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Institute of Freshwater Ecology

RUTLAND WATER: AN ASSESSMENT OF THE EFFECTS OF RECENT REDUCTIONS IN PHOSPHORUS LOADS ON ALGAL GROWTH, ESPECIALLY BLUE-GREEN SPECIES, AND OF THE LIKELY EFFECTS OF PROPOSED CHANGES IN THE CURRENT PHOSPHORUS MANAGEMENT PROGRAMME.

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Principal Investigators:

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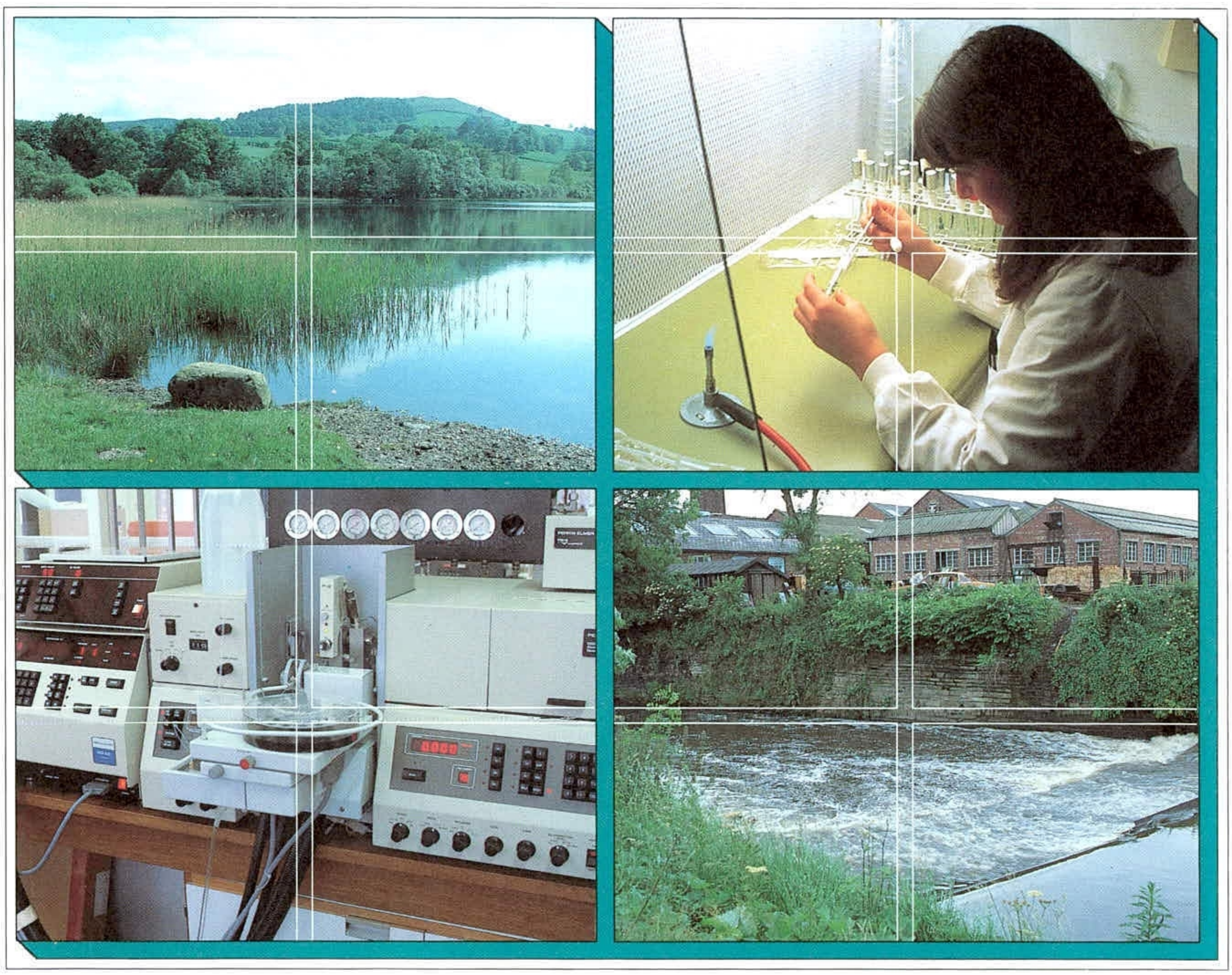
Report to Anglian Water Services Ltd (March 1994)



Natural Environment Research Council

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A report commissioned by Anglian Water Services Ltd - March 1994

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SUMMARY

Rutland Water is a large, totally man-made reservoir in Leicestershire. From a very early stage in its development, it was recognised that water quality problems were likely to occur here, due to eutrophication. In the summer of 1989, several sheep and dogs died after ingesting toxic blue-green algal scums which had formed as a result of nutrient enrichment. In response to this, Anglian Water (AW) began a programme of ferric dosing which was aimed at reducing the potential orthophosphate (OP) load from the main feeder waters, the Rivers Welland and Nene. These sources together account for 95-98% of the pre-treatment OP load to the reservoir.

As a result of the ferric dosing, mean chlorophyll$_a$ concentrations in the reservoir actually decreased slightly in 1992, compared to 1987, even though the potential OP load to the reservoir was 2.5 times higher. Using Reynold's PROTEC1 model to simulate the observed phytoplankton and nutrient dynamics for 1987 and 1992, this study showed that ferric dosing probably removed about 90% of the potential OP load from the Welland and Nene abstractions during 1992.

The use of ferric dosing as a phosphorus (P) management tool in this lake is currently under review and several alternative methods of P control have been proposed. The PROTEC1 simulation was used to evaluate these methods and four are discussed. First, it was found that, if ferric dosing were stopped altogether, chlorophyll$_a$ levels would increase enormously. Moreover, blue-green algal blooms would constitute most of the increase. Second, if ferric dosing were discontinued, but replaced by a 50% reduction in the OP load from the River Nene, a smaller but, nevertheless, significant increase in chlorophyll$_a$ levels would be recorded. Third, if ferric dosing were continued, and supplemented by a 50% reduction in the OP concentration in the River Nene, a relatively small, but sustained, reduction in chlorophyll$_a$ levels would result. Fourth, by reducing the OP levels in the Welland and Nene to AW's target concentration of 10 mg m^{-3}, chlorophyll$_a$ concentrations would be reduced to very low levels and blue-green algal blooms almost eliminated. This is likely to be, by far, the most effective way of limiting phytoplankton growth and biomass accumulation in this reservoir.

CONTENTS

LIST OF FIGURES

Figure 24. Predicted nutrient and chlorophyll$_a$ concentrations in Rutland Water during 1992 assuming a 90% reduction in orthophosphate loading from the Welland and Nene due to ferric dosing and a 50% reduction in orthophosphate in the River Nene. The results are compared to the simulation giving the best fit to the observed data for 1992.

Figure 25. Predicted abundance of each algal group (expressed as chlorophyll$_a$) in Rutland Water during 1992 assuming a 90% reduction in orthophosphate loading from the Welland and Nene due to ferric dosing and a 50% reduction in orthophosphate in the River Nene. The results are compared to the simulation giving the best fit to the observed data for 1992.

Figure 26. Predicted nutrient and chlorophyll$_a$ concentrations in Rutland Water during 1992 assuming only a 50% reduction in orthophosphate in the River Nene. The results are compared to the simulation giving the best fit to the observed data for 1992.

Figure 27. Predicted abundance of each algal group (expressed as chlorophyll$_a$) in Rutland Water during 1992 assuming only a 50% reduction in orthophosphate in the River Nene. The results are compared to the simulation giving the best fit to the observed data for 1992.

Figure 28. Predicted nutrient and chlorophyll$_a$ concentrations in Rutland Water during 1992 assuming that the orthophosphate concentrations in the Rivers Welland and Nene are reduced to 10 mg m^{-3}. The results are compared to the simulation giving the best fit to the observed data for 1992.

Figure 29. Predicted abundance of each algal group (expressed as chlorophyll$_a$) in Rutland Water during 1992 assuming that the orthophosphate concentrations in the Rivers Welland and Nene are reduced to 10 mg m^{-3}. The results are compared to the simulation giving the best fit to the observed data for 1992.

Figure 30. A comparison of the predicted chlorophyll$_a$ concentrations in Rutland Water which would have prevailed in 1992 under 5 different phosphorus management strategies.

1. INTRODUCTION

1.1 Background

1.1.1 *The reservoir*

Rutland Water (**Figure 1**), in Leicestershire, is a very large, pumped-storage reservoir which was built on a tributary of the River Welland. It has a surface area of 1.26×10^3 ha, a maximum depth of 34 m, and a maximum volume of 1.369×10^8 m^3, making it the largest man-made lake in Western Europe. It covers approximately 4% of the old county of Rutland after which it was named

The reservoir was planned in the 1960s and construction began in 1971. The dam was closed in 1975 and the reservoir was full by March 1979. The aim of the project was to satisfy the rising demand for water from the developing towns of Northampton, Peterborough, Corby, Milton Keynes, Daventry and Wellingborough. However, the lake also provides recreational facilities including a nature reserve, trout fishery and sailing club.

Initially, the reservoir began to fill from its 3 natural inflows (North Gwash, (South) Gwash, Egleton Brook) which drain its direct catchment of 7.4×10^3 ha. However, since 1976, these inflows have been supplemented by water pumped from the River Welland at Tinwell and the River Nene at Wansford. Although the water abstracted from these rivers would not have met the water quality standards recommended for direct water supply purposes (World Health Organization 1971), it was suitable for supply to a water storage reservoir.

Water is abstracted from the reservoir to the Wing water treatment works, for water supply purposes, and to the River Witham, to supplement abstraction from this river. In addition, a small amount is released into the River Gwash, downstream of the dam, as compensation flow.

1.1.2 *The eutrophication problem*

A considerable amount of pure and applied investigative science has been carried out on

Rutland Water since its completion, e.g a general overview of the first 10 years [Harper 1982a]; water movements [Maddocks 1982]; nutrients [Low 1982]; the phytoplankton [Ferguson and Harper 1982]; algal sedimentation [Buranathanitt, Cockrell and John 1982]; benthic invertebrates [Bullock, Clarke and Ison 1982; Brown and Oldham, 1982]; zooplankton [Harper and Ferguson 1982]; fish [Harper 1982b]. From a very early stage, it was recognised that water quality problems were likely to occur within the reservoir. Like many lowland lakes and reservoirs, Rutland Water was receiving substantial inputs of plant nutrients (phosphates and nitrates) from treated sewage effluent and agricultural runoff. As the reservoir was also relatively shallow (mean depth 10.7 m), these inputs were expected to result in very high levels of algal production. To make matters worse, the reservoir also tended to stratify in summer. Hence, it was considered to be likely that the combination of over-enrichment and stratification would manifest itself as dense surface blooms of large blue-green algae during calm weather.

The problem of stratification was the first to be addressed. Twelve 'Helixor' air lift pumps (vertical stack pipes into which air is introduced from a compressor located at the Empingham pumping station) were installed into the main body of the reservoir, close to the limnological tower [Knights 1982]. The original plan was to switch on these pumps once stratification had developed so that the water column would become mixed and de-oxygenation of the deeper waters would be prevented. However, in 1978, the 'Helixors' were shown to be too under-powered to overturn a full reservoir if it had already stratified [Low 1982]. As a consequence, in 1979 and 1980, the 'Helixors' were turned on as soon as a small temperature difference had been recorded between the surface and bottom water, i.e. in May, and run almost continuously until the end of August. This strategy of prevention rather than cure was found to be much more effective in reducing stratification [Low 1982]. In more recent years, the period of operation has been further extended, and the helixors are now operated from March to October each year [Daldorph *pers. comm.*]. Even so, thermal layering still develops occasionally during very calm weather.

In spite of the pumps, blue-green algal blooms developed in Rutland water and continued to cause concern for several reasons; firstly, many species were large and could block filters and cause taste and odour problems, thus increasing water treatment costs; secondly, many of these algae could exude toxic substances; and thirdly, the unsightly scums and large particles in the water could affect the recreational use of this waterbody. Eventually, during the dry summer of 1989, a toxic bloom of these algae (mostly *Microcystis*) led to the deaths of several sheep and dogs [NRA 1990]. As a result, a decision was made to reduce phosphate levels within the reservoir to limit algal growth. The options were limited, especially in the short-term. For example, it would have been difficult to significantly reduce the inputs of phosphorus [P] from diffuse sources, such as agricultural runoff, and most of the P-rich sewage effluent from settlements around Rutland Water had already been fed into a perimeter sewerage system which discharged downstream of the reservoir. Moreover, effluent from Oakham sewage treatment works [STW], the only sewage works which had originally discharged directly into the lake, had been diverted to a land treatment area (grass plots), prior to its discharge into the lake, since July 1977. In any case, most of the P entering the system was coming from water abstracted from the Rivers Nene and Welland which continued to bring in a considerable amount of treated sewage effluent from outside the natural catchment area of the reservoir.

In 1990, Anglian Water Services [AW] began an extensive programme of ferric dosing with the aim of precipitating P and reducing concentrations in the lake water. During the first year, $FeSO_4$ was added either directly to the reservoir itself, by barge dosing, or *via* the inlet for water abstracted from the Welland and Nene. Since December 1990, only the latter method has been used but although this has continued on a regular basis, its effects on P levels and on algal growth in the reservoir have never been quantified. Early in 1992, AW approached the Institute of Freshwater Ecology [IFE] with a view to using Reynolds' PROTEC1 model to determine whether, and to what extent, ferric dosing had reduced P concentrations and algal abundance in Rutland water. This work is the outcome of those discussions.In addition, this study also considers the likely effects of changing the current P management strategy if the consent for ferric dosing is withdrawn.

1.2 Aims of the study

The main aims of this project are as follows:

(i) to assess the ability of the Reynolds' algal growth/reservoir management model (PROTEC1) to simulate the dynamics of the main phytoplankton species in Rutland Water.

(ii) to assess the impact of recent reductions in P load on algal growth, especially that of bloom-forming blue-green species, and to estimate the level of P reduction which seems to have been achieved.

(iii) to predict the likely effect of a proposed 50% reduction in the P loading from the River Nene due to P removal from the major STW, both in addition to, and instead of, ferric dosing.

(iv) to predict the likely effect of reducing the OP concentrations in the Rivers Welland and Nene to target values of 10 mg m^{-3} and discontinuing ferric dosing.

1.3 General approach adopted

Data supplied by AW, on the concentrations of dissolved ortho-phosphate [OP] (= soluble, molybdate reactive P, SRP), nitrate-nitrogen [NO_3-N] and silicate-silica [SiO_2], and the rates of flow for the feeder waters of Rutland Water, were used as inputs to Reynolds' PROTEC1 model, version 1.02. Observed changes in nutrients and phytoplankton in the reservoir were, first, simulated for one of the years prior to ferric dosing. Then, the exercise was repeated for one of the years since ferric dosing began. From this it was possible to estimate the likely reduction in OP load which had been achieved by the ferric dosing programme. In addition, by varying the theoretical OP load still further, it was possible to predict the likely outcome of changing the current P management strategy if the consent for ferric dosing is withdrawn.

2. 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] describes 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 water and sewage discharge. The model then calculates the OP, NO_3-N and SiO_2 concentrations which would occur 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 Rutland Water), mixed and mean depths. Assuming no limitation by nutrients, 'inocula' of up to 8 algal species/species groups, characteristic of the 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.

The algae are allowed to grow at these maximum rates 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.

It is assumed that zooplankton can only eat small algae and that during this process OP and NO_3-N are released instantly into the dissolved nutrient pools. Within the model, nitrogen-fixing cyanobacteria normally utilise nitrogen, but at low nitrogen levels, their growth is not limited. 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.

The model is not designed to predict the biomass of different algal species observed in the system exactly. It predicts the maximum amount of algae which would grow in the reservoir (the capacity) given limitations bounded by the supply of nutrients, hydraulic flushing, light and heat inputs. In a lake, the algal biomass may not reach these levels due to other, unmodelled limitations such as viral infections, thermal instability, algal settling etc. It is not, however, possible for the lake to sustain higher levels than the predicted capacity if the limitations included within the model are correctly incorporated.

3. THE DATABASE

All of the original data for input to the model and for comparison with the simulations was supplied by AW. Having reviewed their data holdings in relation to our requirements, AW concluded that the most appropriate 'before treatment' dataset was that held for 1987, and the best 'after treatment' dataset was that for 1992.

In general, the following data were used as inputs to the model:

a) flow rates for all feeder waters, the outflow and water draw-off points;
b) inflow concentrations of OP, NO_3-N and SiO_2
c) open water temperature for the reservoir

In addition, the data summarised below were used to validate each simulation, where appropriate:

a) concentrations of OP, NO_3-N, and SiO_2 in the reservoir
b) total phytoplankton abundance (as chlorophyll$_a$) and the population densities of the major algal species/groups in the reservoir

Ideally, daily values for all parameters are required for input to the model and validation of the output. In practice, weekly or fortnightly values, from which reliable daily estimates can be made, are usually sufficient.

4. INVESTIGATIVE METHODS, DATA HANDLING AND ANALYSIS

Data were provided by AW for the years 1987 (before ferric dosing) and 1992 (after ferric dosing). The data for each year are dealt with separately.

4.1 The data for 1987

In general, the information for 1987 fell well short of a practical dataset for input to the model and validation of the output. The problems encountered with this dataset, and our attempts to circumvent them, are detailed below.

4.1.1 *Hydraulic data*

a) flow rates of the feeder streams

Daily rates of flow for the South Gwash and Egleton Brook feeder streams were provided. Corresponding rates of flow for the North Gwash were estimated as 20% that of the South Gwash on the basis of the relative sizes of the catchment areas of these two streams.

b) water pumping data

Monthly estimates of the total amount of water abstracted from the Rivers Welland and Nene, and that pumped out to the Wing treatment works, River Gwash compensation flow and the River Witham abstraction were supplied. These figures were converted to the required daily values by dividing the monthly totals by the number of days in each month and converting from Megalitres per day ($Ml\ d^{-1}$) to cubic metres per second ($m^3\ s^{-1}$).

c) reservoir volume

A volume-depth table was provided which allowed us to estimate reservoir volume from the measured heights above ordnance data. Volume of water in storage was available at monthly intervals for the period January to May, and July, and at daily intervals for the remaining months. Initial reservoir volume on the 1 January 1987 was estimated as the average of the monthly figures for December 1986 and January 1987. The maximum reservoir volume was 1.369×10^8 m^3.

4.1.2 *Water chemistry*

a) natural inflows

OP, NO_3-N and SiO_2 concentrations, measured at 2-monthly intervals, were available for the South Gwash and North Gwash. Daily values were calculated from these data by linear interpolation between the measured values, and by constant extrapolation for periods beyond the final measured value. As no data were available for Egleton Brook, those for the South Gwash were used for this stream. The inflow of nutrients from the Oakham STW effluent was very small and has been ignored for present purposes.

b) pumped inflows

OP and NO_3-N concentrations for the Welland and Nene were available at more or less weekly intervals throughout 1987. The daily values required by the model were calculated by interpolation and extrapolation, as described above. However, no SiO_2 was available for this period. Initially, the 1992 data was used as a substitute dataset. However, the open water SiO_2 levels predicted using this dataset were very low compared to the measured values for the period May, onwards (**Figure 2**, upper panel). As the data for the 1992 simulations fitted the 1992 measured SiO_2 levels fairly closely (**Figure 2**, lower panel), it was assumed that this would also have been the case if the correct data had been available for 1987. So, the 1992 dataset was adjusted to fit the 1987 measured values more closely (**Figure 2**, upper panel) by increasing the SiO_2 loading by a factor of 10 between 10 April and 18 August.

c) the lake

Lake water chemistry data (OP, NO_3-N and SiO_2) was supplied for open water sites S13, N1 and LT (**Figure 1**). The data had been collected at approximately fortnightly intervals. There was very little variation between sites (**Figure 3**), so LT (Limnological Tower) was taken as a representative site for the reservoir.

The initial in-lake nutrient concentrations required by the model for 1 January were calculated by linear interpolation between the values recorded immediately before and after that date.

4.1.3 *Water temperature*

Water temperature readings were supplied for the Limnological Tower site, only. Most of the data was recorded weekly or fortnightly, but there was no data at all for the period 25 March to 28 July. Data collected from nearby Grafham Water was used for this missing period.

Some depth profiles of water temperature were available for July to December. This showed no evidence of stratification so any stratification which did occur during 1987 must have been temporary.

4.1.4 *Chlorophyll$_a$ values*

Chlorophyll$_a$ measurements were also provided for sites S13, N1 and LT. Samples had been collected at 2-3 week intervals and, on most occasions, the values given for all three sites were similar (**Figure 3**). Again, the Limnological Tower data was used as a representative site for validating the output from the simulation. The initial chlorophyll$_a$ value for 1 January was calculated by interpolation between the measured values immediately before and after that date.

4.1.5 *Phytoplankton*

Phytoplankton counts from fornightly samples were supplied for sites LT, N1 and S13. Only those for site LT were used in this study. Firstly, rare and occasional species (**Table 1**) were eliminated from the dataset and the remaining species were each allocated to one of the eight algal groups recognised by the model (**Table 1**). Then, each algal group was assigned three

Table 1. A list of algal species found in Rutland Water, 1987 and 1992, and the corresponding groups to which they were assigned for the model simulations. AW's species codes for each alga are given to the left of the name, estimated cell volumes (μm^3) are shown to the right.

(i) small algae not using silica ("*Chlorella*")

9354	*Cryptomonas* sp.	2000
9355	*Rhodomonas* spp.	150
9439	*Ankistrodesmus* spp.	150
9420	*Chlamydomonas* spp.	150
9421	*Chlamydomonas* type	150
9441	*Ankistrodesmus falcatus*	250
9451	*Chlorella* type	30
9651	Unicellular flagellates <10μm	200
9652	Unicellular green algae	30
9672	Flagellates misc.	150

(ii) small centric diatoms ("*Stephanodiscus*")

9382	Centric diatoms <10μm	800
9387	*Stephanodiscus* spp.	1000
9388	*Stephanodiscus astraea*	20000
9390	*Stephanodiscus astraea* (not var *minuta)*	10000
9391	*Stephanodiscus hantzschii*	2500
9665	Centric diatoms <5μm	200

(iii) large algae excluding diatoms and blue greens ("*Closterium*")

9462	*Coelastrum* spp.	3000
9463	*Coelastrum microsporum*	3000
9464	*Coelastrum microsporum*	3000
9482	*Oocystis* sp.	200

(iv) all nitrogen-fixers ("*Anabaena*")

9300	*Anabaena* spp.	100
9320	*Aphanizomenon* spp.	130

(v) large, ribbon-shaped diatom ("*Fragilaria*")

9386	*Melosira* spp.	6340
9394	Pennate diatoms general	2000
9408	*Navicula* sp.	600
9409	*Nitzschia* spp.	2500

(vi) large, ungrazed, compact diatoms ("*Asterionella*")

9397	*Asterionella formosa*	500

(vii) large, compact, non-N fixing, blue-green alga ("*Microcystis*")

9338	*Microcystis* spp.	3

(viii) large, ribbon-shaped, non-N fixing blue-green algae ("*Oscillatoria*")

9342	*Oscillatoria* spp.	100

Rare and occasional species not included above

9324 *Chroococcus*
9325 *Coelosphaerium* spp.
9331 *Gomphosphaeria* spp.
9352 *Phacus* spp.
9353 *Trachelomonas* spp.
9357 *Ceratium* spp.
9360 *Gymnodinium* spp.
9375 *Uroglena* spp.
9423 *Eudorina* spp.
9428,9429 *Pandorina* spp.
9435 *Actinastrum* spp.
9445 *Ankyra* spp.
9446 *Ankyra judayi*
9453 Colonial cells round
9457 Coccoid <5μm
9461 *Closteriopsis* sp.
9484 *Pediastrum* spp.
9485 *Pediastrum* sp.
9489 *Pediastrum duplex*
9495 *Scenedesmus* spp.
9505, 9506 *Scenedesmus quadricauda*
9507 *Schroederia* sp.
9509 *Sphaerocystis* spp.
9510 *Tetraedron* spp.
9520 *Cosmarium* spp.
9525 *Staurastrum* spp.
9676 *Chroomonas* spp.
9679 *Elakothrix* sp.
9680 *Rhoicosphenia* sp.

morphological values for maximum dimension, volume and surface area, and classified according to whether it is grazed, a diatom or a nitrogen fixer (**Table 2**). The biomass of each algal group, in terms of chlorophyll$_a$ concentration, was then calculated from the algal counts. This involved calculating the biovolume (e.g. mm^3 l^{-1}) of each alga from estimated cell volumes (**Table 1**) and summing these to give a value for each of the eight groups. Finally, these values were summed to give a total chlorophyll$_a$ value for each date, as estimated from the algal counts, which could be compared to the measured values.

4.2 The data for 1992

The data for 1992 were more detailed than for 1987, and far better suited for input to the model and validation of the output.

4.2.1 *Hydraulic data*

a) flow rates of the feeder streams

Daily rates of flow were available for the South Gwash and Egleton Brook. Daily rates of flow for the North Gwash were estimated as 20% that of the South Gwash, based on the catchment area ratios of these streams.

b) water pumping data

Daily values were provided for the volumes of water pumped into and out of the reservoir, i.e. *from* the Welland and Nene and *to* the Wing Works, River Witham and River Gwash.

c) reservoir volume

Readings of reservoir volume, taken at 3 daily intervals, were provided.

Table 2. Sizes and other key descriptors representative of the 8 algal groups described in the text.

Algal group	volume (μm^3)	maximum dimension (μm)	surface area (μm^2)	diatom?	grazed?	N-fixer?
(i) Chlorella	180	7	154	N	Y	N
(ii) Stephanodiscus	393	10	314	Y	Y	N
(iii) Closterium	4520	360	4550	N	Y	N
(iv) Anabaena	29000	75	6200	N	N	Y
(v) Fragilaria	18690	160	27570	Y	Y	N
(vi) Asterionella	5160	130	6690	Y	Y	N
(vii) Microcystis	4200000	200	126000	N	N	N
(viii) Oscillatoria	46600	1000	24300	N	Y	N

4.2.2 *Water chemistry*

a) natural inflows

Nutrient concentrations (OP, NO_3-N and SiO_2), measured at weekly intervals, were provided for the South Gwash and North Gwash. No data were available for Egleton Brook, so nutrient concentrations for the nearby South Gwash were used as a substitute for this missing dataset. These data were converted to daily values by linear interpolation between the measured values and extrapolation, at a constant value, beyond the last measured value.

b) pumped inflows

Weekly measurements of nutrient concentrations (OP, NO_3-N and SiO_2) in the water abstracted from the Rivers Welland and Nene, prior to ferric dosing, were provided. These data were converted to daily values by linear interpolation between the measured values and extrapolation, at a constant value, beyond the last measured value.

c) the lake

Although lake nutrient concentrations (OP, NO_3-N and SiO_2) were given for 3 sites (N1, S13 and LT - see **Figure** 1), only those for site LT were used in the simulations. Daily values were estimated from these weekly or fortnightly measurements by linear interpolation between the measured values and extrapolation, at a constant value, beyond the last measured value. The measured values for 31 December 1991 were used as the initial values for the model.

4.2.3 *Water temperature*

Weekly and fortnightly measurements of open water temperature were supplied for the limnological tower site (LT), only. These values were converted to daily values by linear interpolation between the measured values and extrapolation, at a constant value, beyond the last measured value.

4.2.4 *Chlorophyll$_a$ values*

Open water chlorophyll$_a$ values for were provided for sites N1, S13 and LT (**Figure 1**). The values had been recorded at monthly intervals from January to April, and from November to December, and at weekly intervals from May to October.

4.2.5 *Phytoplankton*

Algal counts were provided for open water sites N1, S13 and LT (**Figure 1**). Algal size measurements and cell per colony estimates were available for 1993, only. From these data, the equivalent chlorophyll$_a$ concentrations for each algal group and the estimated total chlorophyll$_a$ concentrations were calculated as described in **section 4.1.5**.

4.3 Summary

In order to run the model successfully and validate the results, daily, weekly or, at least, fortnightly, data are required for the simulation runs. Unfortunately, the data for 1987 fell well short of the minimum requirement (**Tables 3, 4**). In contrast, the data for 1992 was well within this requirement (**Tables 3, 4**).

Table 3. A comparison of the sampling frequency of data supplied by AW for the 1987 and 1992 simulations. Daily and 'weekly' (including fortnightly) data are adequate for the simulation runs; 'other' data (e.g. monthly summary data) are not.

		1987			1992		
		daily	weekly	other	daily	weekly	other
N. Gwash	flow	(✓)			✓		
	OP			(✓)		✓	
	NO_3			(✓)		✓	
	SiO_2			(✓)		✓	
Egleton Brook	flow	✓			✓		
	OP			✓		✓	
	NO_3-N			✓		✓	
	SiO_2			✓		✓	
S. Gwash	flow	✓			✓		
	OP			✓		✓	
	NO_3-N			✓		✓	
	SiO_2			✓		✓	
R. Nene	flow			✓	✓		
	OP		✓			✓	
	NO_3-N		✓			✓	
	SiO_2			✓		✓	
	flow			✓	✓		
	OP		✓			✓	
	NO_3-N		✓			✓	
	SiO_2			✓		✓	
R. Gwash	flow			✓	✓		
R. Witham	flow			✓	✓		
Wing	flow			✓	✓		
Reservoir	temp.			✓		✓	

Table 4. Frequency of records supplied by AW as a validation dataset for the simulations. Daily and 'weekly' (including fortnightly) are adequate for comparison with the model simulations; 'other' sampling intervals are not.

	1987			1992		
	daily	weekly	other	daily	weekly	other
Zooplankton			✓		✓	
Chlorophyll			✓		✓	
OP		✓			✓	
NO_3-N		✓			✓	
SiO_2		✓			✓	
Reservoir volume			✓	✓		

5. OBSERVED HYDROLOGICAL, TEMPERATURE, NUTRIENT AND PHYTOPLANKTON FEATURES FOR 1987 AND 1992

5.1 Hydraulic loads

Rutland Water is fed by three natural inflows and two pumped supplies which are abstracted from the Rivers Nene and Welland. The three natural inflows accounted for only 18.1% of the total inflow to the reservoir in 1987 and even less, i.e. 8.7%, in 1992. The remainder came from the Rivers Nene and Welland which contributed 47.5% and 34.4% of the total inflow, respectively, in 1987, and 74.4% and 16.9%, respectively, in 1992.

The total amount of water entering the reservoir during 1987 was 52,124 $x10^3$ m^3 (**Table 5**), while the corresponding figure for 1992 was 98,471 $x10^3$ m^3 - an increase of about 89% (**Table 5**). Most of this increase was due to greater abstraction from the River Nene, as the contributions from the N.Gwash, S.Gwash, River Welland and Egleton Brook varied by only about 10%, between these years. In total, 195% more water was pumped into the reservoir from the River Nene during 1992 (73,230 $x10^3$ m^3) than during 1987 (24,748 $x10^3$ m^3).

The measured volume of the reservoir ranged between 1.196 $x10^8$ m^3 (87.4%) and 1.316 $x10^8$ m^3 (96.1%) in 1987, and 0.866 $x10^3$ m^3 (63.2%) 1.311 $x10^8$ m^3 (95.8%) in 1992. Reservoir volumes predicted on the basis of inflow and outflow data provided by AW were expected to be similar to those estimated from the reservoir depth measurements, using the volume/depth calibration curve. However, this was only true for 1992 (**Figure 4**, lower panel). During 1987, the predicted volume fell well short of the measured volume throughout the year (**Figure 4**, upper panel). This suggested some inconsistencies within the original dataset.

5.2 Phosphorus inputs

This report focuses on ortho-phosphate (OP) as it is this form of phosphorus which is most available to the algae and is, potentially, the main growth limiting nutrient.In 1987, the estimated total OP loading to Rutland Water was 37,854 kg (**Table 6**), 80.5% of which entered the reservoir in the water abstracted from the River Nene. In 1992, the equivalent

Table 5. A comparison of the hydraulic loadings to Rutland Water during 1987 and 1992

		Total water in ($x10^3$ m^3)		% Change total water in between 1987 & 1992	% water in	
		1987	1992		1987	1992
Inflows	N. Gwash	1456	1302	-11	2.8	1.3
	S. Gwash	7287	6512	-11	14.0	6.6
	Egleton Brook	701	759	8	1.3	0.8
Abstraction	Nene	24748	73230	195	47.5	74.4
	Welland	17932	16668	-7	34.4	16.9
	Total	52,124	98,471	89	100.0	100.0

Table 6. A comparison of the potential orthophosphate loadings to Rutland Water during 1987 and 1992, prior to ferric dosing.

		Total OP load to the reservoir (kg y^{-1})		% Change in OP load between 1987 & 1992	% of total OP load to the reservoir	
		1987	**1992**		**1987**	**1992**
Inflows	**N. Gwash**	517	186	-64	1.4	0.2
Inflows	**S. Gwash**	1249	929	-26	3.3	1.0
Inflows	**Egleton Brook**	119	111	-7	0.3	0.1
Abstraction	**Nene**	30494	89588	194	80.6	94.0
Abstraction	**Welland**	5475	4499	-18	14.5	4.7
	Total	37,854	95,313	152	100.0	100.0

values were 95,313 kg (**Table 6**) and 94%, respectively. Very little of the OP entering the system was attributable to the three natural inflows (5% in 1987, 1.3% in 1992). While the OP load from all other inflows was less in 1992 than 1987, the OP load in the water from the Nene abstraction, before treatment, was 194% higher in 1992 than 1987. The overall effect of this was an increase of 152% in the potential OP loading to the reservoir in 1992, compared to 1987. The actual OP load reaching the reservoir is not known as the effect of ferric dosing has never been quantified.

By examining the dataset for 1992, it was found that the total phosphorus [TP] concentration in the inflows was usually 1.4 times the OP concentration while, in the lake, it was generally 1.7 times the OP concentration. If we assume these relationships between TP and OP, then the total areal loading of P for Rutland Water was 4.2 g m^{-2} in 1987, and 10.6 g m^{-2} in 1992. The 1987 areal P loading is 42 times the 'permissible' and 21 times the 'dangerous' loading rate given in the OECD (1982) guidelines for a water such as Rutland with a mean depth of about 10 m. The corresponding value for 1992 is 106 times the 'permissible' and 53 times the 'dangerous' level for such a waterbody. The OECD (1982) categories reflect the likelihood of algal blooms and other biological manifestations of nutrient enrichment.

5.3 Water Temperature

The open-water temperatures recorded for 1987 and 1992 showed a similar pattern in each year (**Figure 5**). The annual minimum temperatures (3°C in 1987, 4.3°C in 1992) were recorded in late January and early February, respectively. The temperature rose steadily throughout spring and early summer reaching the annual maxima of 21.7°C at the end of June in 1987, and 21.6°C at the end of July in 1992. In both years, a secondary peak followed (19.3°C in mid-August, 1987, 19.0°C in early August, 1992) before the temperature dropped steadily during late summer and autumn, reaching typical winter values of 3°C and 5.4°C by the end of 1987 and 1992, respectively. It should be noted, however, that some of the temperature data used for 1987 came from Grafham Water rather than Rutland Water (see **section 4.1.3**).

5.4 Nutrients in the Reservoir

The nutrient levels measured in the open water of the reservoir represent only that which is unused by biological processes, such as phytoplankton production. During 1987, OP levels ranged between 33 mg m^{-3} (April) and 130 mg m^{-3} (October) (**Figure 6**). The mean OP level for the year was 85 mg m^{-3}. In contrast, the OP levels in 1992 tended to be lower, ranging from 6 mg m^{-3} (September) to 87 mg m^{-3} (January), with a mean OP value of only 31 mg m^{-3}. All of these values are high and typical of a nutrient-enriched system. Although the potential OP loading from the Nene and Welland before ferric dosing is much higher in 1992 than 1987, the mean OP concentration in the reservoir is much lower in the latter year. This suggests that the actual OP loading of the treated water which discharges into the reservoir is much smaller than that of the untreated water.

Open-water nitrate concentrations were also high, ranging between 2100 mg m^{-3} (October) and 3750 mg m^{-3} (June) in 1987, and 3740 mg m^{-3} (January) and 6860 mg m^{-3} (April) in 1992 (**Figure 6**). The mean nitrate level for 1987 was 2913 mg m^{-3}, and that for 1992 was 5512 mg m^{-3}. The latter figure reflects the increased nutrient loading to Rutland Water during 1992, as nitrate is not removed by ferric dosing.

In both 1987 and 1992, silica levels tended to be high (around 3000 mg m^{-3}) in summer, autumn and winter, and lower (<1000 mg m^{-3}) in spring (**Figure 6**). The rapid decline in silica levels in March 1987 and April 1992 is typical of many freshwater lakes in temperate latitudes and suggests a spring diatom bloom.

5.5 Phytoplankton

A total of approximately 60 species of planktonic algae were listed in the AW records for Rutland Water 1987 and 1992 (**Table 1**). We have classed about 50% of these as only 'rare and occasional', and not of sufficient importance in terms of overall biomass (total cell volume, chlorophyll$_a$ content) to merit consideration in the attempts to match the phytoplankton cell counts to the measured chlorophyll$_a$ values (see below). Nevertheless, the predominance of small green algae in this group (mainly Chlorococcales), is one indication of the rich nature of this reservoir. Other Chlorococcales (in groups (i) and (iii) in **Table 1**), small *Stephanodiscus* spp. (ii), *Aulacoseira* (*Melosira* - v) and *Asterionella* (vi) in addition to the classic blue-green species (vii and viii) are further evidence of eutrophication.

Interestingly, however, phytoplankton population and crop densities are not so in keeping with the nutrient-rich character of the reservoir. Neither of the mean annual chlorophyll$_a$ values calculated from the data shown in **Figure 7** - 9.8 mg m^{-3} and 7.4 mg m^{-3} for 1987 and 1992 respectively - is remarkable. What is more, some of these values probably stem from samples restricted to the epilimnetic layer, which would tend to over-estimate the reservoir-wide values, although others may relate to periods of blue-green algal blooms which tend to accumulate around the edge of the lake causing an under-estimate of open-water chlorophyll$_a$ levels. While it is likely that the overall chlorophyll$_a$ levels in 1992 are less than those recorded in 1987, it should be stressed that (i) chlorophyll$_a$ levels were not measured during March 1992 when a considerable spring bloom of diatoms seems to have occurred, and (ii) there are also no pigment data for the (critical?) period June and July 1987.

Chlorophyll$_a$ values estimated from the cell counts (by assigning appropriate pigment per unit cell volume values) certainly reflect the moderate chlorophyll$_a$ concentrations actually measured (**Figures 8 and 9**). There are only two peaks of any significance in 1987 i.e. that corresponding to a value of *ca* 15 μg chlorophyll$_a$ l^{-1} due to *Stephanodiscus* in March, and *ca* 35 μg chlorophyll$_a$ l^{-1} with *Anabaena* dominant in June. There is relatively greater showing of small green algae in 1992, but considerably less Anabaena and *Stephanodiscus* compared to 1987.

Not surprisingly, the fit of the pigment levels calculated from the cell counts to the measured values is rather poor for the 1987 dataset (**Figure 10**). In contrast, the fuller set of information for 1992 has resulted in an exceedingly close resemblance between the derived and the measured values. This is all the more remarkable, since conventional measurements of chlorophyll$_a$ overestimate the actual levels of this pigment, as they do not take account of 'dead' chlorophyll$_a$, i.e. pheophytin$_a$, which has a spectrophotometric absorbance peak which is at virtually the same wavelength as that of the 'healthy' pigment.

5.6 Making sense of the data - hydraulic and phosphorus balances, and the relationships between P loading, in-lake OP concentration and chlorophyll$_a$ levels

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 Rutland Water make sense. This approach has been found to be very useful in a wide variety of situations (Bailey-Watts, May, Kirika, and Lyle 1992; Bailey-Watts and Kirika 1993; Bailey-Watts 1994). These relationships consider the total phosphorus (TP) rather than orthophosphate (OP); in comparing the measured OP values with the predicted TP values, it has been assumed that, for the inflows, TP is 1.4 times the OP figure, and, for the lake, TP is 1.7 times the OP value (see **section 5.2**).

These relationships predict a mean annual TP concentration of 257 mg m^{-3} in 1987 and 615 mg m^{-3} in 1992 for Rutland Water, which are equivalent to predicted mean annual OP concentrations of 151 mg m^{-3} for 1987 and 362 mg m^{-3} for 1992. The predicted mean OP concentration for 1987 is approximately double the observed value of 85 mg m^{-3}. As much of the OP entering a lake is changed into particulate form, actual OP values are often less than the predicted figure, and these rather crude models cannot be used as accurate predictors of the real situation. All in all, the predicted and measured values are close enough to suggest that the 1987 data do make sense. The situation for 1992 is rather different as the predicted value of 362 mg OP m^{-3} is very much greater than the observed value of 31 mg OP m^{-3}. However, the predicted value is based on the potential P loading from the Welland and Nene before treatment. So, a very high predicted value compared to the measured value suggests that the ferric dosing is significantly reducing the P loading to the reservoir. It is difficult to say, from these data what that reduction is but it may be around 70-90%.

6. MODEL SIMULATIONS OF OBSERVED NUTRIENT AND PHYTOPLANKTON DYNAMICS IN 1987 AND 1992

6.1 Simulating the 1987 data

In the first instance, the simulation was run using initial settings specific to Rutland Water, as shown in **Table 7**. Estimated daily values for lake water temperature, rate of flow and nutrient concentrations of the 5 feeder waters, and rate of flow of the 3 pumped outflows were used as inputs to the model. At this stage, it was assumed that all of the expected links between nutrient availability, flushing rate or zooplankton grazing and phytoplankton growth would operate to control the observed chlorophyll$_a$ levels and that there was no recycling of nutrients from the sediments.

The initial results of this simulation are shown in **Figure 11**. While temporal shifts in the measured concentration of NO_3-N are very well represented by the model, the correspondence between the measured and predicted values for the remaining parameters are disappointing. The marked discrepancy between measured and predicted SiO_2 concentrations during the latter half of the year is almost certainly due to the lack of SiO_2 data for the Welland and Nene during 1987. Clearly, using 1992 data as a substitute for the missing 1987 data was inappropriate. This conclusion was also supported by the fact that a good fit between predicted and observed SiO_2 concentrations for this lake had been achieved using the correct data for 1992 (**see section 6.2**). The 1992 data was therefore adjusted for use with the 1987 simulation in order to improve the goodness of fit between the modelled and measured SiO_2 concentrations. This involved increasing the SiO_2 loading by a factor of 10 between mid April and the end of July. This considerably improved the SiO_2 simulation for 1987, as shown in **Figure 12**, but it seems unlikely that such a large SiO_2 loading came entirely from the inflows during this year. Recycling of SiO_2 from the sediments, as is known to occur in Rutland Water during the summer months [Ferguson 1981, *cit.* Low, 1982], almost certainly made a significant contribution to the silica budget in 1987.

Although the SiO_2 and NO_3-N simulations now fitted very well, the predicted and measured values for chlorophyll$_a$ and OP, and for the individual algal groups, were still rather disappointing (**Figure 12, 13**). However, it was not possible to improve the model any further on the basis of the 1987 data alone. During the progress of this study it became clear that the data for 1987 were not as complete as originally envisaged and the measured chlorophyll$_a$ concentrations, in particular, were so infrequent that it was difficult to be sure how well the simulations fitted the observations. However, in the 1992 simulations, it became apparent that around 80% of the diatoms produced in Rutland Water do not appear in the plankton (see **section 6.2**). So, it was also assumed that this was the case in 1987 and the simulation was adjusted accordingly. As this improved the fit of the predicted values (**Figures 14,15**) whilst accounting for the substantial loss of SiO_2 from the lake in spring, it was concluded that 80% of the diatom production in 1987 was also lost from the chlorophyll$_a$ measurements and phytoplankton counts. The possible reasons for this are discussed in **section 6.2**.

Although the results of the simulations for 1987 were not as clear as might have been hoped, due to problems with the original dataset, the final predictions do fit the measured values reasonably well and seem to give a good indication of how the lake functioned before ferric dosing began.

6.2 The effect of ferric dosing on phosphorus loadings to the reservoir

Having simulated the 1987 data as accurately as possible, within the limitations of the dataset, the model could then be used to examine the situation which prevailed in 1992. Starting values for this year were supplied, as shown in **Table 7**, and, again, estimated daily values for water temperature, rate of flow and nutrient concentration in the 5 inflows, and rate of flow of the three outflows were provided as input. However, on this occasion, there was an additional, unknown variable which had to be estimated before the simulation could proceed. Since 1990, P had been removed from the water abstracted from the rivers Welland and Nene by ferric dosing, but the effect of this procedure on the P loading to the reservoir has never been quantified. So, the initial aim of the 1992 simulation was to run the model on data which had been collected from the reservoir since the ferric dosing began and estimate the rate of P removal. This was carried out by applying a stepwise reduction in the levels of OP in the water from these sources until the predicted OP levels in the open water in January, when biological activity was very low and removal of P minimal, were similar to the measured

Table 7. Initial values for the simulation of nutrient and phytoplankton dynamics in Rutland Water during 1987 and 1992.

		1987	1992
Number of inflows		5	5
Number of outflows		3	3
Maximum surface area (ha)		1260	1260
Maximum volume (m^3)		1.369×10^8	1.369×10^8
Initial volume (m^3)		1.196×10^8	0.868×10^8
Initial lake nutrient concentration (mg m^{-3})	OP	100	87
	NO_3-N	2785	4160
	SiO_2	2931	3130
Initial proportion of chlorophyll$_a$ per algal group	(i)	0.142	0.373
	(ii)	0.568	0.621
	(iii)	0.001	0.001
	(iv)	0.284	0.001
	(v)	0.001	0.001
	(vi)	0.001	0.001
	(vii)	0.002	0.001
	(viii)	0.001	0.001

concentrations. The results of these simulations (**Figure 16**) strongly suggest that ferric dosing is removing about 90% of the OP from this water supply before it enters the lake.

Assuming this 90% reduction, the model also predicts very closely the measured NO_3-N and SiO_2 levels (**Figure 17**), although the predicted chlorophyll$_a$ levels are considerably higher than the measured values. These results also suggest that chlorophyll$_a$ is, actually, being produced at the rate predicted by the model, otherwise the levels of 'unused' nutrients in the water column would be much higher. Clearly, some factor which has not been taken into account by the model is preventing the accumulation of chlorophyll$_a$ in the reservoir from reaching the predicted levels.

By examining the predictions for the individual algal groups, it became clear that diatoms - such as *Stephanodiscus*, *Fragilaria* and *Asterionella* - were far less abundant than predicted by the model (**Figure 18**). There were two possible reasons for this. The first was that diatoms were not growing in the reservoir at the rate predicted by the model. This seemed unlikely as the simulation for silica concentrations seemed to fit the observed values very well, and the rapid reduction in silica and orthophosphate levels in spring was consistent with rapid diatom growth usually observed in temperate lakes and reservoirs. The second was that diatoms were growing but not appearing in the water column in large numbers. This seemed the more likely explanation in view of the water chemistry data. The reason for the non-appearance of these algae could not be determined from the available data, but may have been due either to extensive blooms of benthic diatoms in the shallows, or to a significant loss of these heavy algae from the plankton due to settling. Whatever the reason, the model suggests that approximately 80% of the diatom growth in Rutland Water is 'lost' and does not contribute to the chlorophyll$_a$ values of standing crop recorded in the open water. If these diatoms are removed from the simulations the predicted values fit the measured values much more closely (**Figures 19,20**). The apparent over-prediction of blue-green algal densities by the model in late summer and autumn (**Figure 20**, '*Oscillatoria*') is almost certainly due to an underestimate of these algae in the algal counts. These algae tend to float to the surface and accumulate in sheltered bays down-wind of the prevailing wind in Rutland Water [Myers, Oldham & Ferguson 1982] and will not be detected in samples from the open-water.

In view of the above, it was concluded that the 1992 situation was best simulated by a 90% reduction in OP concentrations in the water abstracted from the rivers Welland and Nene, and an 80% loss of diatom growth from the open water chlorophyll$_a$ levels due to settling or benthic diatom growth. In addition, it was also concluded that the level of blue-green algal production predicted by the model was probably correct but, in reality, the algae were more likely to appear in bays around the edges of the reservoir rather than contribute to the open water chlorophyll$_a$ levels. This simulation was taken as the baseline simulation for 1992.

6.3 Limiting factors

Within the model it is possible to eliminate the effects of grazing, nutrient limitation and flushing in order to determine which factor, or combination of factors, is limiting the accumulation of algal biomass at any particular time. In 1992, it was found that, although P and grazing were not limiting individually, grazing (in removing algal cells) and phosphorus (in stemming cell production), together, were the dominant limiting factors until the end of May, while phosphorus, alone, was the major limiting factor later in the year (**Figure 21**). These results suggest that any changes leading to an increase or decrease in the OP loading to the reservoir would, similarly, result in an increase or decrease in chlorophyll$_a$ levels.

6.4 The effects of no phosphorus treatment

The model was then run to simulate the effect of discontinuing the current phosphorus removal programme whilst allowing P loadings to the reservoir to continue at 1992 levels. These results are compared to the baseline results for 1992 in **Figure 22**. The model clearly predicts that open-water OP concentrations would increase markedly early in the year, followed by a dramatic rise in chlorophyll$_a$ levels from June onwards. The most noticeable effect of this would be a massive build-up of blue-green algae ('*Oscillatoria*') in the latter half of the year (**Figure 23**).

6.5 The effects of reducing the OP concentration in the River Nene, whilst continuing ferric dosing

The model was also run to simulate the effects of reducing the OP concentration in the River Nene by 50%, as might be achieved by stripping phosphorus from the STW effluents which enter the river upstream of the abstraction point, whilst continuing the ferric dosing programme. **Figure 24** shows that the overall effect of this would be a marked reduction in chlorophyll$_a$ levels in autumn and a slight decrease during the rest of the year, and a marginal, though sustained, reduction in open-water OP concentrations, and a small increase in NO_3-N and SiO_2 levels. Some of the reduction in chlorophyll$_a$ would be reflected in a slight decrease in the abundance of all algal species, but the most marked reduction would be in the biomass of blue-green algae in late-summer and autumn (**Figure 25**).

6.6 The effects of reducing the OP concentration in the River Nene and discontinuing ferric dosing

If ferric dosing were discontinued, and P stripping of the STW effluent were to be undertaken as an alternative solution to the problem, the result would be somewhat different. Instead of an overall reduction in chlorophyll$_a$, the model predicts a massive increase in algal biomass in the reservoir, compared to 1992 levels (**Figure 26**). Moreover, most of this increase would appear in the form of blue-green algal blooms during the summer and autumn (**Figure 27**).

6.7 The effects of achieving target OP concentrations of 10 mg m^{-3} in the Rivers Welland and Nene

Finally, the effects of reducing OP concentrations in the Rivers Welland and Nene to AW's target values of 10 mg m^{-3} were simulated. The model predicts a marked reduction in chlorophyll$_a$ levels, especially in the latter half of the year, compared to the 1992 simulation (**Figure 28**), and an almost complete disappearance of blue-green algal blooms (**Figure 29**). Predicted OP levels were also lower early in the year, while NO_3-N and SiO_2 concentrations increased slightly due to a reduced rate of uptake of these nutrients by the growing algae.

6.8 The comparative effects of the different phosphorus management strategies

Figure 30 shows that, in comparison with some of the other P management strategies being considered by AW, ferric dosing is very effective in reducing phytoplankton biomass in Rutland Water. In spite of a potential OP load which was 2.5 times higher in 1992 than 1987, the simulations show that the actual load, after treatment, was low enough to keep chlorophyll$_a$ levels down to a mean of 10 mg m^{-3}, and a maximum of 23 mg m^{-3}. If there had been no reduction in the actual OP load during 1992, it is likely that enormous chlorophyll$_a$ concentrations (maximum 199 mg m^{-3}; mean 35 mg m^{-3}) would have been recorded, mostly in the form of blue-green algae.

However, the current P management strategy is under review. **Figure 30** shows that, removing 50% of the OP from the River Nene, in addition to ferric dosing, would result in a small, but sustained, reduction in phytoplankton biomass throughout the year (mean 8 mg m^{-3}; max. 19 mg m^{-3}). In contrast, removing 50% of the OP from the River Nene, but discontinuing ferric dosing, would produce an overall increase in chlorophyll$_a$ levels (mean 21 mg m^{-3}; maximum 87 mg m^{-3}) compared to observed 1992 levels. Achieving AW's target OP concentration of 10 mg m^{-3} in both the Nene and the Welland would be, by far, the most effective method of reducing phytoplankton biomass in the reservoir. This would reduce chlorophyll$_a$ levels to a maximum of 16 mg m^{-3} and an annual mean of 4 mg m^{-3}, whilst almost eliminating blue-green algal blooms, altogether.

7. CONCLUSIONS

1. The hydraulic load to Rutland Water was 1.89 times higher in 1992 than 1987.

2. In 1987, 48% of the inflowing water came from the River Nene and 34% from the River Welland. In 1992, those figures were 74% and 17%, respectively.

3. The potential OP load to Rutland Water was 2.5 times higher in 1992 than 1987 although, after ferric dosing, the actual load in 1992 was much lower.

4. In 1987, 81% of the OP load came from the River Nene abstraction. In 1992, the corresponding value for the raw, untreated, water was 94%.

5. Comparisons between the simulations and original data for 1987 were not as good as might have been hoped, due to limitations of the original dataset. The relationship between predicted and observed values for 1992 was very good.

6. Ferric dosing appeared to have reduced the potential OP load to Rutland Water by 90%, in 1992.

7. A combination of grazing by zooplankton and low phosphorus levels limited $chlorophyll_a$ accumulation during April/May, 1992; later in the year, P, alone, was the main limiting factor.

8. There was evidence of considerable diatom growth in spring, but these algae did not appear in the phytoplankton counts. This was thought to be due either to the settling out of these heavy algae from the plankton in calm weather, or to benthic diatom growth.

9. Mean $chlorophyll_a$ levels were very similar in 1987 (mean 9.8 mg m^{-3}) and 1992 (mean 7.4 mg m^{-3}), in spite of a massive increase in the potential OP load in the latter year. $Chlorophyll_a$ concentrations would have been much higher in 1992 in the absence of ferric treatment. Most of this increase would have appeared as blue-green algal blooms.

10. A 50% reduction in OP in the River Nene, in addition to ferric dosing, would produce a minor reduction in $chlorophyll_a$ levels compared to ferric treatment, alone.

11. A 50% reduction in OP in the River Nene, with no ferric treatment, would result in a significant increase in OP levels, in spring, and in $chlorophyll_a$ levels, later in the year. Most of the increase in $chlorophyll_a$ would appear as blue-green algae.

12. If OP concentrations in the Rivers Welland and Nene were reduced to 10 mg m^{-3}, $chlorophyll_a$ levels would be unlikely to rise above 15 mg m^{-3} in spring and would maintain levels of about 5 mg m^{-3} in summer and autumn. There would be very little production of blue-green algae.

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FIGURES

Figure 1. Sketch map of Rutland Water showing sampling sites N1, S13 and LT, points of entry of the 5 inflows and locations of 3 pumped outflows.

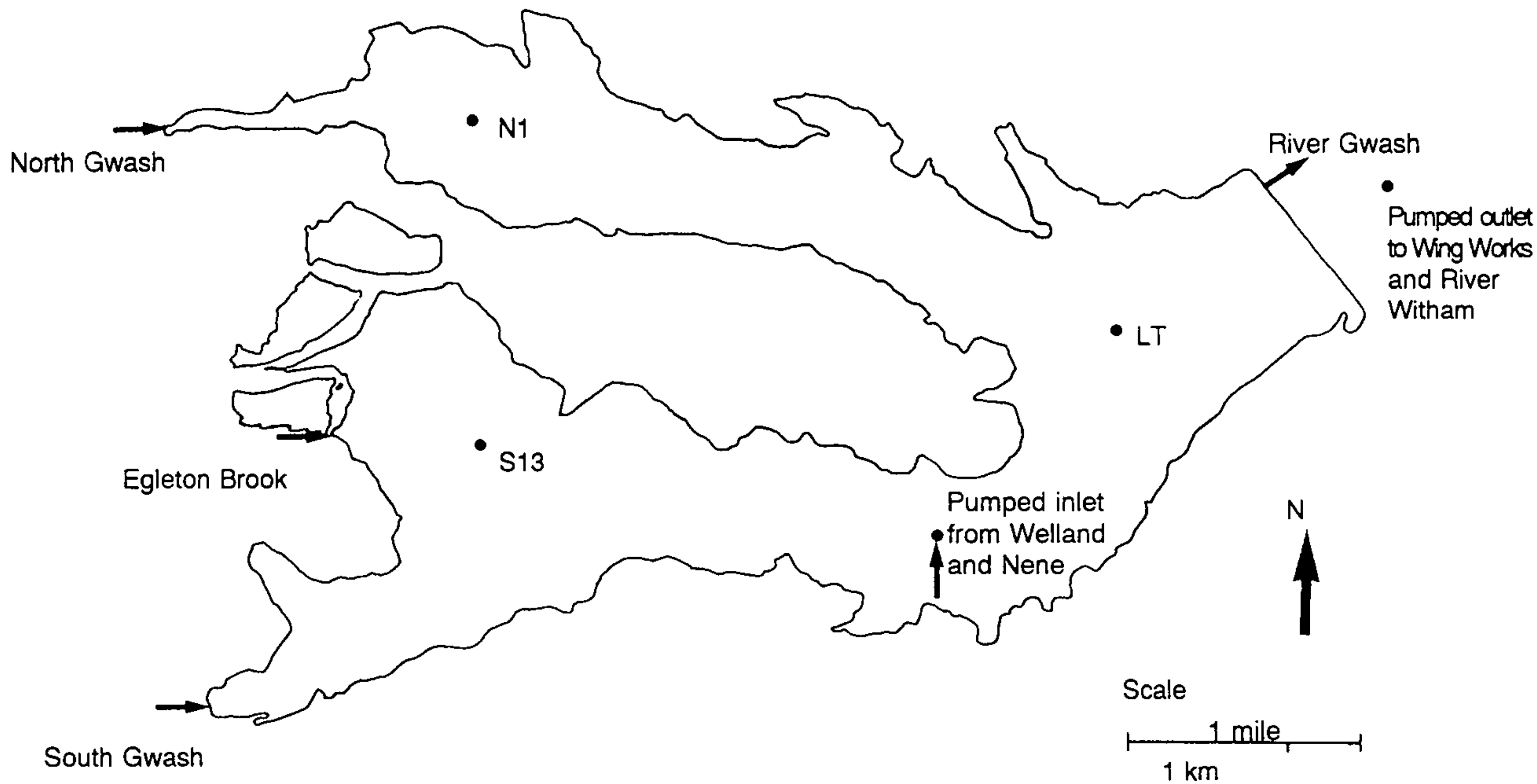


Figure 2. Predicted and observed concentrations of SiO_2 in Rutland Water during 1987 and 1992. The upper panel shows the predicted values before and after the SiO_2 loading was increased by a factor of 10 between mid April and the end of July, 1987.

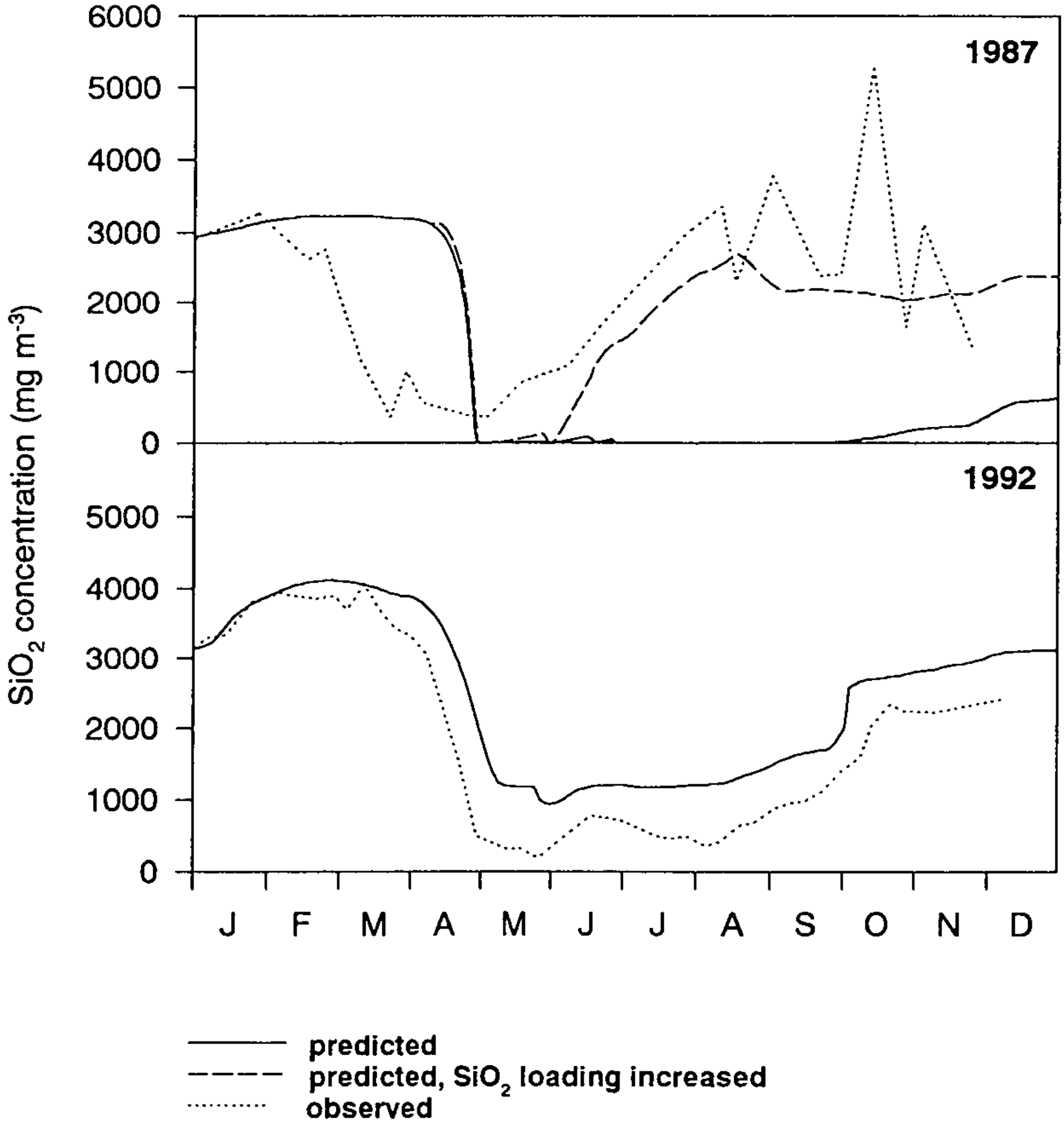


Figure 3. Observed chlorophyll$_a$, orthophosphate (PO_4-P), nitrogen (NO_3-N) and silica (SiO_2) concentrations in Rutland Water at sites LT, N1 and S13 during 1987.

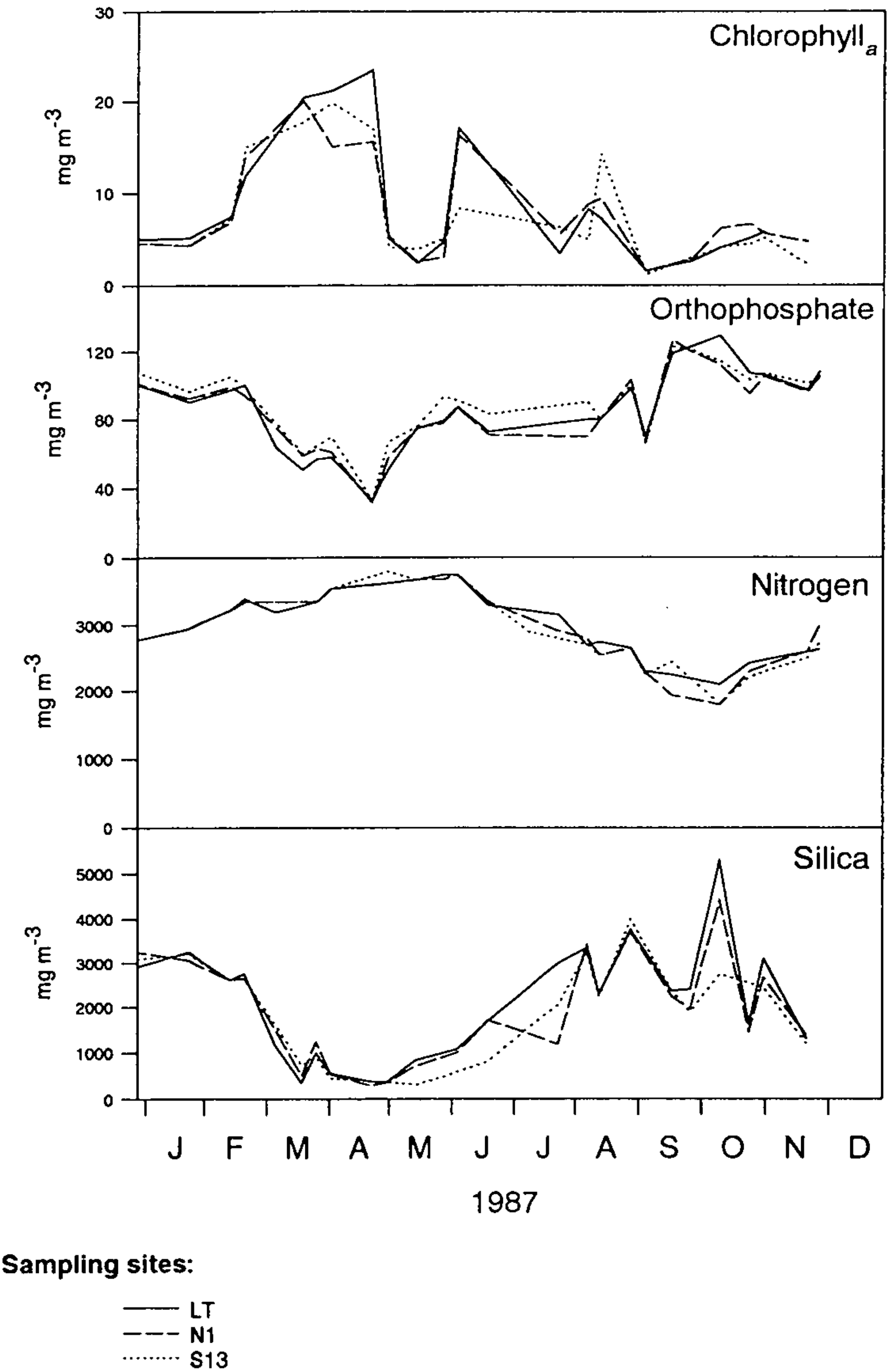


Figure 4. The relationship between observed reservoir volume (%full) and that predicted from the inflow and outflow volumes for 1987 and 1992.

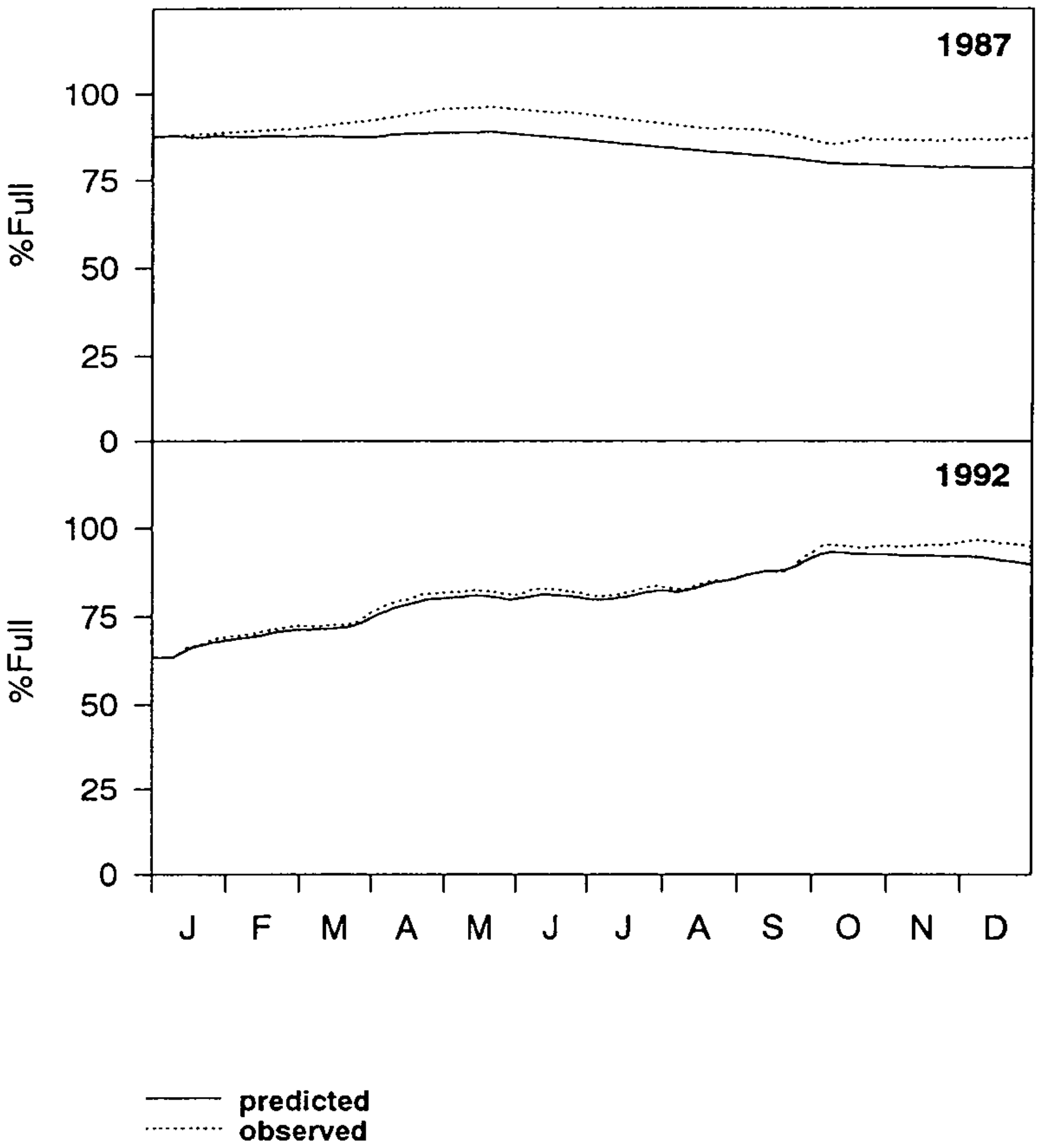


Figure 5. Surface water temperature of Rutland Water for 1987 and 1992. The period of missing data which was replaced by data from Grafham Water is shown in the upper panel.

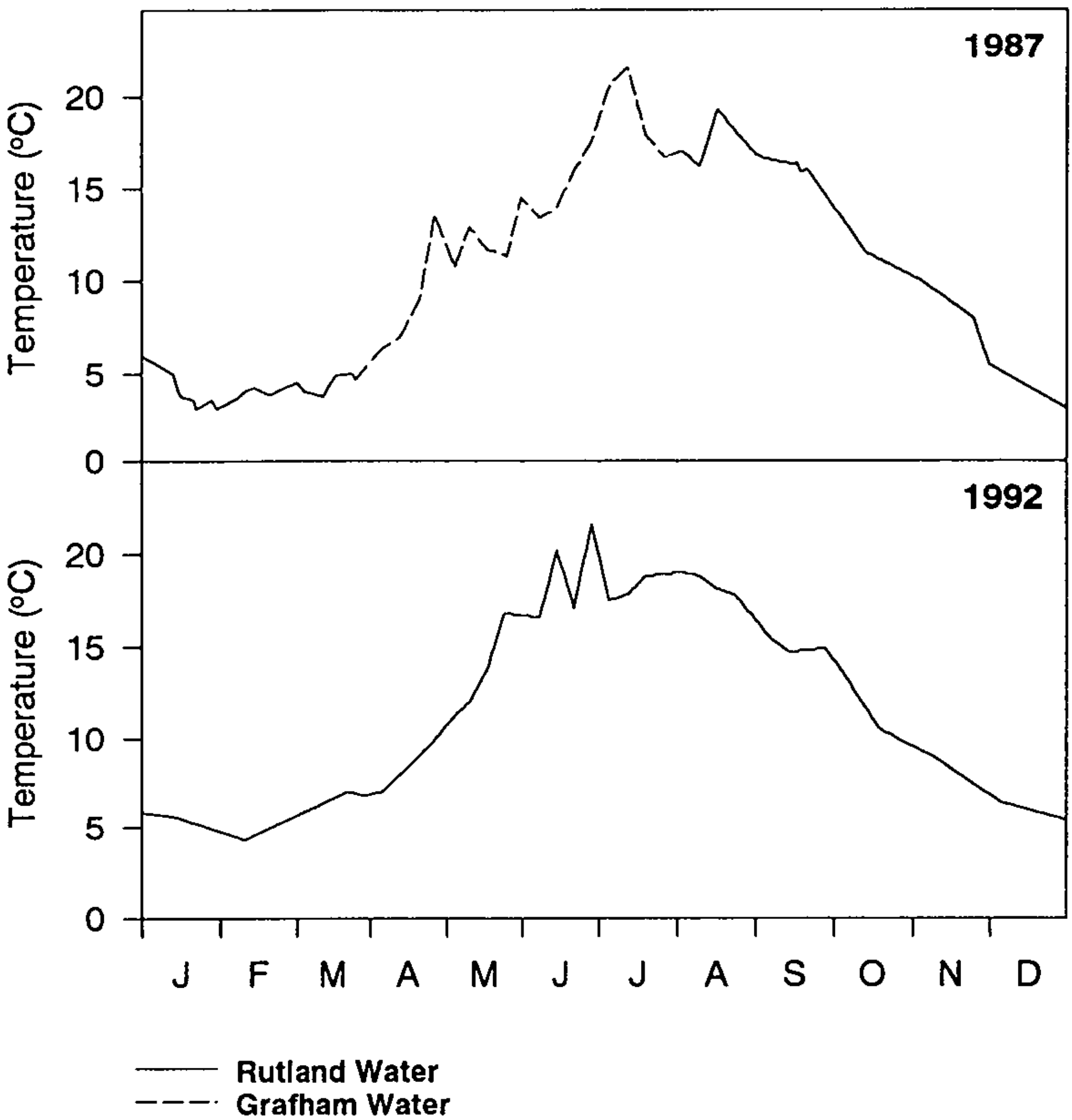


Figure 6. Observed concentrations of orthophosphate (PO_4-P), nitrogen (NO_3-N) and silica (SiO_2) in Rutland Water during 1987 and 1992.

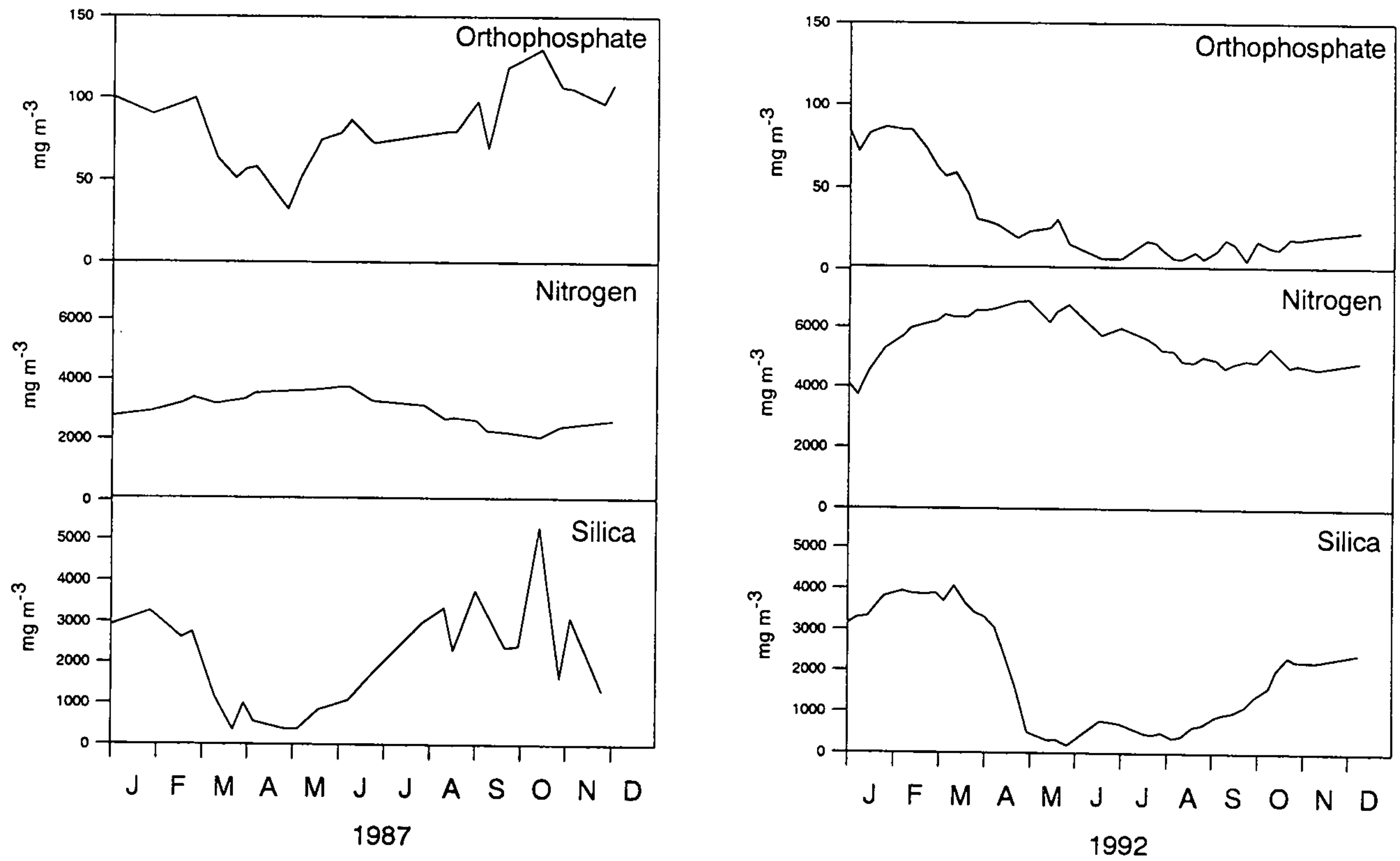


Figure 7. Measured chlorophyll$_a$ levels in Rutland Water for 1987 and 1992.

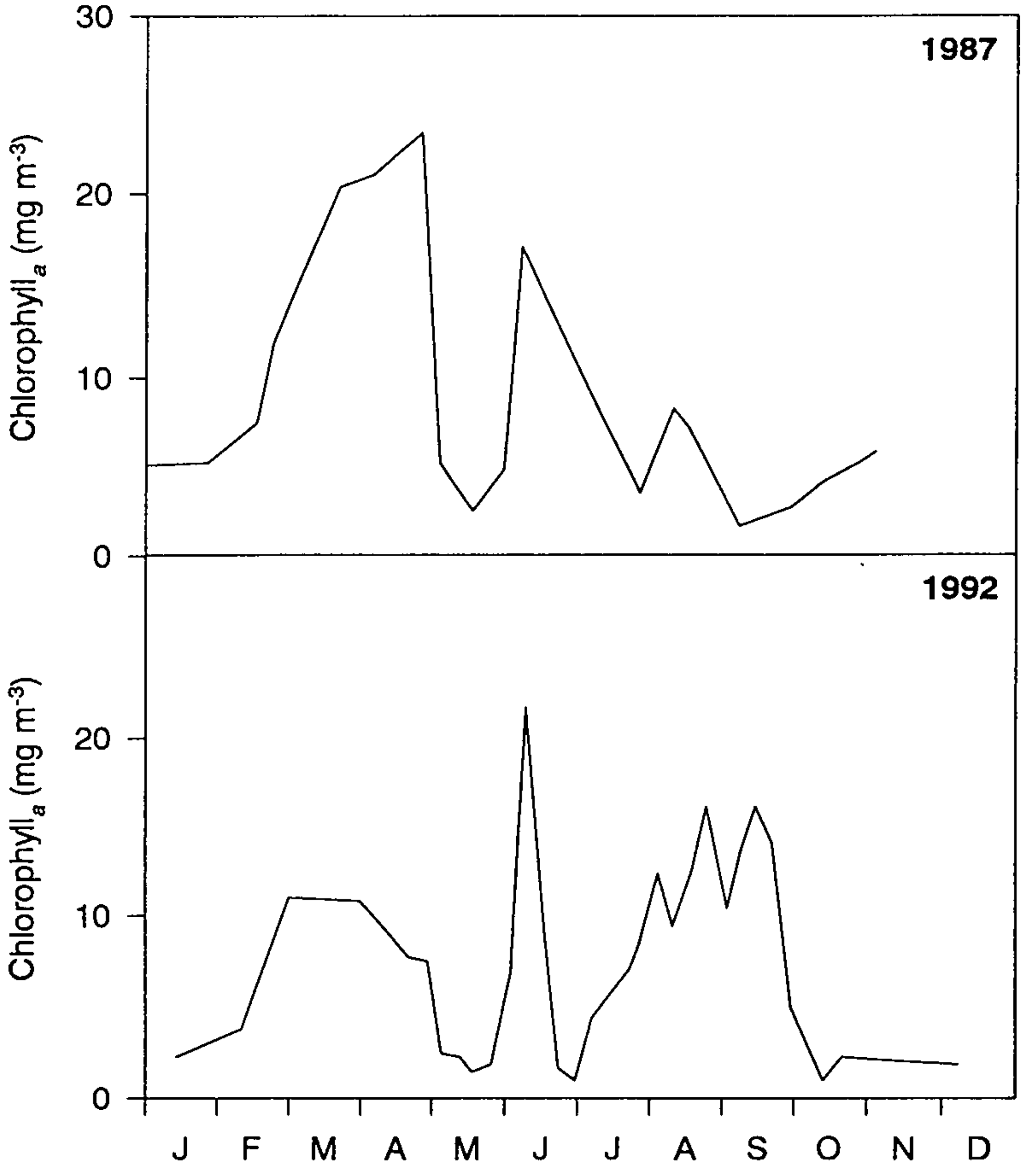


Figure 8. Open-water chlorophyll$_a$ values for each algal group for 1987, as estimated from the algal count data.

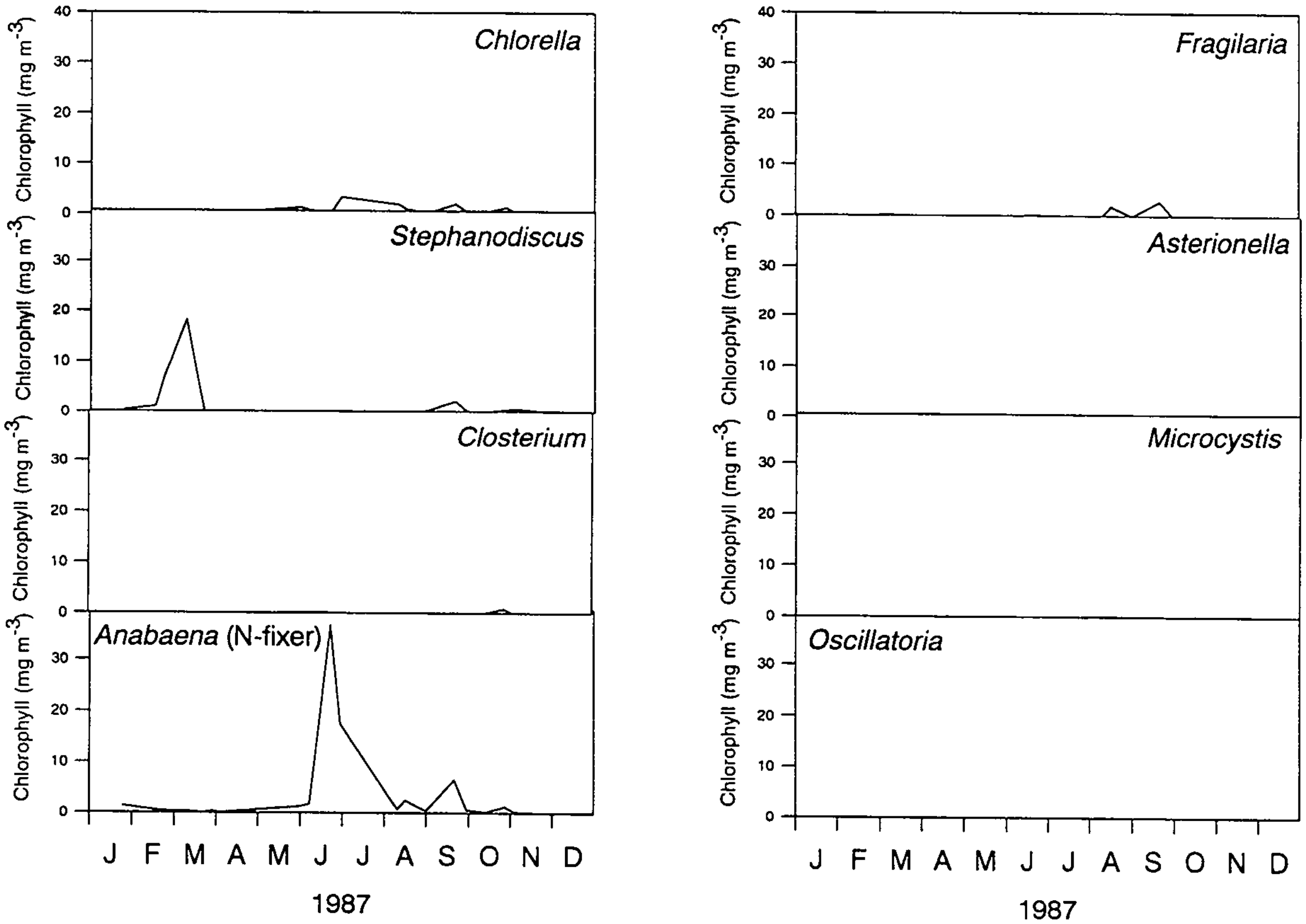


Figure 9. Open-water chlorophyll$_a$ values for each algal group for 1992, as estimated from the algal count data.

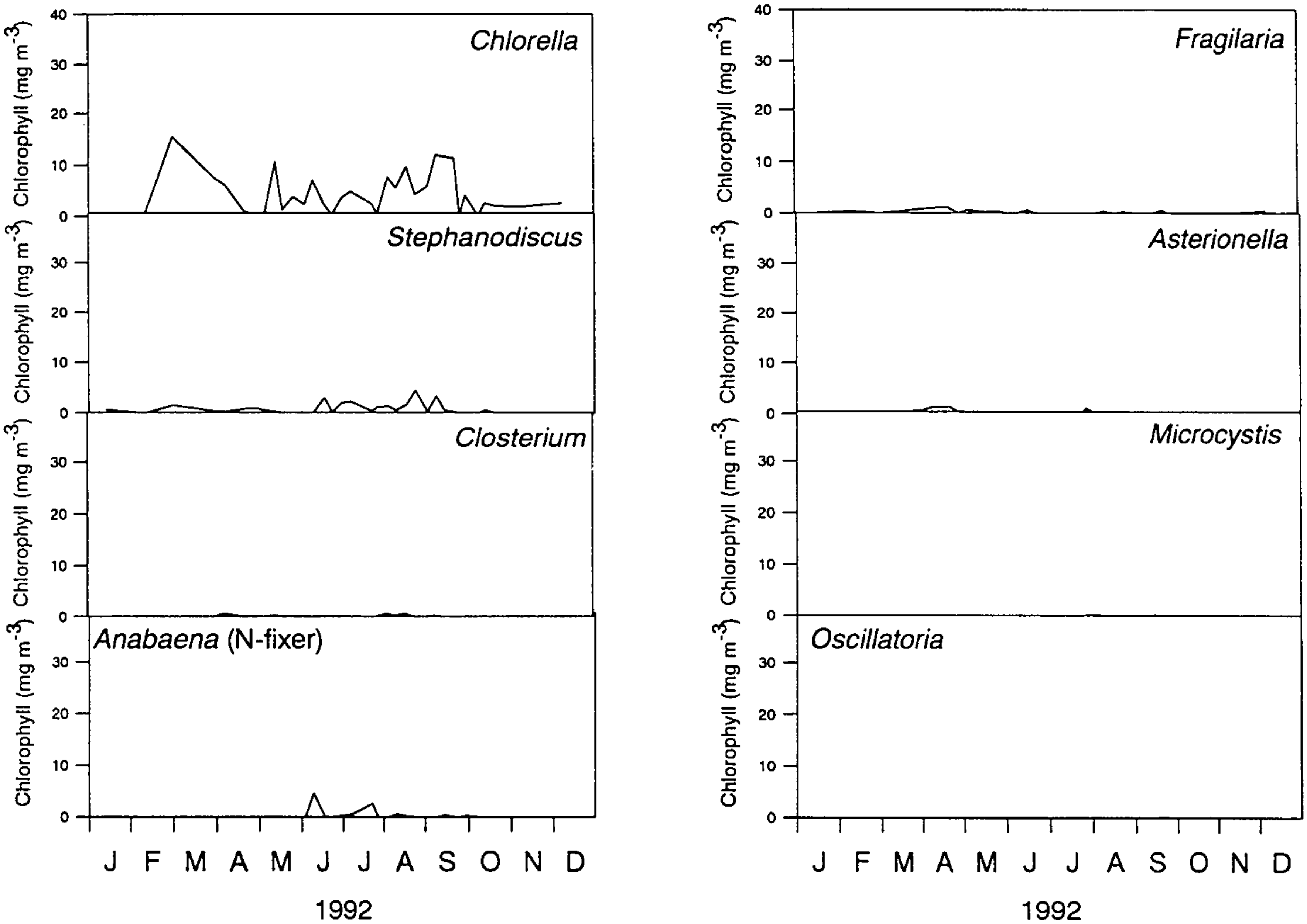


Figure 10. The relationship between measured total chlorophyll$_a$ concentrations and those estimated from the algal count data for 1987 and 1992.

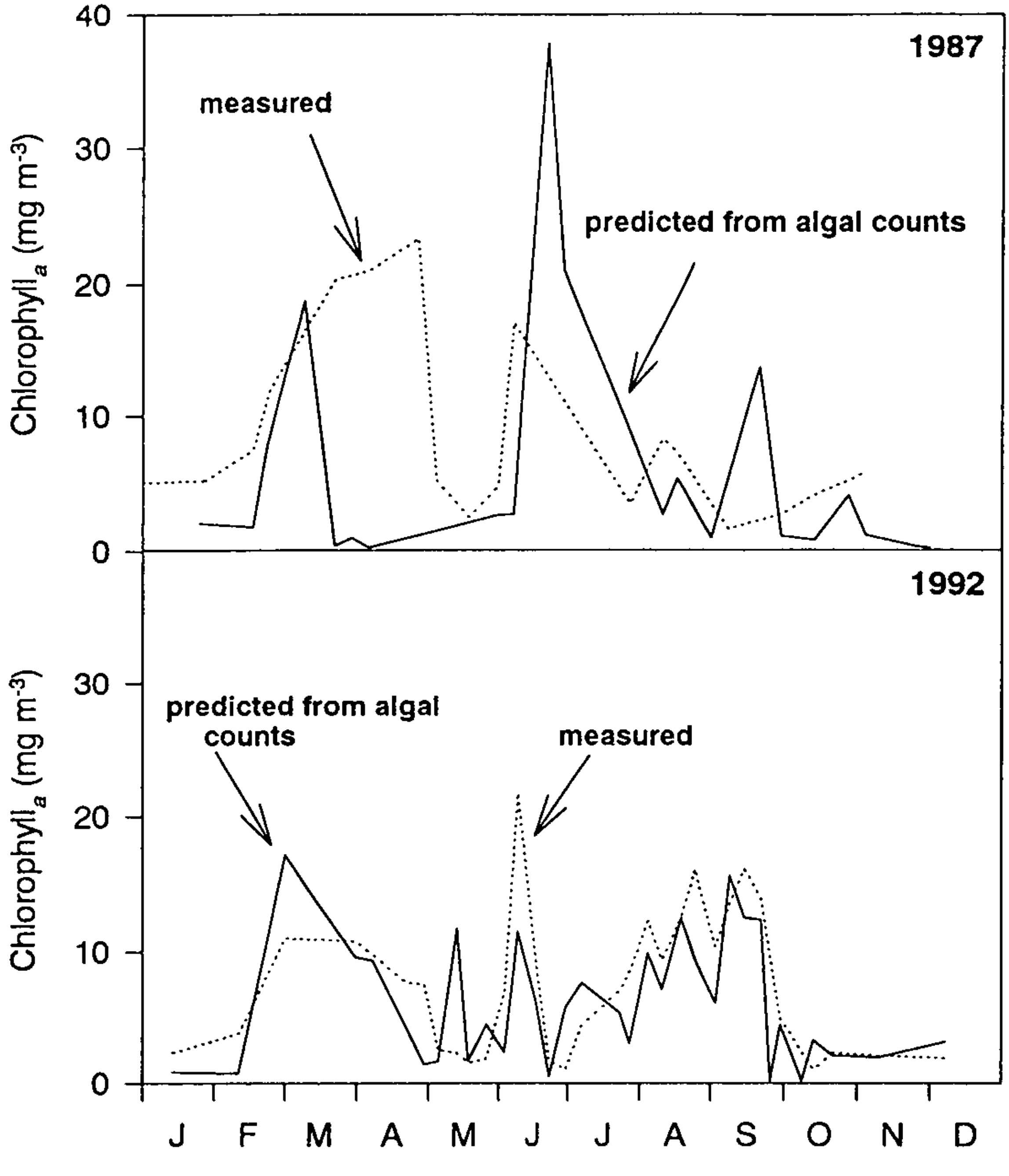


Figure 11. Initial simulation of nutrient and chlorophyll, concentrations in Rutland Water during 1987 compared to the observed values.

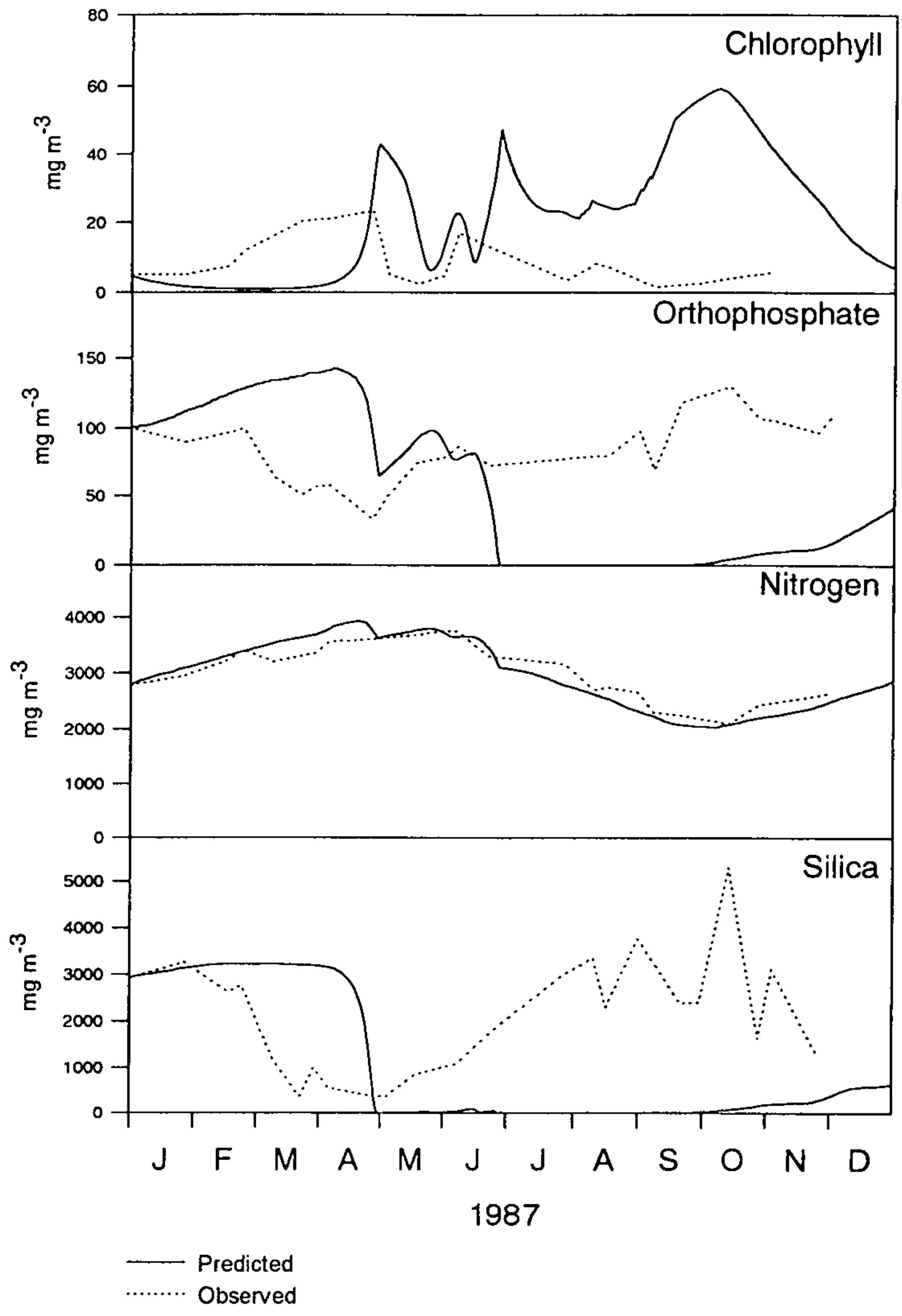


Figure 12. Initial simulation of nutrient and chlorophyll$_a$ concentrations in Rutland Water during 1987 compared to the observed values. SiO_2 loadings between mid-April and the end of July have been increased by a factor of 10.

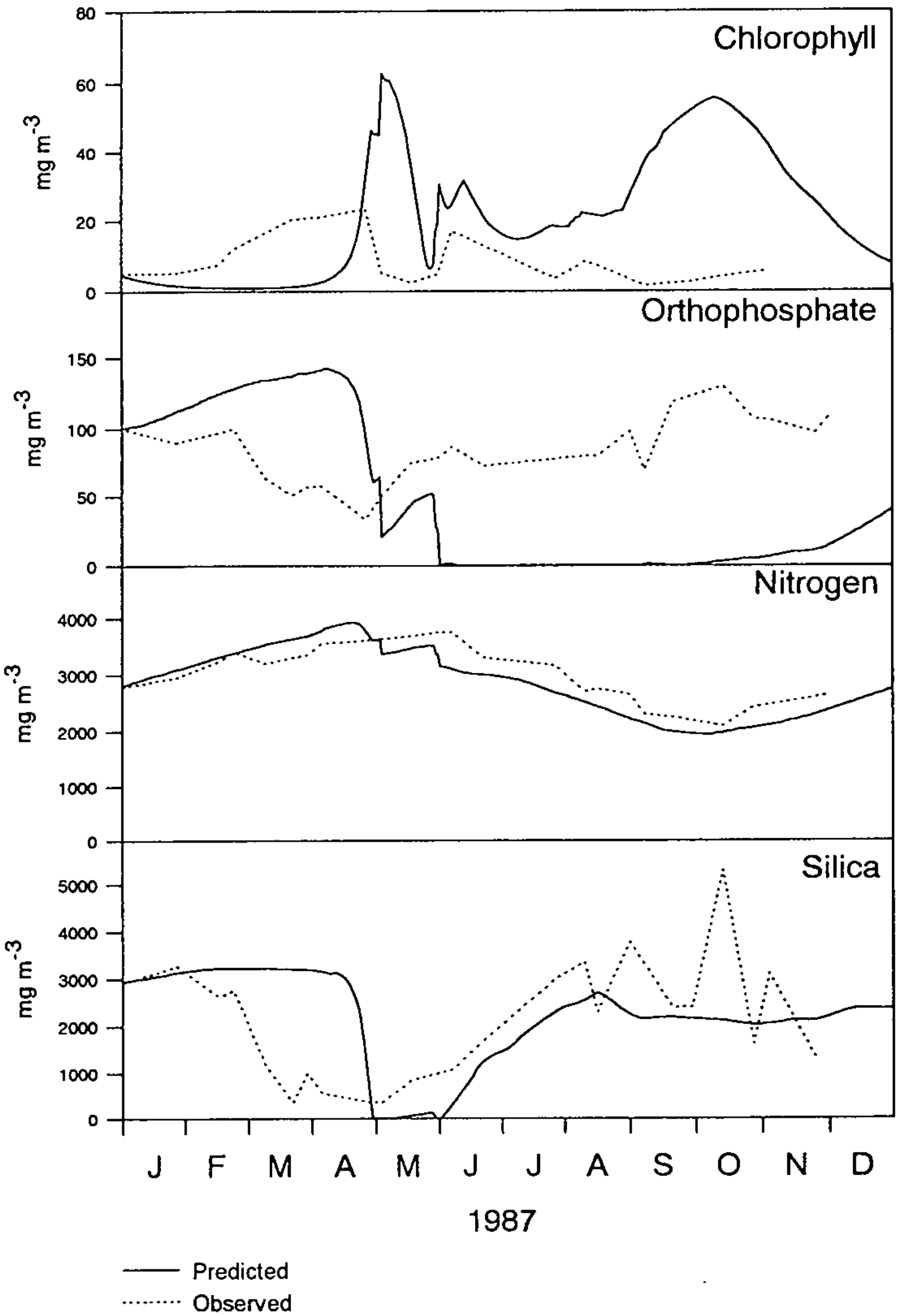


Figure 13. The predicted and observed abundances (expressed as chlorophyll$_a$) of each algal group during 1987. SiO_2 loadings between mid-April and the end of July have been increased by a factor of 10.

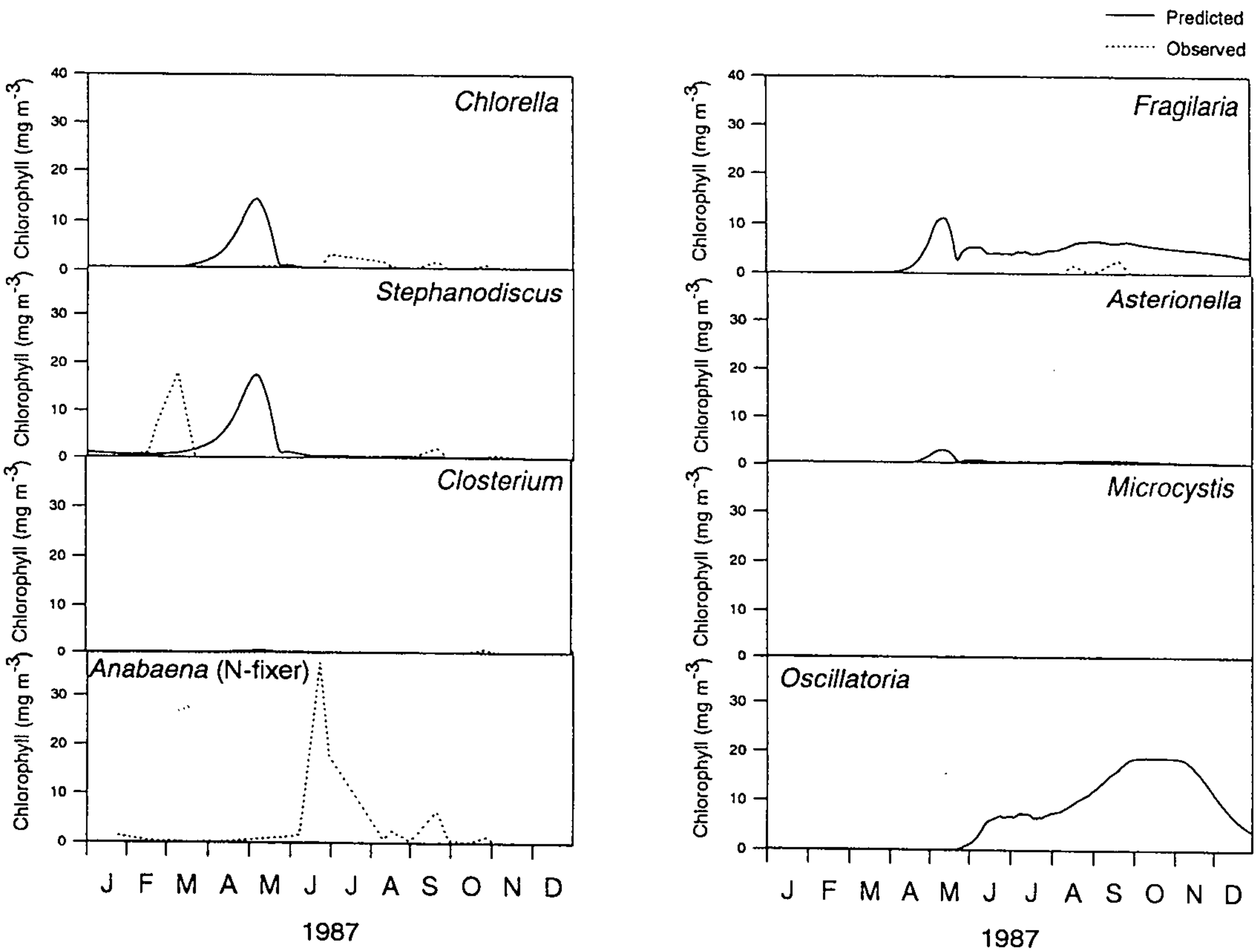


Figure 14. Predicted and observed nutrient and chlorophyll$_a$ concentrations in Rutland Water during 1987. Diatom abundance has been reduced by 80% to simulate the effect of settling or benthic diatom growth which does not appear in the plankton.

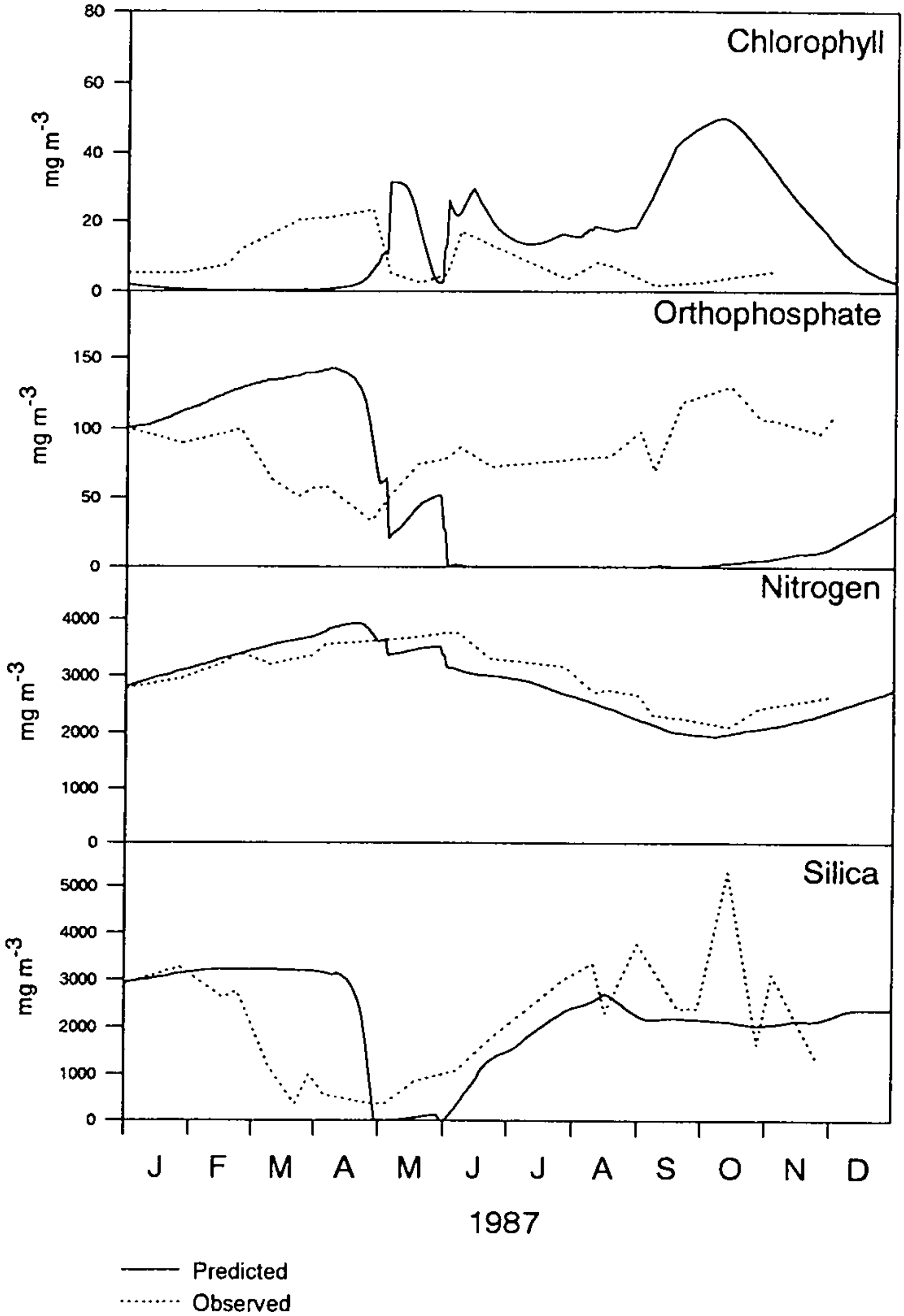


Figure 15. The predicted and observed abundances (expressed as chlorophyll$_a$) of each algal group during 1987. The simulation includes increased SiO_2 loading and an 80% reduction in diatom abundance.

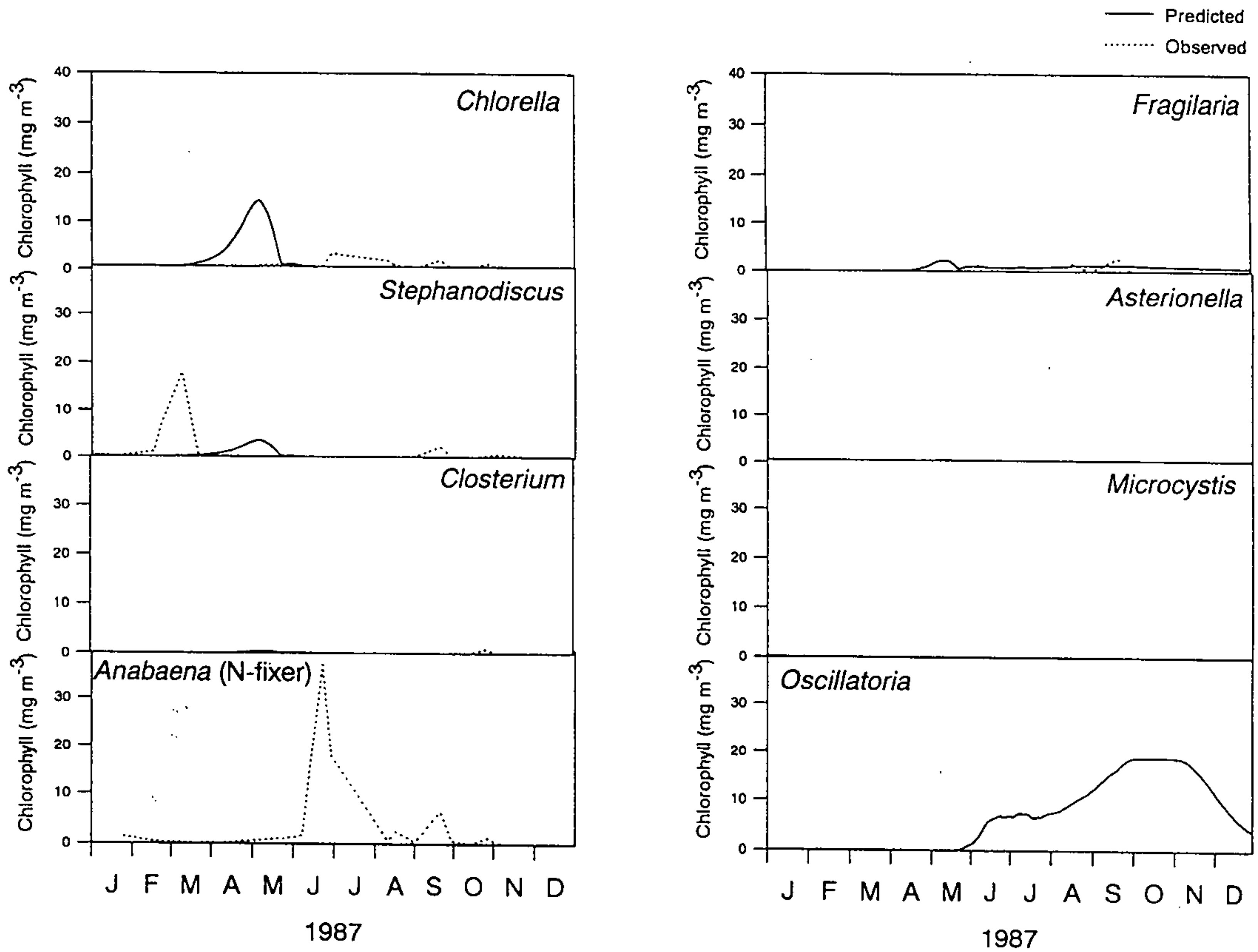


Figure 16. The predicted effect of stepwise reductions in orthophosphate concentrations in the water abstracted from the Welland and Nene on orthophosphate levels in Rutland Water.

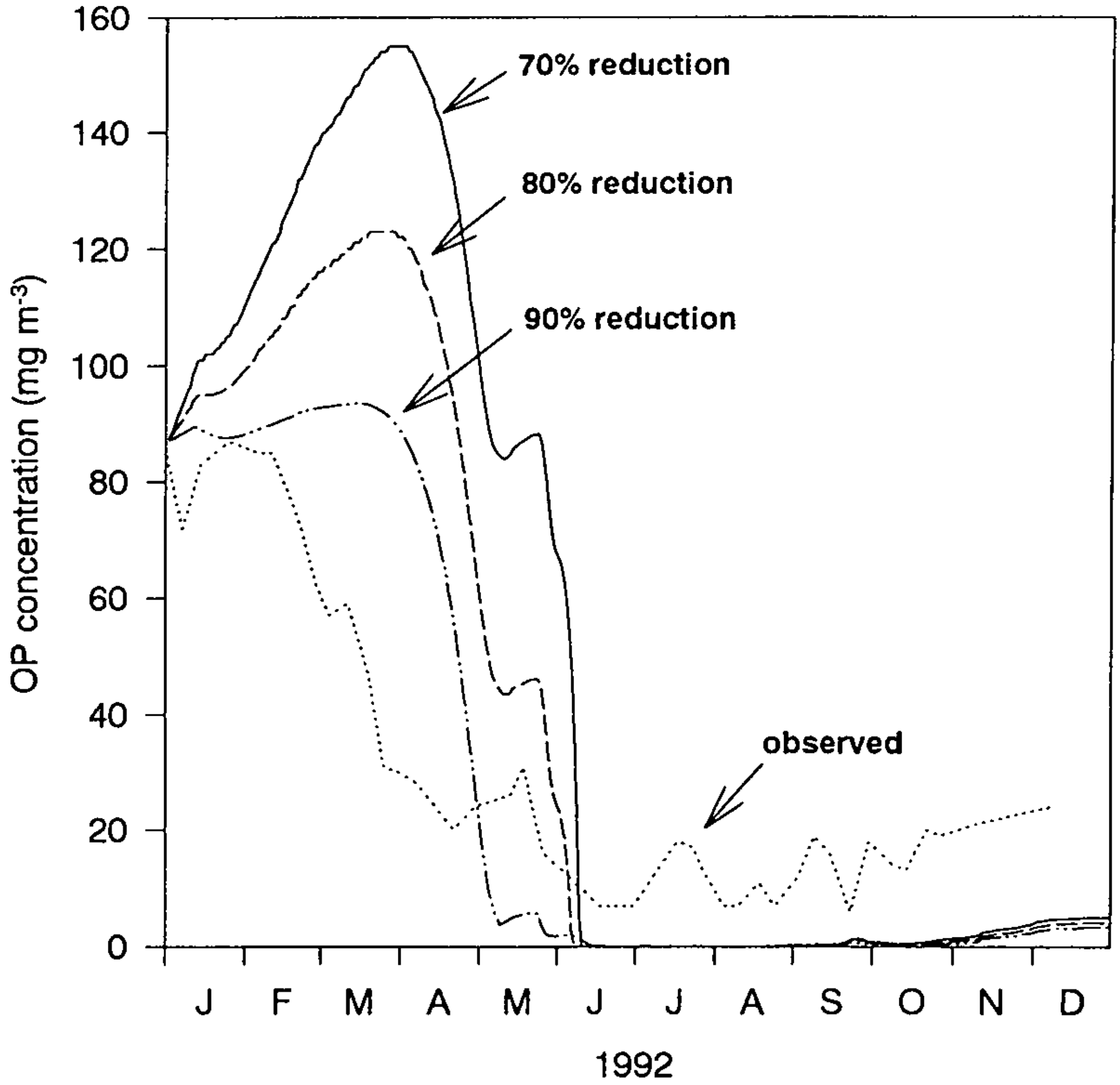


Figure 17. Predicted and observed nutrient and chlorophyll$_a$ concentrations in Rutland Water during 1992 assuming a 90% reduction in orthophosphate loading from the Welland and Nene due to ferric dosing.

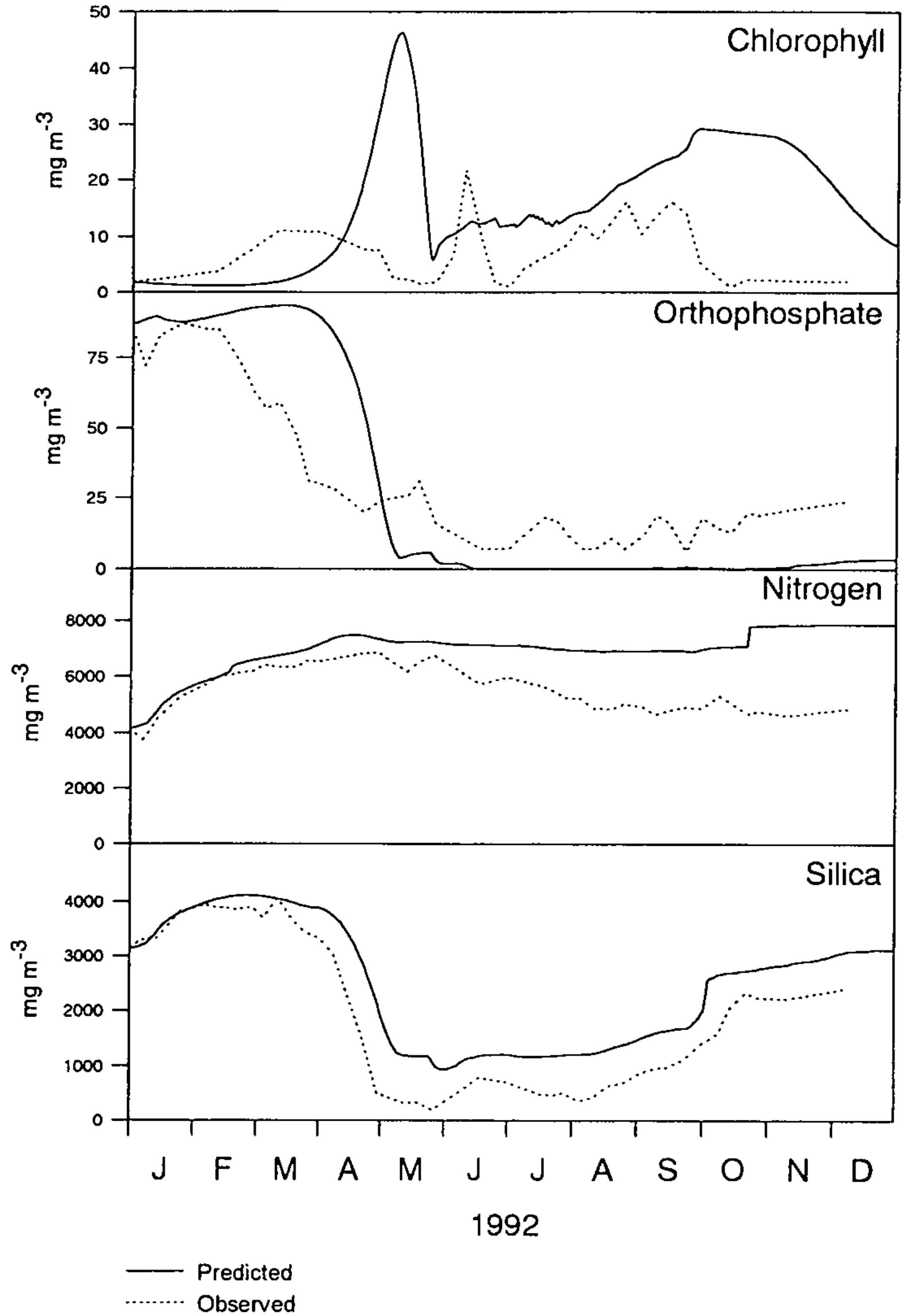


Figure 18. The predicted and observed abundance of each algal group (expressed as chlorophyll$_a$) in Rutland Water during 1992 assuming a 90% reduction in orthophosphate loading from the Welland and Nene due to ferric dosing.

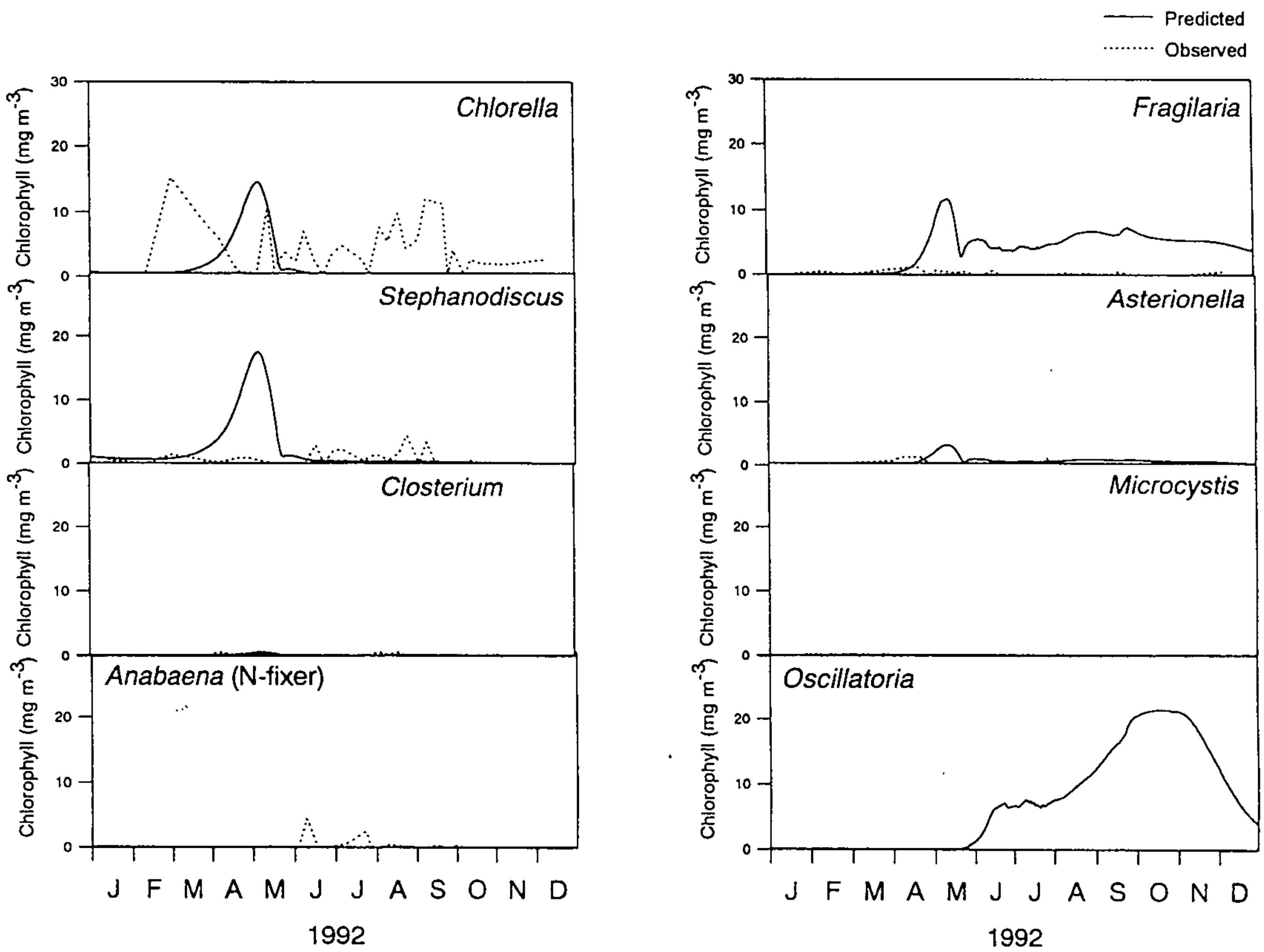


Figure 19. Predicted and observed nutrient and chlorophyll$_a$ concentrations in Rutland Water during 1992 assuming a 90% reduction in orthophosphate loading from the Welland and Nene due to ferric dosing and an 80% loss of diatoms due to sinking or benthic diatom production.

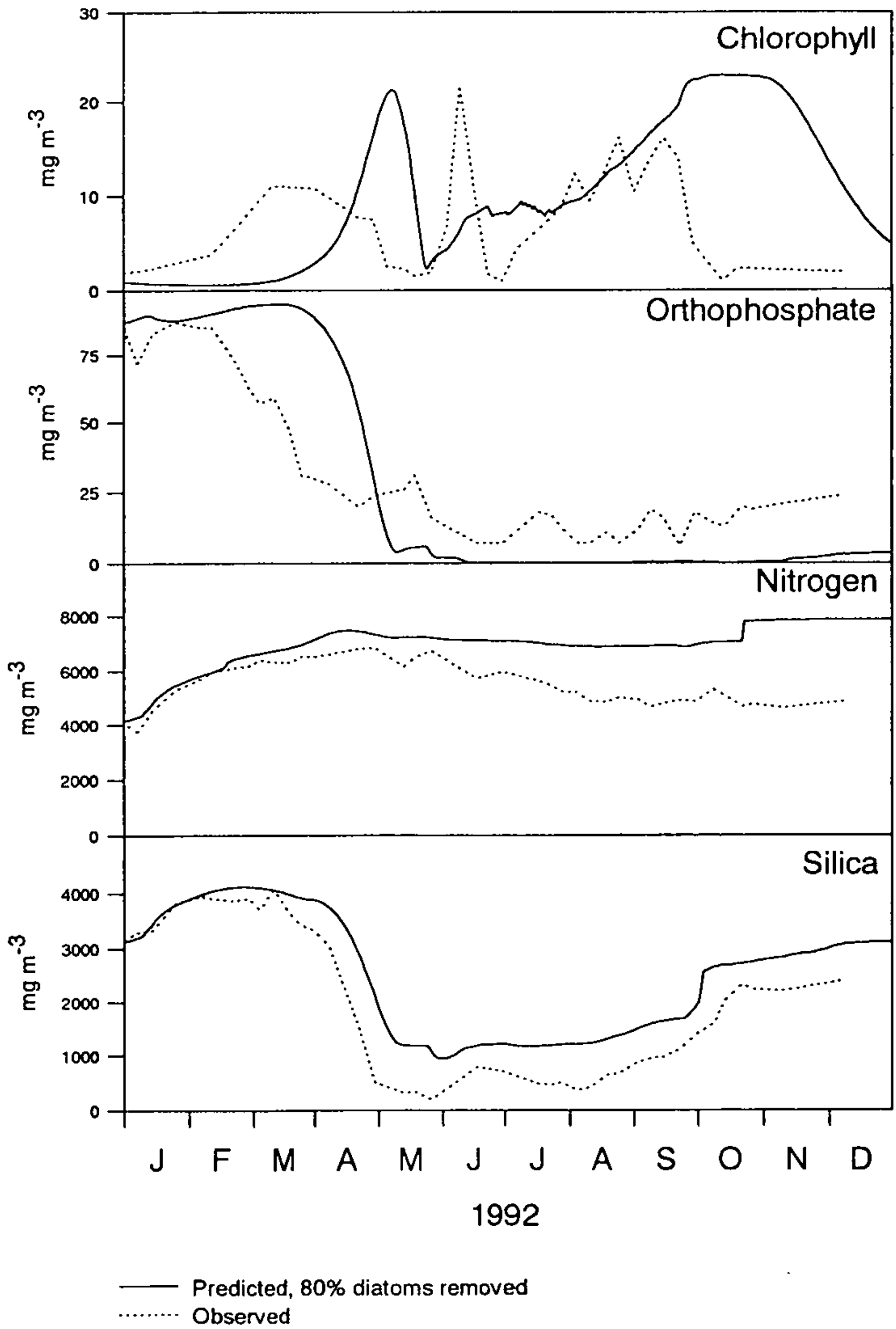


FIG19.SPG

Figure 20. The predicted and observed abundance of each algal group (expressed as chlorophyll$_a$) in Rutland Water during 1992 assuming a 90% reduction in orthophosphate loading from the Welland and Nene due to ferric dosing and an 80% loss of diatoms due to sinking or benthic diatom production.

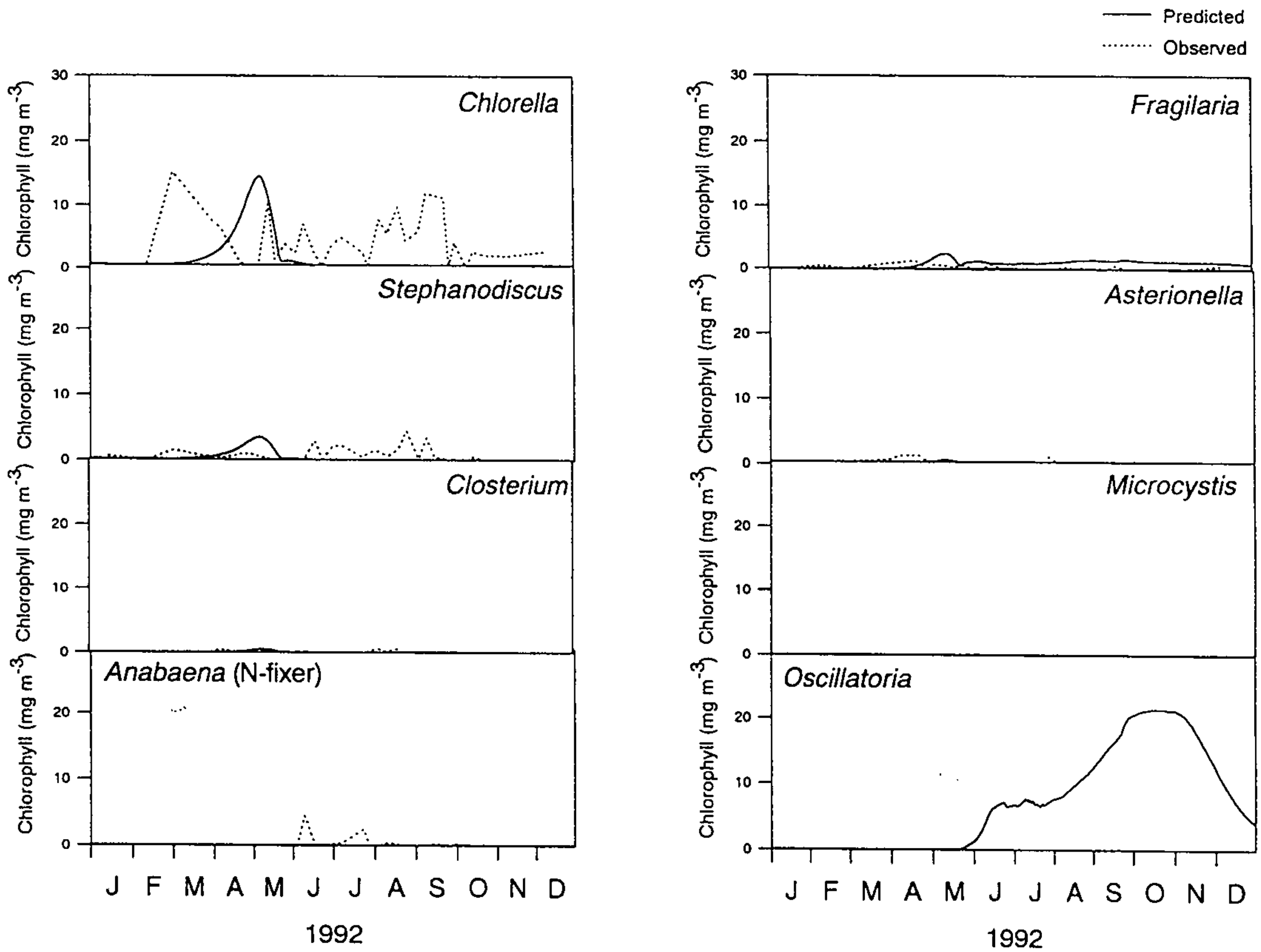


Figure 21. Predicted chlorophyll$_a$ levels in Rutland Water with phosphorus limitation and grazing limitation switched off separately and together. The results are compared to the simulation giving the best fit to the observed data for 1992.

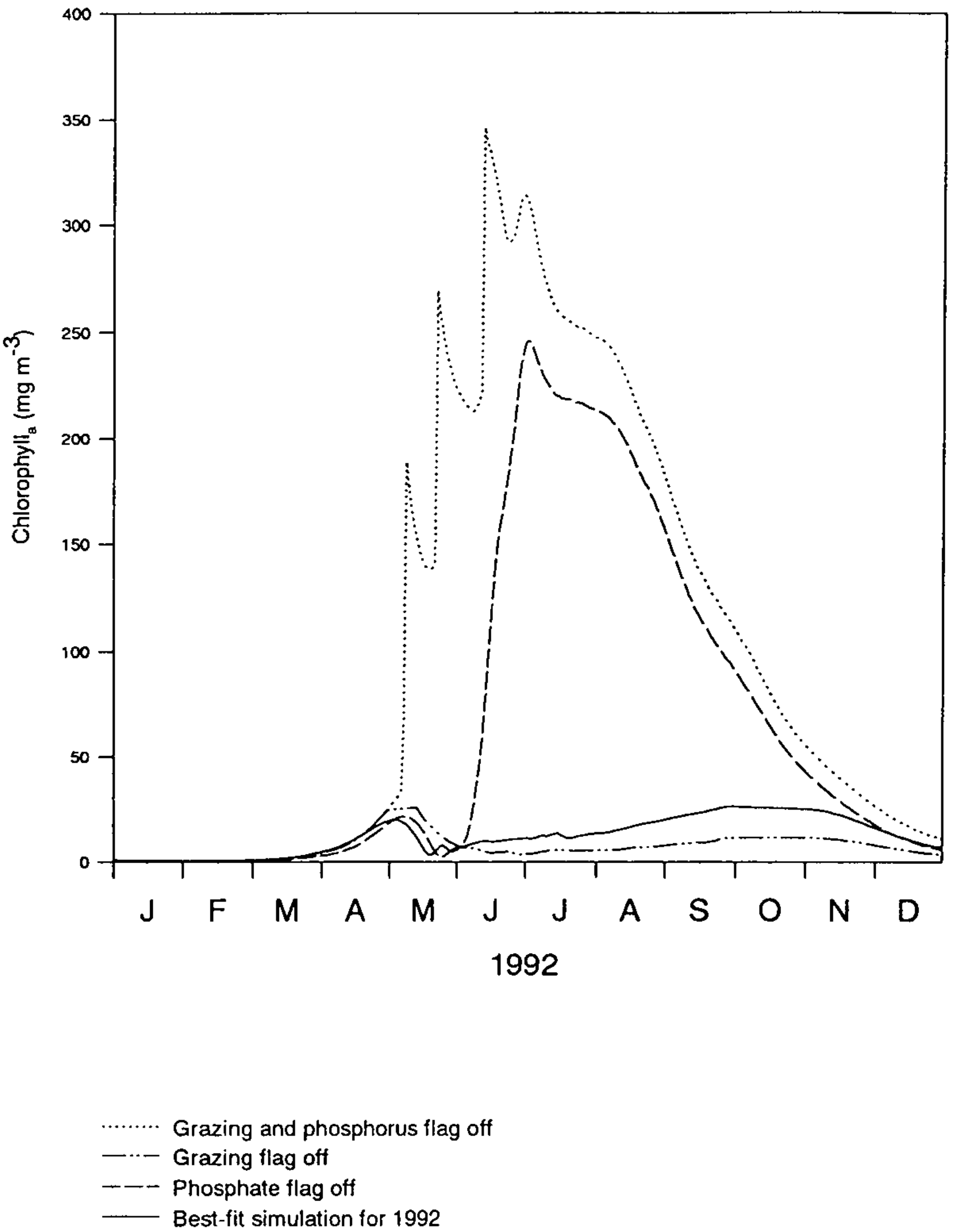


Figure 22. Predicted nutrient and chlorophyll$_a$ concentrations in Rutland Water for 1992 assuming no orthophosphate reduction in the inflows. The results are compared to the simulation giving the best fit to the observed data for 1992.

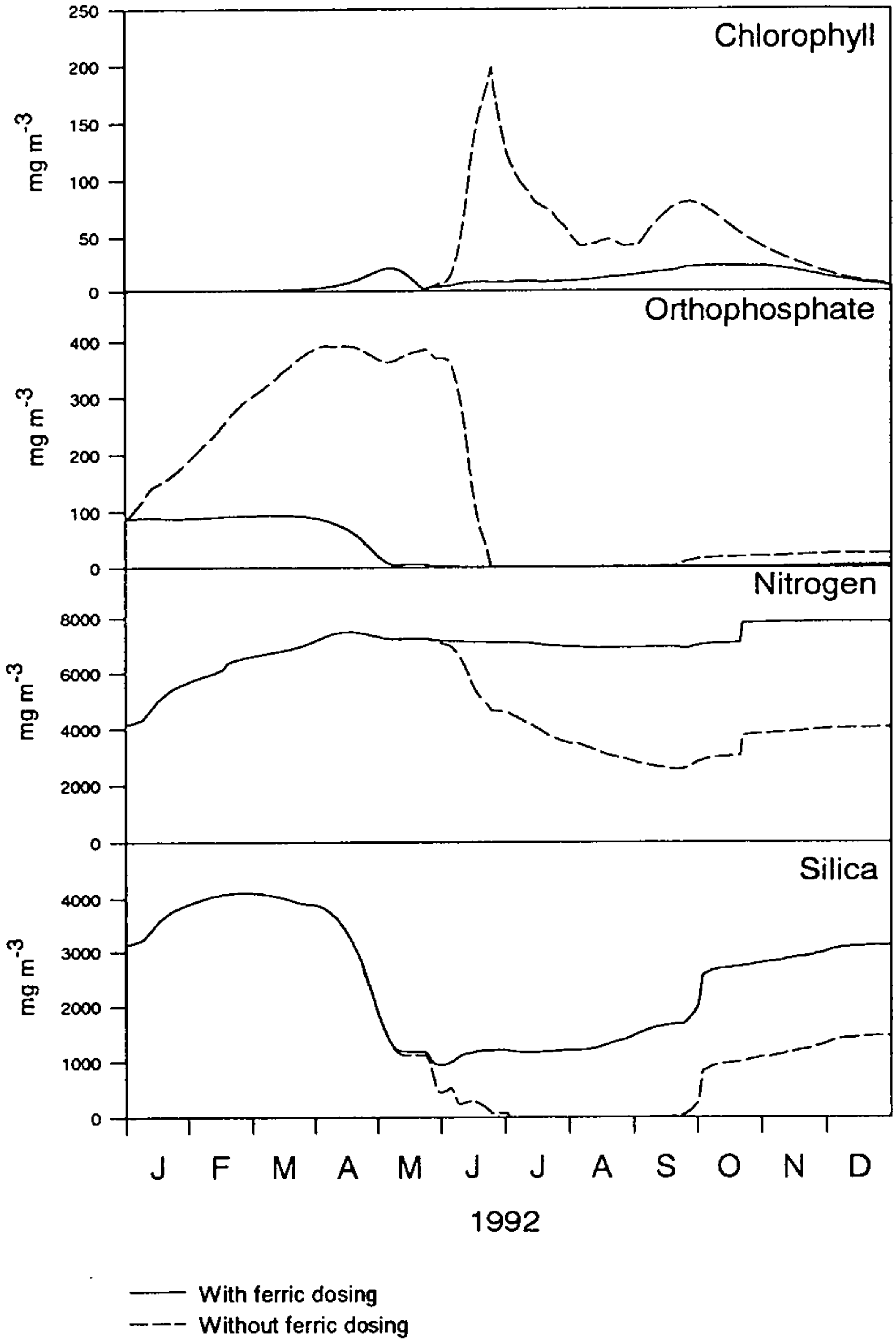


Figure 23. Predicted abundance of each algal group (expressed as chlorophyll$_a$) in Rutland Water during 1992 assuming no orthophosphate reduction in the inflows. The results are compared to the simulation giving the best fit to the observed data for 1992.

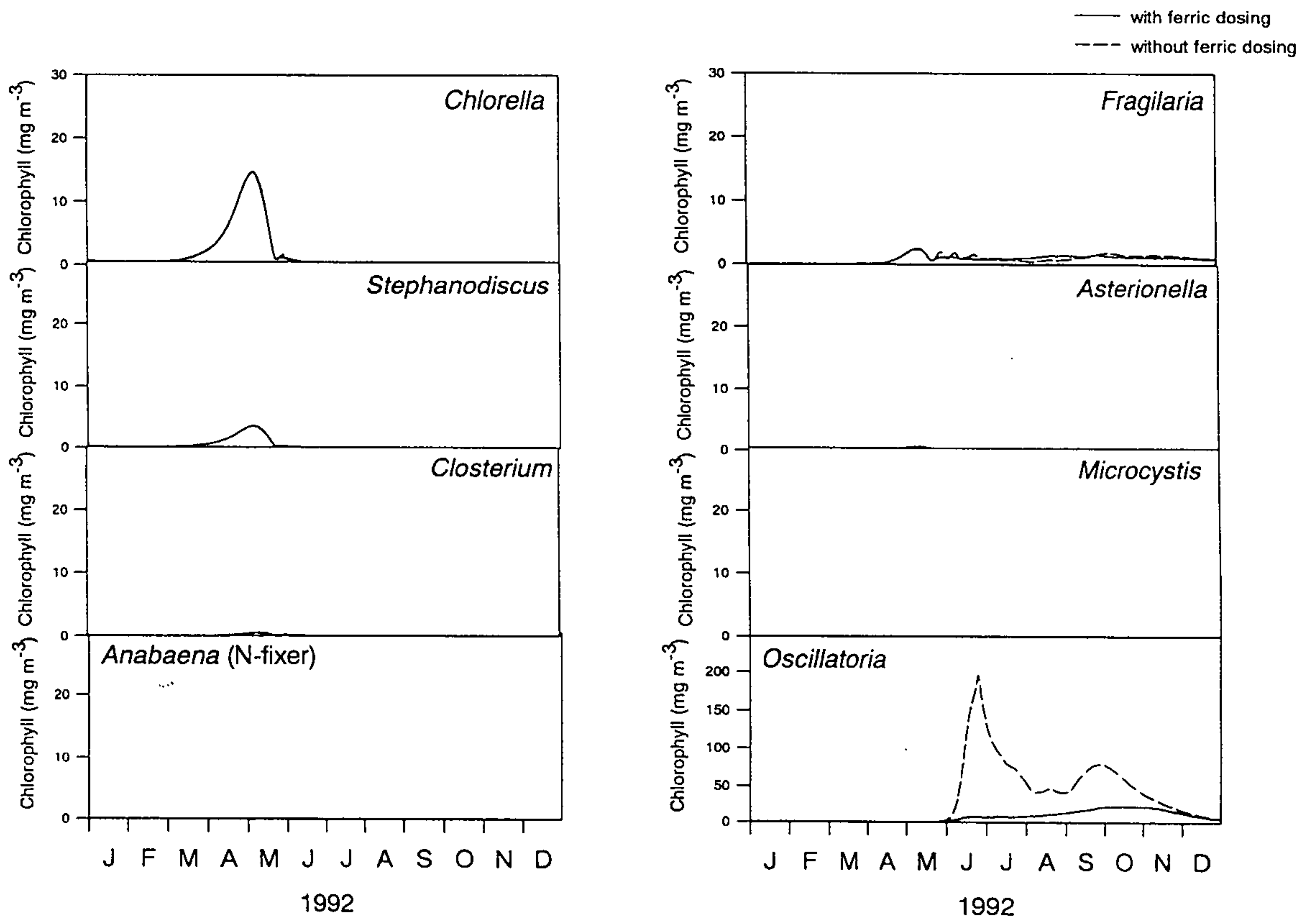


FIG23.SPW

Figure 24. Predicted nutrient and chlorophyll, concentrations in Rutland Water during 1992 assuming a 90% reduction in orthophosphate loading from the Welland and Nene due to ferric dosing and a 50% reduction in orthophosphate in the River Nene. The results are compared to the simulation giving the best fit to the observed data for 1992.

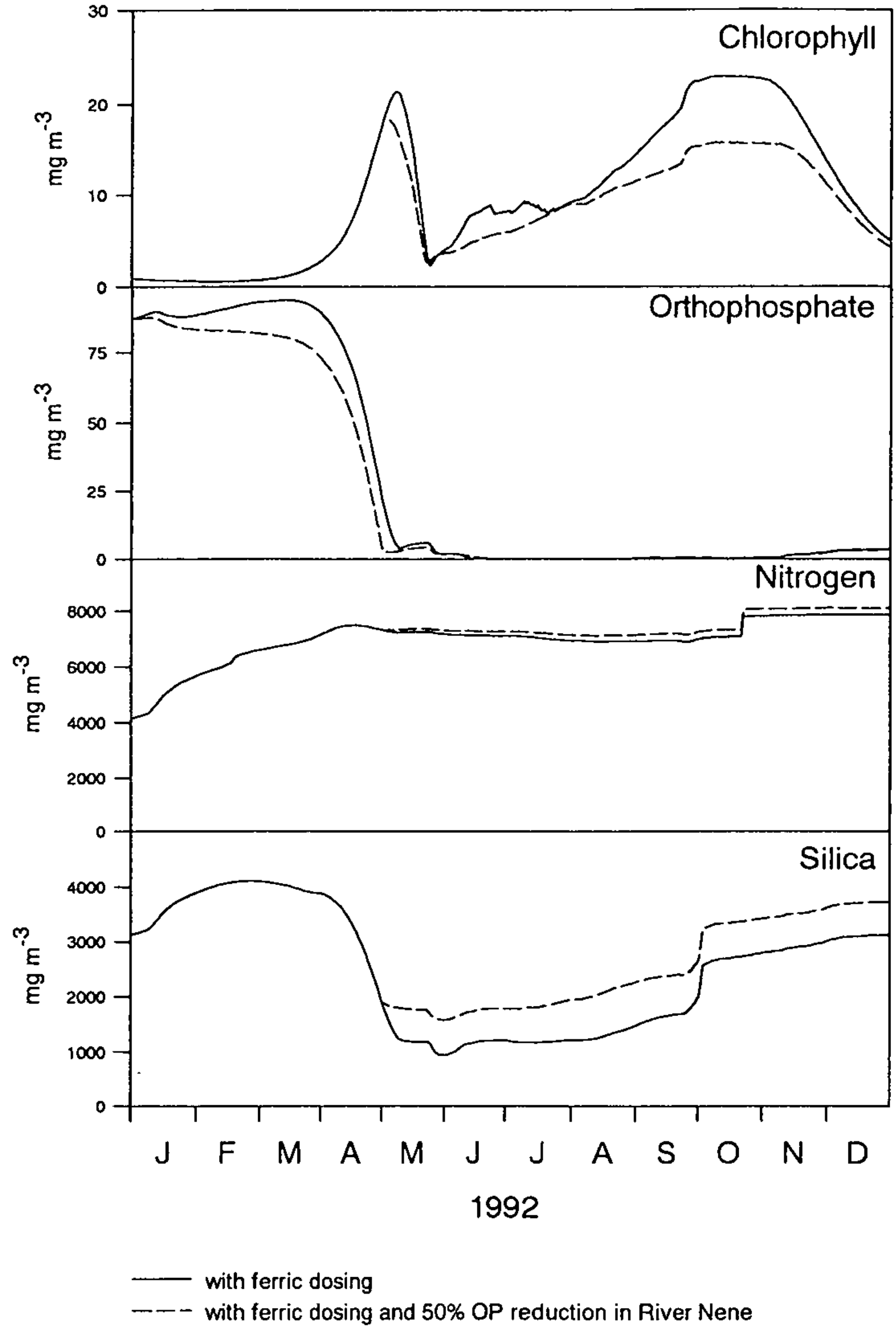


Figure 25. Predicted abundance of each algal group (expressed as chlorophyll$_a$) in Rutland Water during 1992 assuming a 90% reduction in orthophosphate loading from the Welland and Nene due to ferric dosing and a 50% reduction in orthophosphate in the River Nene. The results are compared to the simulation giving the best fit to the observed data for 1992.

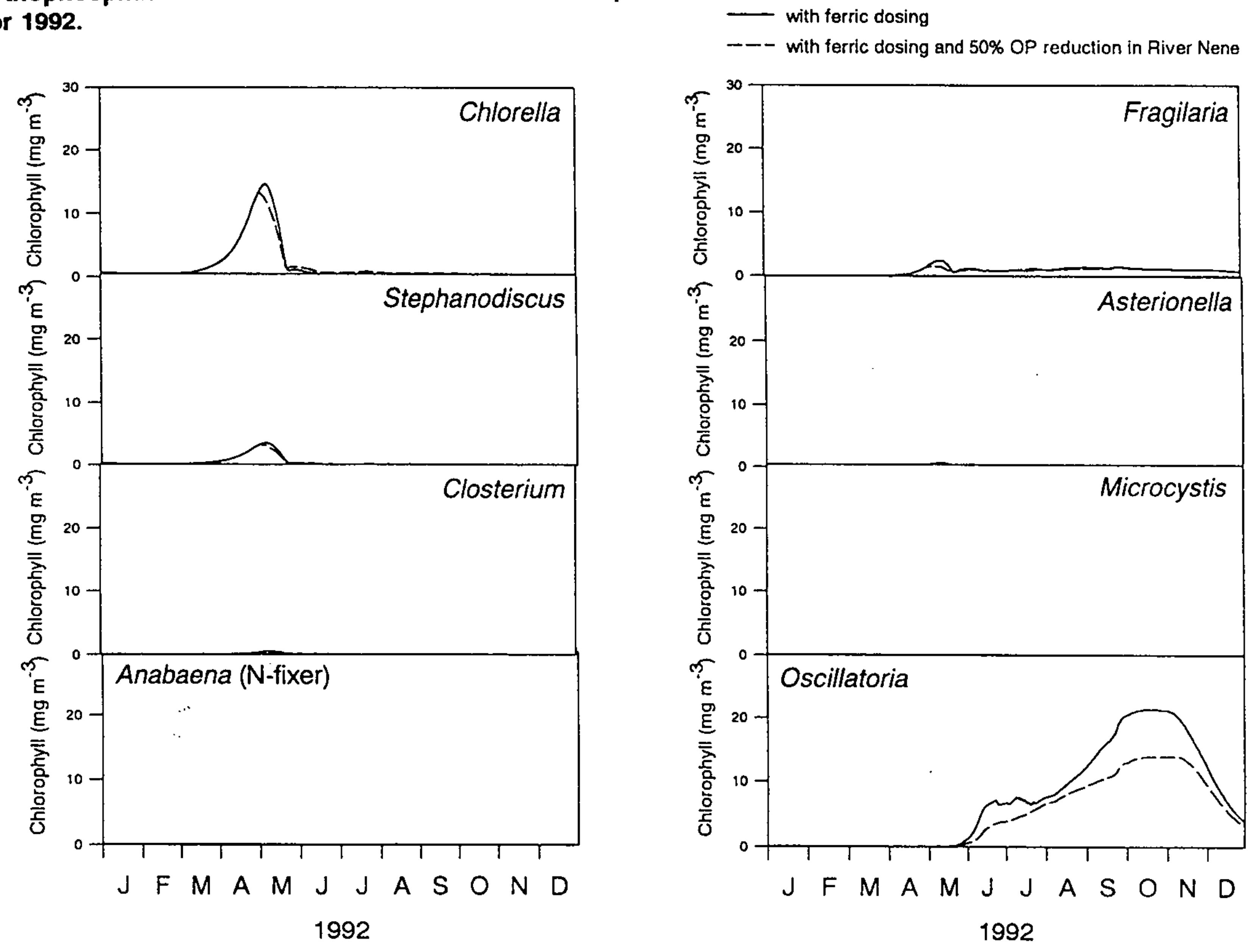


Figure 26. Predicted nutrient and chlorophyll$_a$ concentrations in Rutland Water during 1992 assuming only a 50% reduction in orthophosphate in the River Nene. The results are compared to the simulation giving the best fit to the observed data for 1992.

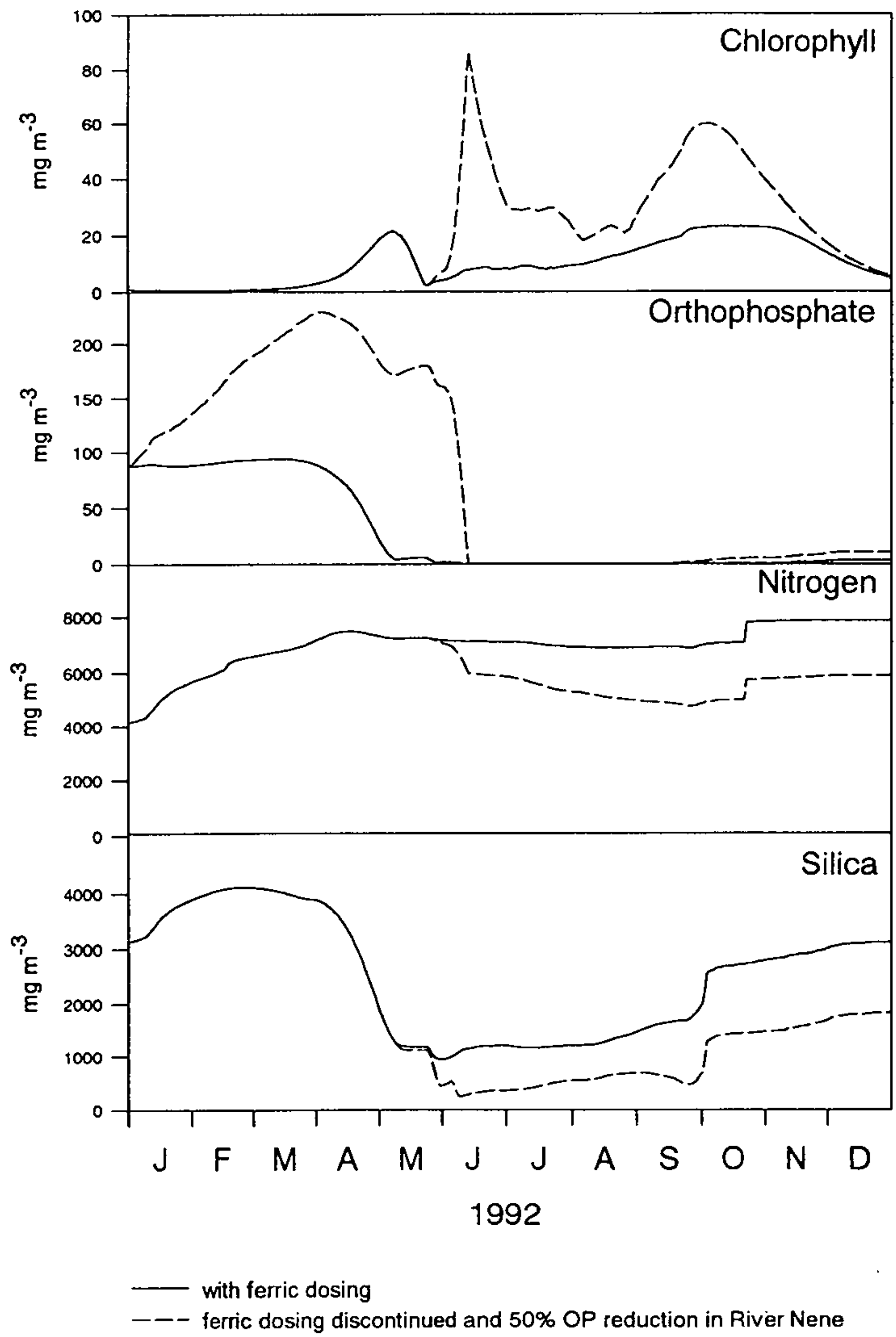


Figure 27. Predicted abundance of each algal group (expressed as chlorophyll$_a$) in Rutland Water during 1992 assuming only a 50% reduction in orthophosphate in the River Nene. The results are compared to the simulation giving the best fit to the observed data for 1992.

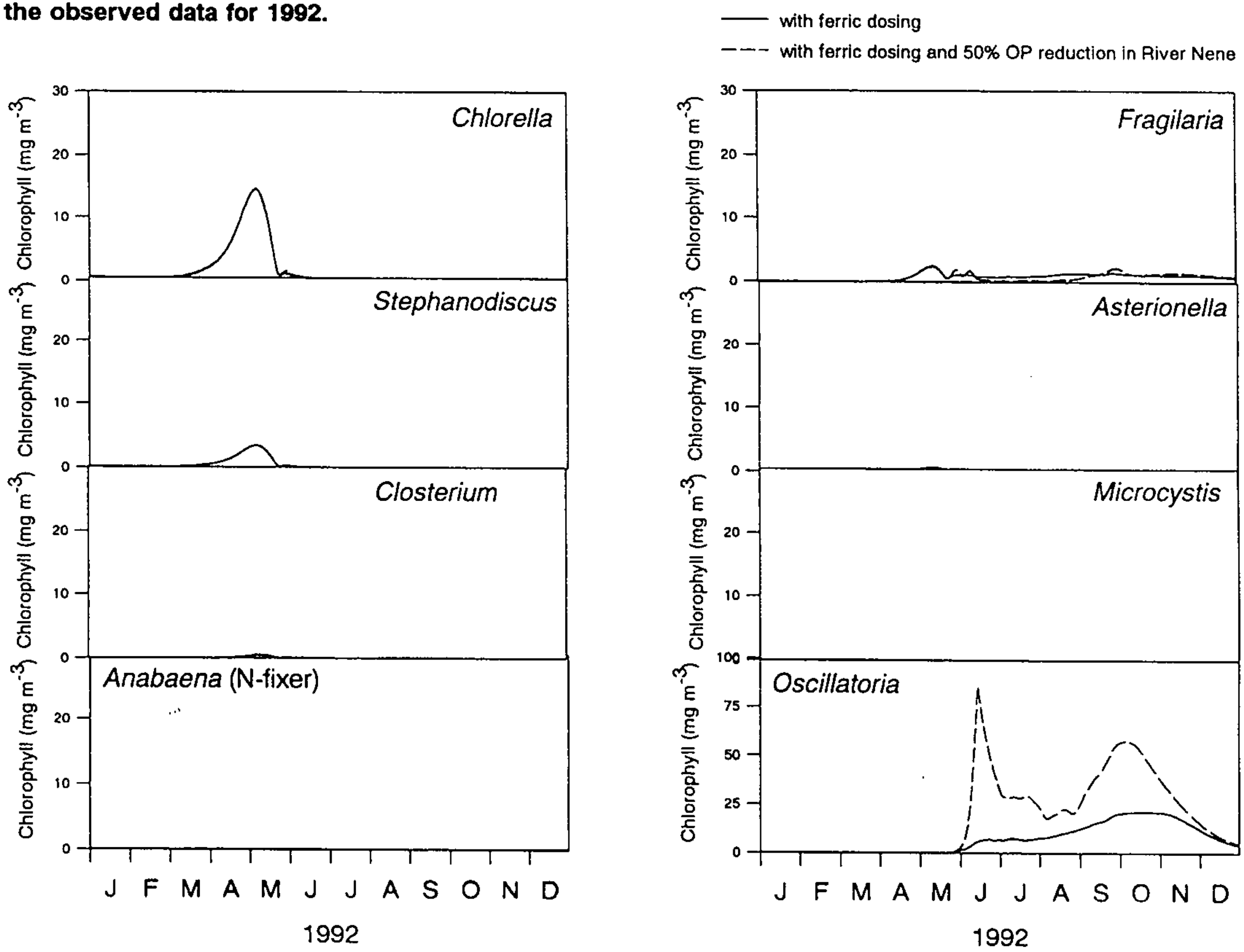
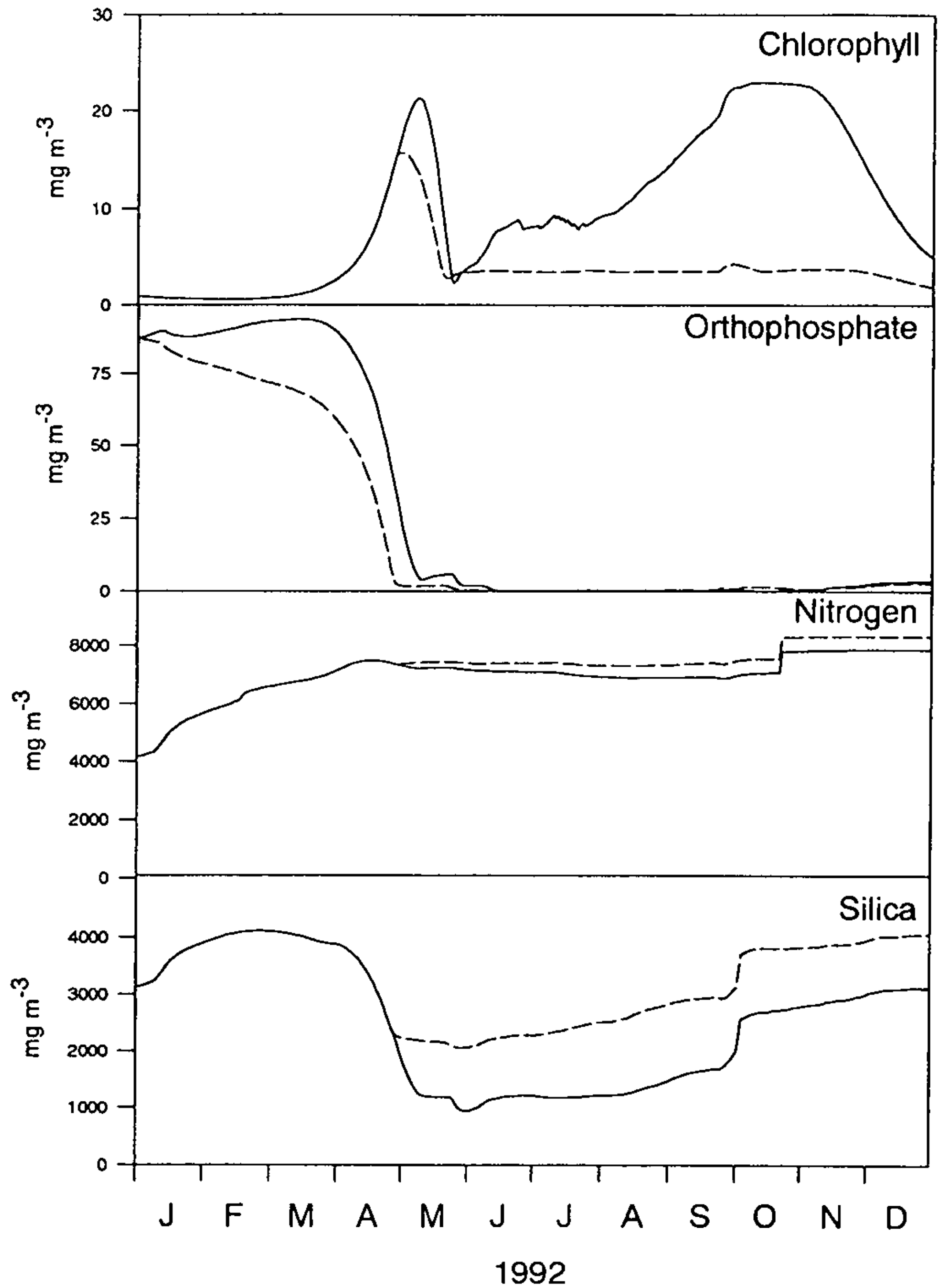


Figure 28. Predicted nutrient and chlorophyll$_a$ concentrations in Rutland Water during 1992 assuming that the orthophosphate concentrations in the Rivers Welland and Nene are reduced to 10 mg m^{-3}. The results are compared to the simulation giving the best fit to the observed data for 1992.



—— with ferric dosing
– – – ferric dosing discontinued;OP concentrations in Rivers Welland and Nene reduced to 10 mg m^{-3}

Figure 29. Predicted abundance of each algal group (expressed as chlorophyll$_a$) in Rutland Water during 1992 assuming that the orthophosphate concentrations in the Rivers Welland and Nene are reduced to 10 mg m^{-3}. The results are compared to the simulation giving the best fit to the observed data for 1992.

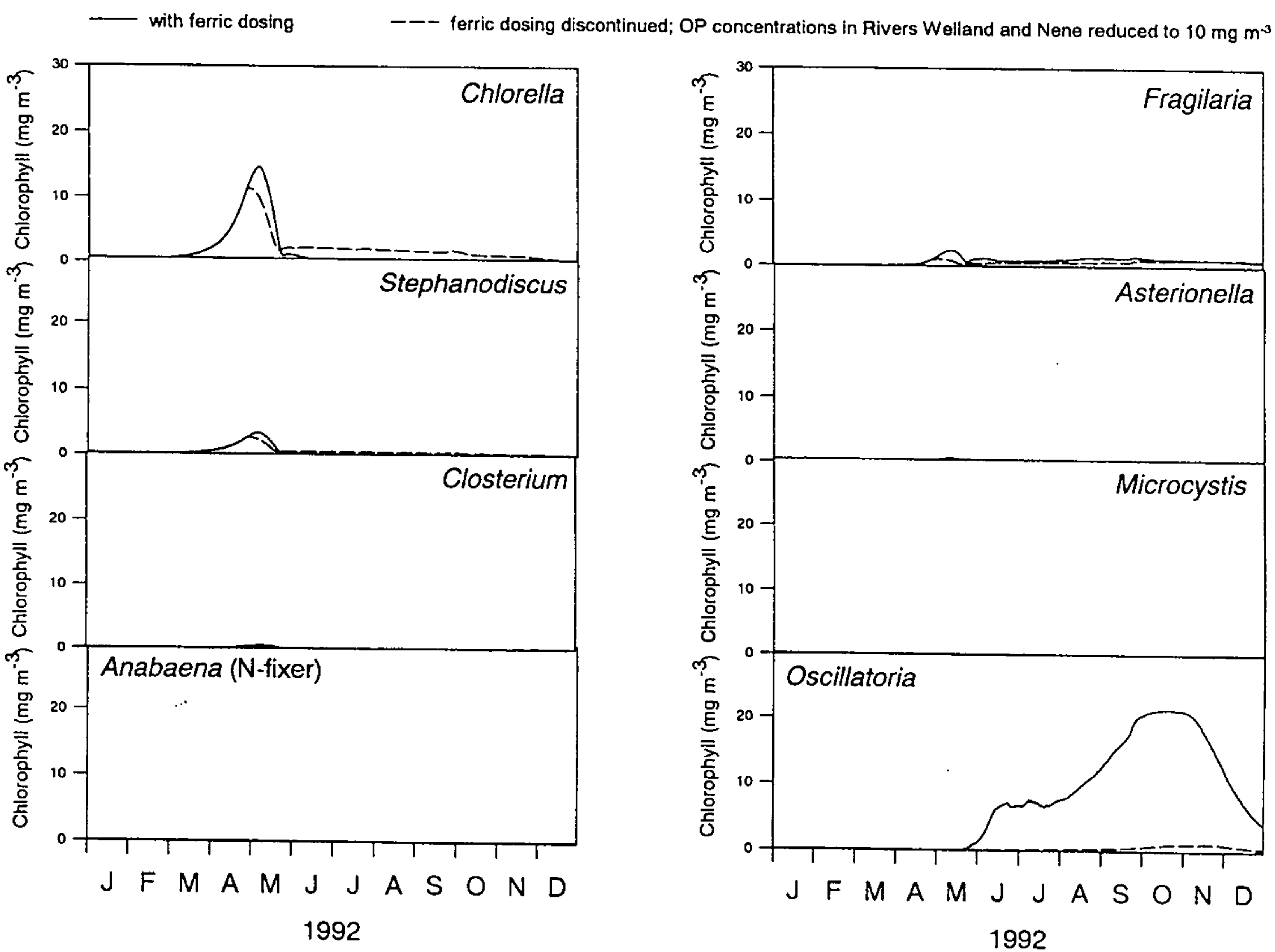


Figure 30. A comparison of the predicted chlorophyll$_a$ concentrations in Rutland Water which would have prevailed in 1992 under 5 different phosphorus management strategies.