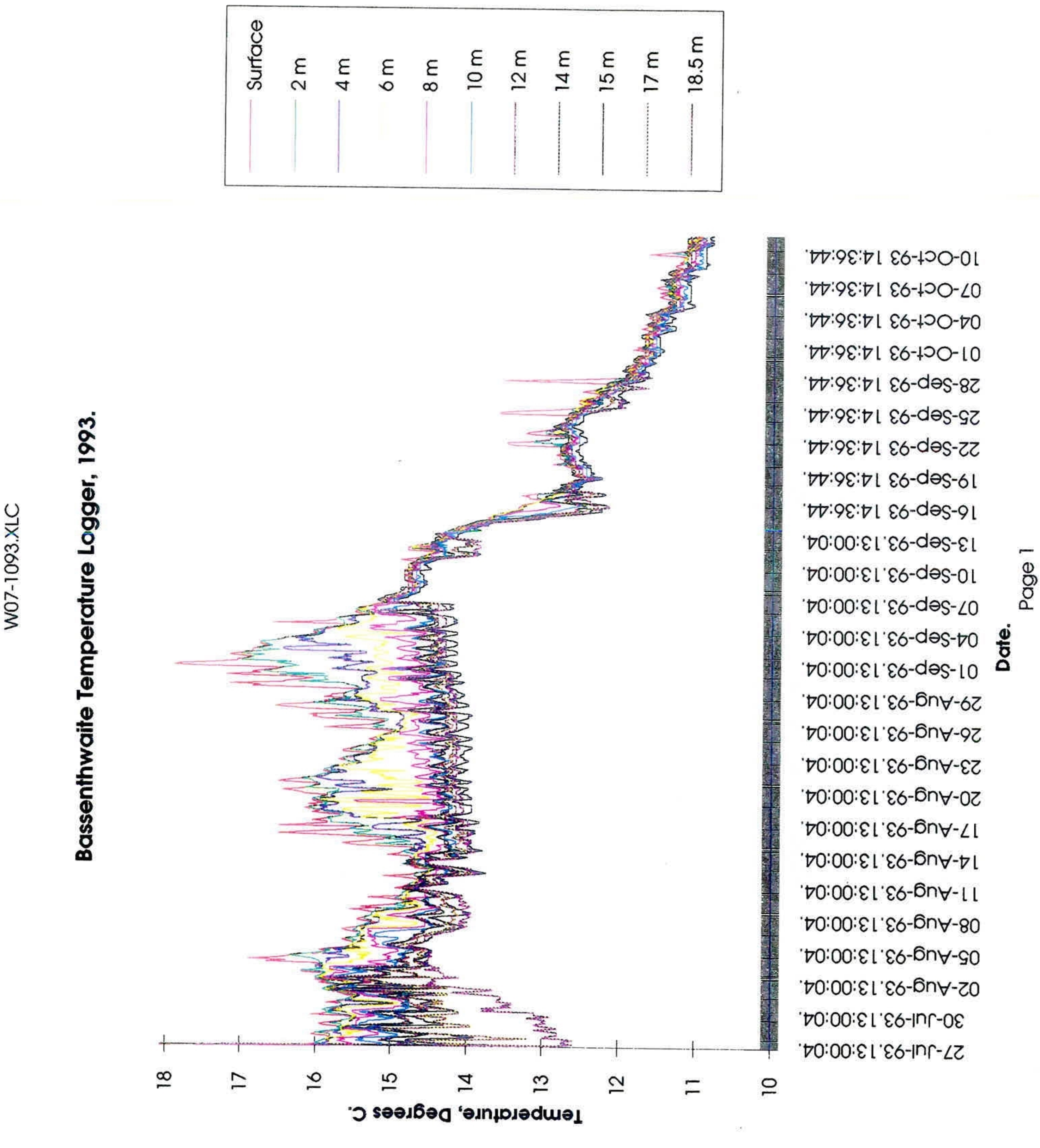


Figure 13. Predicted levels of chlorophyll$_a$ and nutrients in the lake after pulsed OP release from the sediments was included in the model. The predicted values are shown in relation to the measured values.

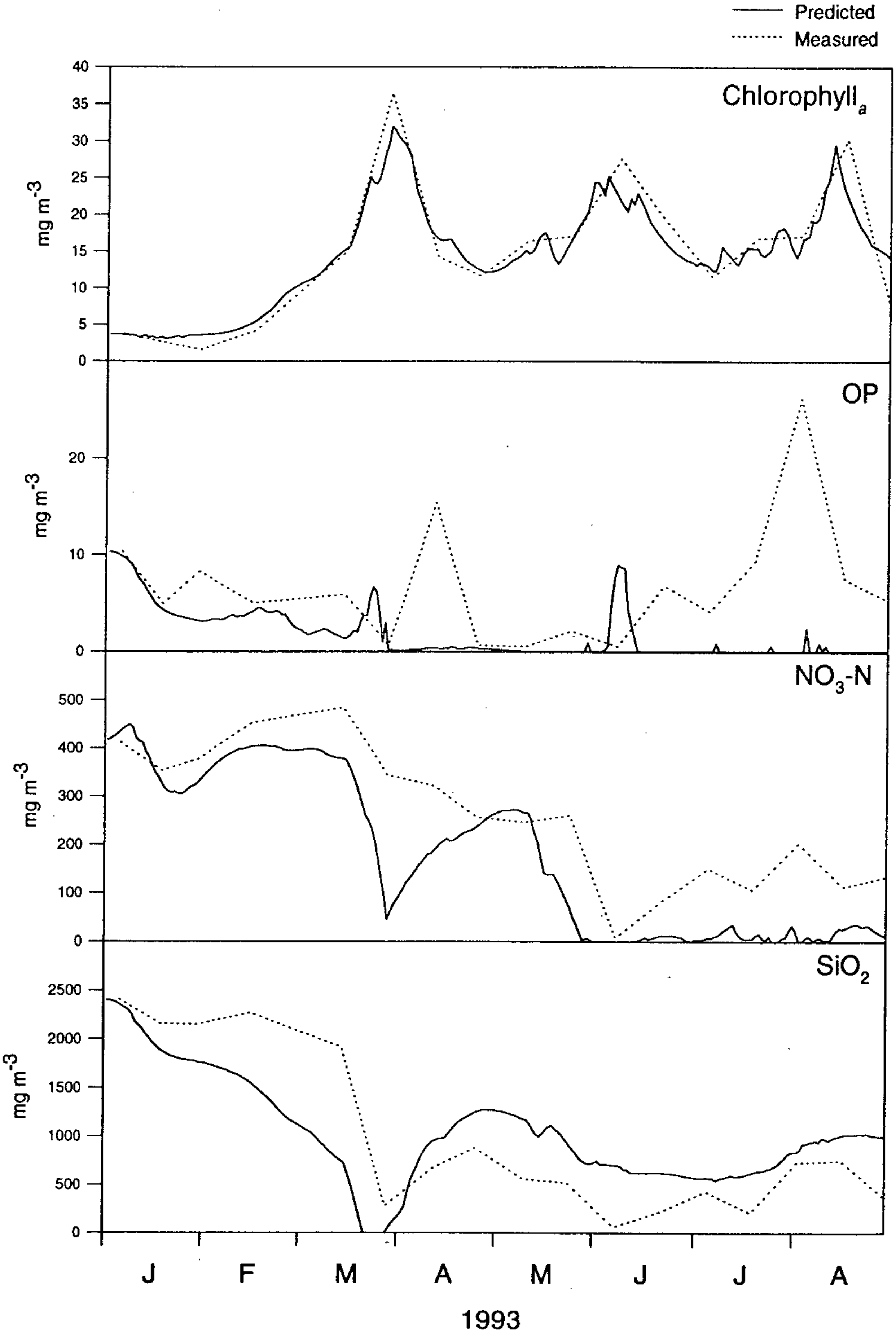


Figure 14. Measured open water chlorophyll$_a$ values for each algal group and values predicted by the model, assuming phosphorus release from the sediments.

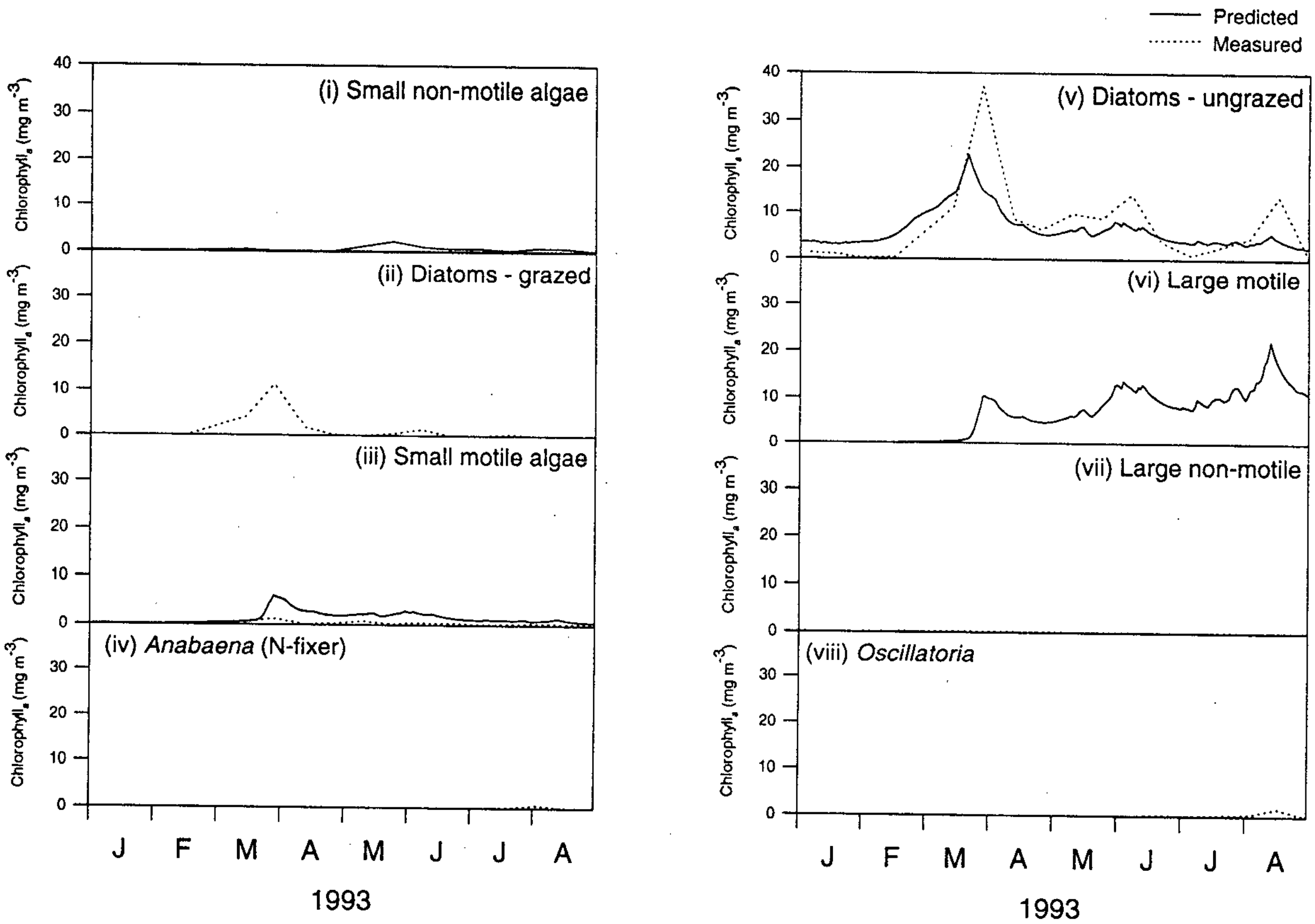


Figure 15. Comparison of predicted chlorophyll$_a$ levels, before and after removal of 80% of the OP from Keswick STW effluent.

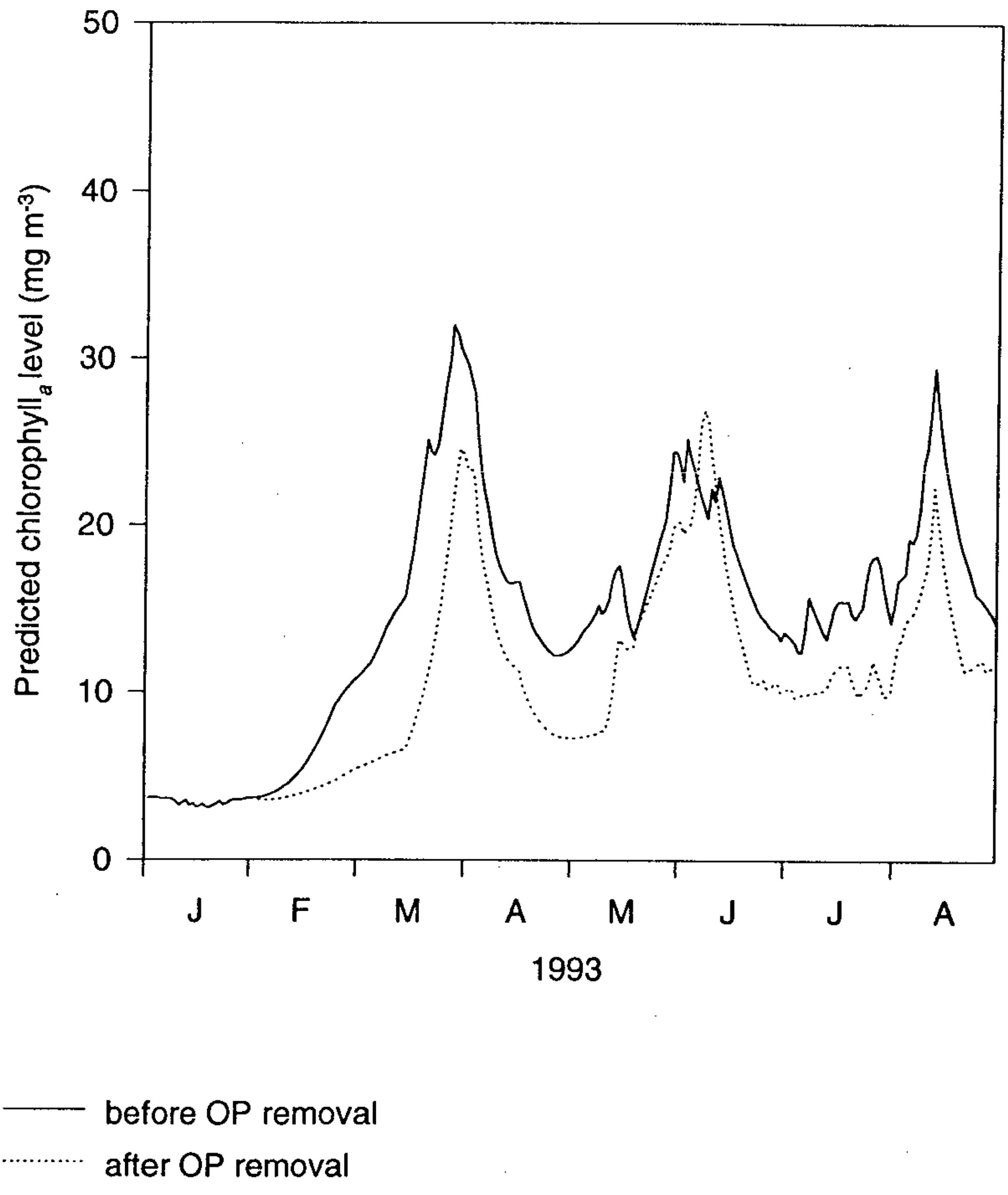
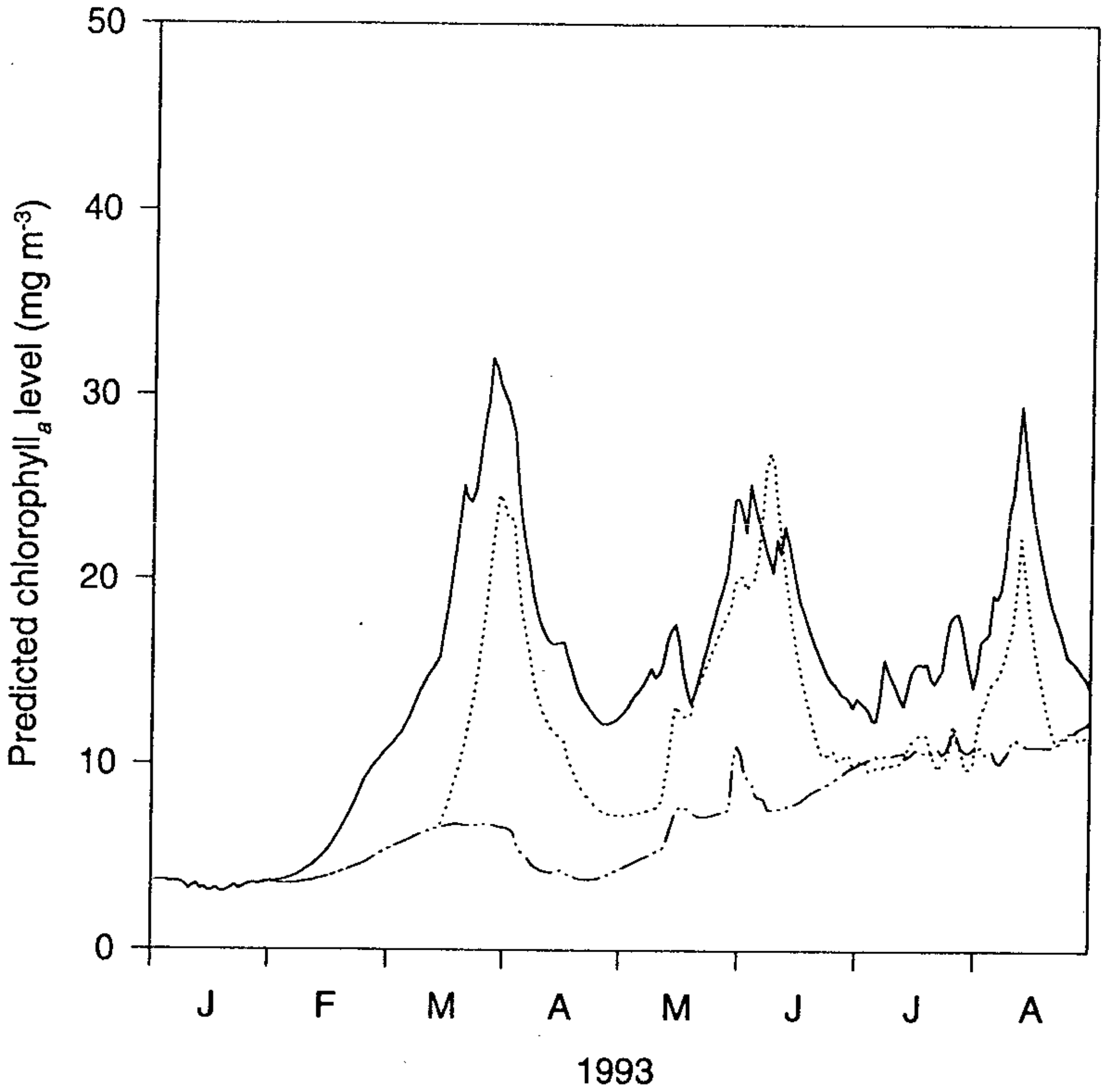


Figure 16. Comparison of predicted chlorophyll$_a$ levels, before and after removal of 80% of the OP from Keswick STW effluent, and with 80% of the OP from Keswick removed, assuming no release of P from the sediments.

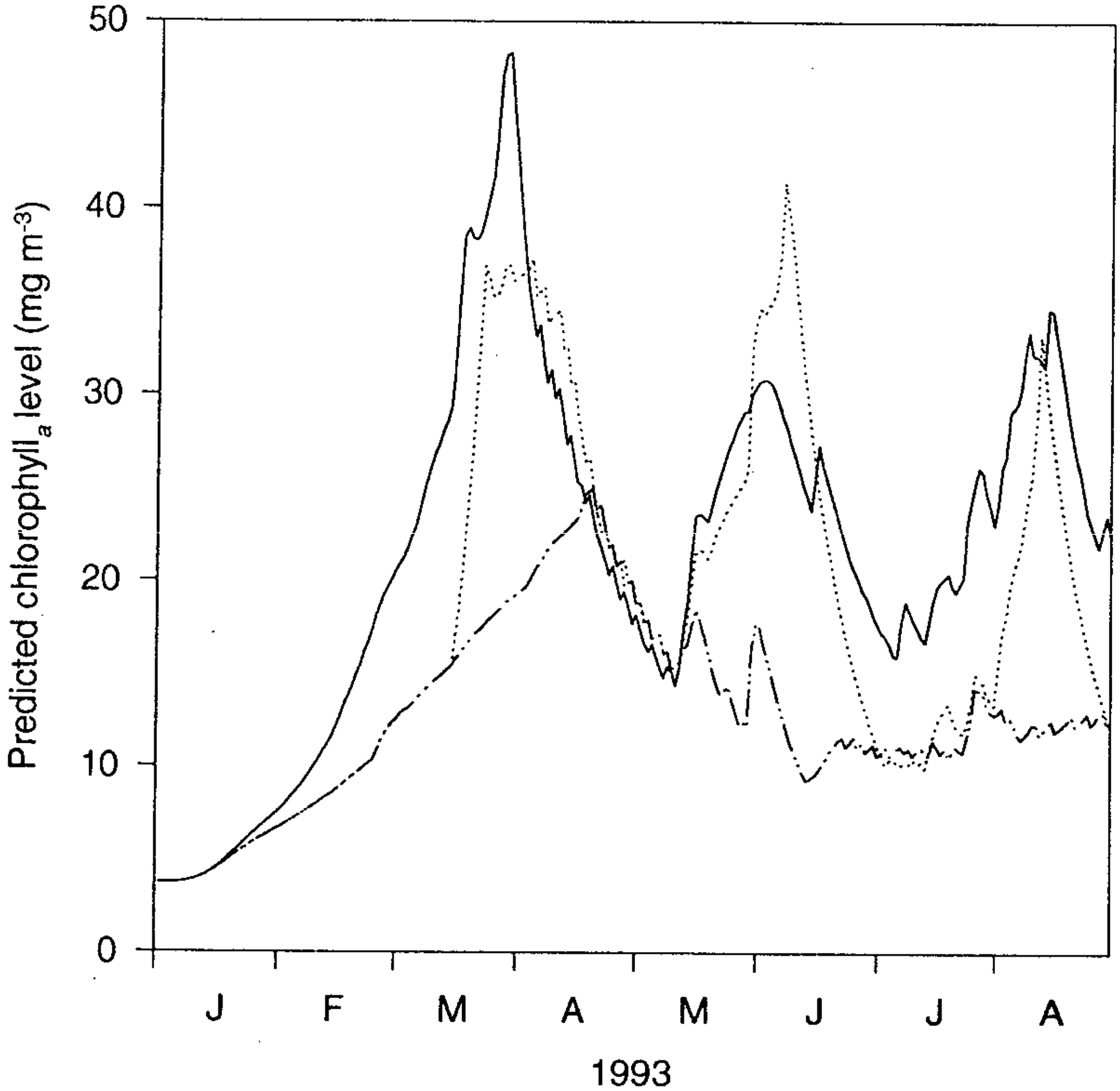


——— before OP removal, with OP release from sediments

·········· after OP removal, with OP release from sediments

—·—·— after OP removal, assuming no release of OP from sediments

Figure 17. Comparison of predicted chlorophyll$_a$ levels before and after removal of 80% of the OP from Keswick STW effluent with no flushing, and with and without OP recycling from the sediments.



——— before OP removal, no flushing, with OP recycling from sediments

········ after OP removal, no flushing, with OP recycling from sediments

—·—·— after OP removal, no flushing, without OP recycling from sediments