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Broads Research Progress Report 1991

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NRA

National Rivers Authority

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1. INTRODUCTION

The National Rivers Authority is carrying out a series of research projects aimed at the restoration of the Norfolk Broads. These projects are funded as part of the Anglian Region Operational Investigation Programme, with contributions to funding made by the Broads Authority[1] and English Nature. Although each project has specific aims they all contribute to the restoration strategy and each depends on the others for information. For these reasons this report deals with the following Projects.

540 Control of Phosphorus in Broadland
541 Control of sediment release of phosphorus by Iron Dosing
528 The use of Biomanipulation

1.1 Pollution as a Problem in the Broads.

In comparison with their condition at the end of the 19th century the Broads now suffer from the effects of pollution in the form of nutrient enrichment. Although a systematic study of the Broads only began in the 1950s it is clear from a wealth of photographic, anecdotal and naturalists' records, backed by palaeolimnological work that these lakes once contained a much more diverse and abundant biota. At this time the Broads were clear water lakes with several species of aquatic plants and associated invertebrate animals. The water in the area is calcareous and the alkalinity generated, combined with the habitat structure provided by the diverse plant life, provided a naturally productive lake resulting in numerous plant and fish eating birds and a prolific fishery.

Since the 1950's the majority of these lakes have become dominated by phytoplankton. This created turbid water and as a result the once abundant submerged aquatic plants disappeared. Without these aquatic plants the larger aquatic invertebrates such as snails and insects do not occur, resulting in less food for adult fish and a generally lower fish stock dominated by small young fish of less value to anglers. In addition many of the herbivorous waterfowl are no longer found in great numbers and thus the area is perceived by both conservationists and the general public as requiring urgent remedial action. Cyanobacteria (blue-green algae) dominate many of the Broad's phytoplankton during the summer and further impetus to restorative work has been provided by the recent realisation that they pose a significant health risk, particularly where immersion sports such as dingy sailing are involved.

1.2 The Restoration Programme ---- Initial 'Chemical Approach'

Research carried out in the 1970's demonstrated that these changes had taken place as a result of enrichment by nitrogen and phosphorus caused by the development of agriculture and expanded sewerage systems of a growing population. The Broads had changed from diverse plant dominated lakes to hypertrophic phytoplankton dominated systems, driven by the input of nutrients particularly phosphorus. These conclusions resulted in an attempt at the end of the 1970's to resolve the problem by reducing the input of phosphorus from sewage effluents. At the time this approach had not gained wide acceptance within the United Kingdom and the initiative was restricted to one river catchment. However by 1986 most major sewage discharges to the more important northern Broads area (Rivers Ant, Bure & Thurne) had phosphorus removal as part of the treatment process.

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[1] Broads Authority funding includes contribution from Soap & Detergent Industry Association.

Although it was expected that recovery of these lakes would take several years, a decade after effective phosphorus controls had been instigated very little biological change had occurred. The lakes were still plankton dominated and despite considerable efforts to re-stock the broads with a range of aquatic plants none of the introductions had survived.

Part of the problem was thought to be the large reservoir of nutrients held in the sediment of these lakes and rivers. During the summer period decomposition close to the sediment water interface generates conditions which allow phosphorus and nitrogen (as ammonium ions) to enter the productive water column maintaining phytoplankton growth. In many of the lakes the internal supply of phosphorus is an order of magnitude greater than that received from the catchment. Thus the 'historic' nutrient load is still enriching the Broads and economic means of controlling this nutrient source is still required. The approach being investigated is the addition of Ferric Chloride by injection to the sediment.

1.3 Development of the Restoration Programme A Systemic Approach: Biomanipulation

During the 1980's observations carried out on a number of isolated lakes began to reveal that other non-chemical factors were important in controlling the type of community found in these lakes. Several examples were found where clear water, with little phytoplankton occurred despite high nutrient levels. This was due to large populations of herbivorous zooplankton which were able to exist because, for a variety of reasons, planktivorous fish were not present in the lake. These observations backed by a series of experiments carried out in small ponds (Irvine, Moss & Balls 1989) demonstrated that both clear water plant dominated lakes and turbid phytoplankton dominated lakes could exist at the same nutrient concentrations.

From these observations it became clear that feedback mechanisms operate in lakes which tend to stabilise their existing community structure. In weed-less, plankton dominated lakes, numerous young fish are produced in response to abundant planktonic food. The feeding pressure of these young fish results in a reduction of the population of large herbivorous zooplankton, which in turn allows maximal phytoplankton growth. In contrast, in a shallow, clear water, weed dominated lake, planktivorous fish populations are reduced, as a result of a smaller food supply and more effective predation by piscivores such as Pike (Esox lucius). The weeds provide cover and larger herbivorous zooplankton are able to survive. By filtration of the water these animals are able to assist in the reduction of phytoplankton, maintaining clear water and promoting the growth of aquatic plants.

These mechanisms have now been widely recognised and have resulted in the concept of Bio-manipulation (The promotion of grazing by zooplankton by reduction of planktivorous fish) as a potential tool for lake restoration. Its importance lies in the ability to move a lake from a stable plankton dominated system to one containing clear water, capable of sustaining aquatic plants. The effectiveness of the system is however still under debate, it is still not certain under what range of nutrient conditions it will be effective and little is known about how to stabilise the system. Available evidence suggests that although bio-manipulation can create clear water at high nutrient concentrations the resulting system is unstable and can quickly revert.

1.4 The Current Restoration Strategy for the Broads

The restoration of the Broads can no longer be seen as a simple process of reducing nutrient supply but rather as a sequence of actions which in combination will produce the desired results. With our current state of knowledge the restoration strategy can be seen as a number of steps:

1) The reduction of the phosphorus (limiting nutrient) supply from the catchment by phosphorus removal from effluents or, where appropriate, by the isolation of the lake.

2) The reduction of the internal (sediment) phosphorus supply and the creation of conditions under which the sediment acts as a nutrient sink rather than source.

3) The use of bio-manipulation techniques to change the lake community from plankton to plant dominance by:

- **3.1** Obtaining clear water by the removal of fish
- **3.2** Establishing aquatic plants in the clear water conditions.

1.5 The Implementation of the Restoration Strategy

1.5.1 Control of Phosphorus (Project 540) (section 2)

The removal of phosphorus from sewage and other effluents is continuing in both the Rivers Ant and Bure. The consistency of removal has varied in recent years and progress during 1991 is reviewed in section 2 of this report.

1.5.2 Control of Phosphorus Release from Sediment. (Project 541)

In some of the smaller lakes sediment has been removed by suction dredging and internal phosphorus loads have been substantially reduced. This is an expensive process and before similar action is taken in larger broads greater certainty about its success will be required. In particular it will be important to establish whether other techniques such as chemical fixation are more cost effective or if biomanipulation can be successful even when phosphorus release occurs. A detailed report has been produced on this aspect of the research (Jackson 1992).

1.5.3 Biomanipulation (establishment of clear water) (Project 528)

Cockshoot Broad (Section 3)

Work on bio-manipulation began in 1989 with the removal of the majority of the fish from Cockshoot Broad, an isolated, suction dredged lake. The effects of this on the lake plankton are being investigated and are dealt with in section 3.

Hoveton Great Broad (Section 4)

Isolation of lakes and complete removal of fish is clearly impractical in many of the larger lakes of the Broads and an alternative approach of isolating large (1-2 ha) areas of a broad are now being tested in Hoveton Gt Broad (section 4).

Pound End Hoveton Little Broad (Section 5)

A similar experiment is being conducted in a natural bay of Hoveton Little Broad, Pound End. This lake is positioned further downstream on the River Bure and may be less amenable to biomanipulation. The success of biomanipulation is probably dependent on the edibility of the phytoplankton by grazing cladoceran zooplankton. In the middle and lower tidal river Bure the phytoplankton contains a greater proportion of filamentous cyanobacteria in the phytoplankton. There is evidence that these are not effectively grazed by zooplankton and the effectiveness of this approach in the middle stretches of the river is being tested at this site

Biomanipulation (restoration of aquatic plants) (Section 6)

Despite establishing clear water conditions in Cockshoot Broad there has been very little growth of aquatic plants in the main basin of the lake. The reasons for this are unclear but grazing by coot (Fulica atra) may be an important factor during the recolonisation process. A series of experiments have been established in a number of different sites to examine the effect of grazing on plant establishment.

Indirect Effects of Biomanipulation. (Sections 7 & 8)

The removal of fish has a direct effect on the phytoplankton via the grazing by zooplankton. However there will also be indirect effects that act via the benthos. There is very little information about the effect of fish removal on benthic processes . The release of nutrients from the sediment can be increased by chironomids and the removal of fish may result in larger chironomid populations as a result of reduced predation. Benthivorous fish disturb the upper sediment layers and this may change the nature of the sediment. Without fish the sediment may become more anoxic, filamentous algal mats can form and chironomid larvae can be replaced by oligochaete worms. These algal mats may be detrimental to plant growth and the changes in the sediment will cause changes in the ability of the sediment to retain nutrients.

2. PHOSPHORUS REMOVAL FROM SEWAGE TREATMENT WORKS AND ITS EFFECT ON THE PHOSPHORUS LOAD IN THE RIVER

2.1 Introduction

Monitoring of the final effluent phosphorus concentration from the River Ant and Bure sewage treatment works continued throughout 1991. The sewage treatment works sampled were Briston, Aylsham, Belaugh and R.A.F. Coltishall on the River Bure; and Horning, Stalham, South Repps and Worstead on the River Ant. Samples of the final effluent were collected, using an automatic sampler, at hourly intervals over a seven day period and bulked into a single container. The final effluent was analysed weekly for soluble reactive phosphorus (S.R.P.) and total phosphorus (Total P.).

In order to assess whether phosphorus stripping is having any impact upon the river system, samples are collected at Ingworth Guaging Station, Horstead Mill and Wroxham Rail Bridge on the River Bure. In previous years a weekly bulked sample was collected at each river site, however, in 1991 the sampling frequency dropped to fortnightly and a spot sample has replaced the autosampling method. All of the samples are analysed for phosphorus concentration and suspended solids, whereas chlorophyll a concentration is only determined for Horstead Mill and Wroxham Rail Bridge. Loading values for phosphorus and suspended solids are calculated from the guaged record of flow at Ingworth and Horstead. The flow values for Wroxham Rail Bridge are calculated from catchment area estimates based on the flow recorded at Horstead Mill.

Fortnightly spot samples have continued to be taken at sites along the River Bure and from the Broads. The phosphorus cycles at Belaugh, Wroxham, Ranworth and South Walsham Broads are presented.

2.2 Phosphorus Discharge From The Sewage Treatment Works

2.2.1 River Bure Sewage Treatment Works - Concentration of Phosphorous Discharged to the River

Generally, the annual average phosphorus concentrations recorded during 1991, for the three Anglian Water S.T.W.'s, have remained at the levels achieved during 1990 (Tables 2.1-2).

At both Briston and Belaugh S.T.W.'s the reduction in the annual average concentration of phosphorus during 1991 does not reflect the S.T.W.'s performance throughout the year (Figures 2.1 & 2. 3). The reduction in concentration of S.R.P. and Total P. achieved during the latter part of 1990 was not maintained at either Briston or Belaugh during the spring of 1991. The effectiveness of the phosphorus stripping process improved throughout 1991 although a further increase in phosphorus concentration was recorded at Briston S.T.W. towards the end of the year.

The annual average concentration of S.R.P. and Total P. for Aylsham S.T.W. is higher than the averages recorded for previous years, except 1989 when there was a period of ineffective phosphorus removal. This increase in the annual average concentration is due to a period of less effective stripping during the winter / spring quarter of 1991. An improvement in the reduction of concentration of both S.R.P. and Total P. was recorded throughout the rest of 1991. The reduction in S.R.P. concentration was more marked than that recorded for Total P. (Figure 2.2). This difference is illustrated by an increase

Table 2.1 River Bure, sewage treatment works soluble reactive phosphorus concentration

		mg/l phosphate -P			
		Briston	Aylsham	Belaugh	RAF Colt.
Quarterly					
April-June	1986	8.01	13.31	10.30	-
July-Sept	1986	8.17	13.15	9.77	-
Oct-Dec	1986	1.49	1.60	5.78	-
Jan-March	1987	4.04	1.07	2.70	8.16
April-June	1987	1.81	2.06	2.70	8.78
July-Sept	1987	1.54	2.27	2.81	8.63
Oct-Dec	1987	0.52	0.31	0.75	8.25
Jan-March	1988	0.41	0.61	0.80	3.33
April-June	1988	4.79	1.13	1.38	1.66
July-Sept	1988	3.94	2.64	1.79	4.09
Oct-Dec	1988	0.88	1.04	3.04	3.63
Jan-March	1989	0.88	1.32	1.85	2.60
April-June	1989	1.31	1.61	2.52	5.01
July-Sept	1989	8.11	4.88	3.69	4.40
Oct-Dec	1989	8.92	8.48	4.74	3.21
Jan-March	1990	6.58	2.03	3.13	1.48
April-June	1990	0.84	0.99	1.79	3.77
July-Sept	1990	0.51	0.74	1.14	3.23
Oct-Dec	1990	0.12	1.44	0.43	2.58
Jan-March	1991	2.33	3.31	0.80	7.64
April-June	1991	1.75	2.13	1.55	4.32
July-Sept	1991	0.75	1.49	1.13	3.88
Oct-Dec	1991	1.53	0.97	0.74	3.87
Annually					
1986 mean		6.08	10.00	8.64	-
1987 mean		1.94	1.43	2.24	8.44
1988 mean		2.46	1.35	1.25	2.49
1989 mean		4.72	3.69	3.17	2.57
1990 mean		1.83	1.27	1.56	2.09
1991 mean		1.59	1.98	1.06	4.93

Table 2.2 River Bure, sewage treatment works total phosphorus concentrations

		mg/l Total -P			
		Briston	Aylsham	Belaugh	RAF Colt.
Quarterly					
April-June	1986	-	-	-	-
July-Sept	1986	-	-	-	-
Oct-Dec	1986	1.15	2.35	6.14	-
Jan-March	1987	6.84	2.31	4.03	8.93
April-June	1987	2.59	3.49	3.80	10.57
July-Sept	1987	2.54	2.87	4.90	9.18
Oct-Dec	1987	2.22	1.91	2.85	8.78
Jan-March	1988	1.53	2.18	2.82	4.30
April-June	1988	6.76	2.67	3.11	3.19
July-Sept	1988	4.70	3.76	3.88	2.96
Oct-Dec	1988	1.95	2.63	1.02	1.90
Jan-March	1989	2.04	3.17	3.86	1.38
April-June	1989	2.25	6.04	4.44	3.46
July-Sept	1989	10.40	8.33	5.86	3.31
Oct-Dec	1989	9.98	7.23	6.67	1.97
Jan-March	1990	7.97	3.22	4.43	2.78
April-June	1990	1.83	2.40	2.91	2.71
July-Sept	1990	1.09	2.02	1.59	2.20
Oct-Dec	1990	0.62	2.62	1.03	1.54
Jan-March	1991	3.58	4.33	1.18	8.57
April-June	1991	2.74	3.71	2.41	5.44
July-Sept	1991	1.37	3.23	1.88	5.78
Oct-Dec	1991	2.17	2.95	1.28	4.97
Annually					
1986 mean		1.15	2.35	6.14	-
1987 mean		3.52	2.89	3.92	9.31
1988 mean		3.78	2.80	3.23	3.84
1989 mean		6.02	6.46	5.18	3.90
1990 mean		2.45	2.54	2.41	3.22
1991 mean		2.47	3.56	1.69	6.19

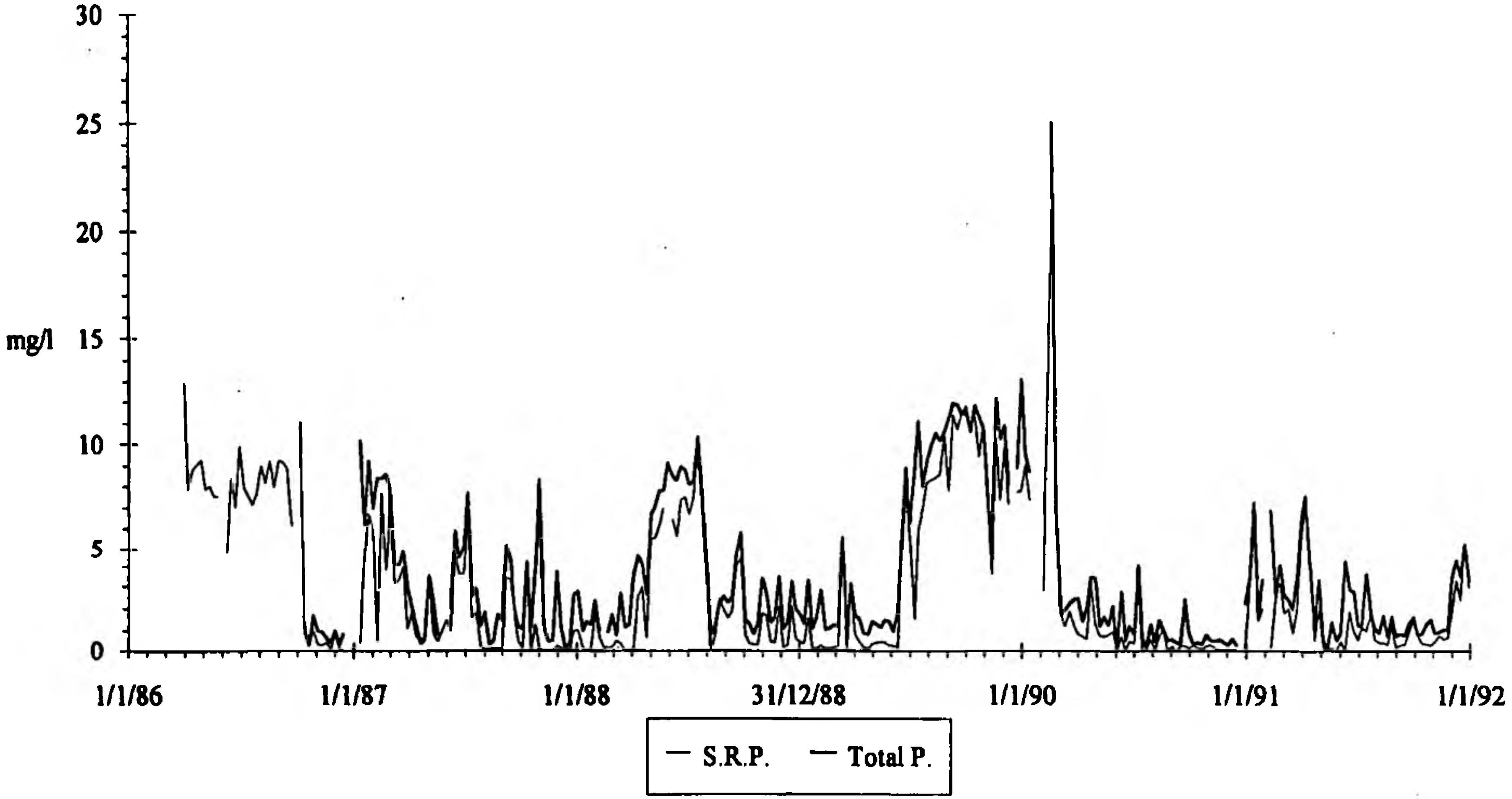


Figure 2.1 Briston Sewage Treatment Works: final effluent phosphorus concentration.

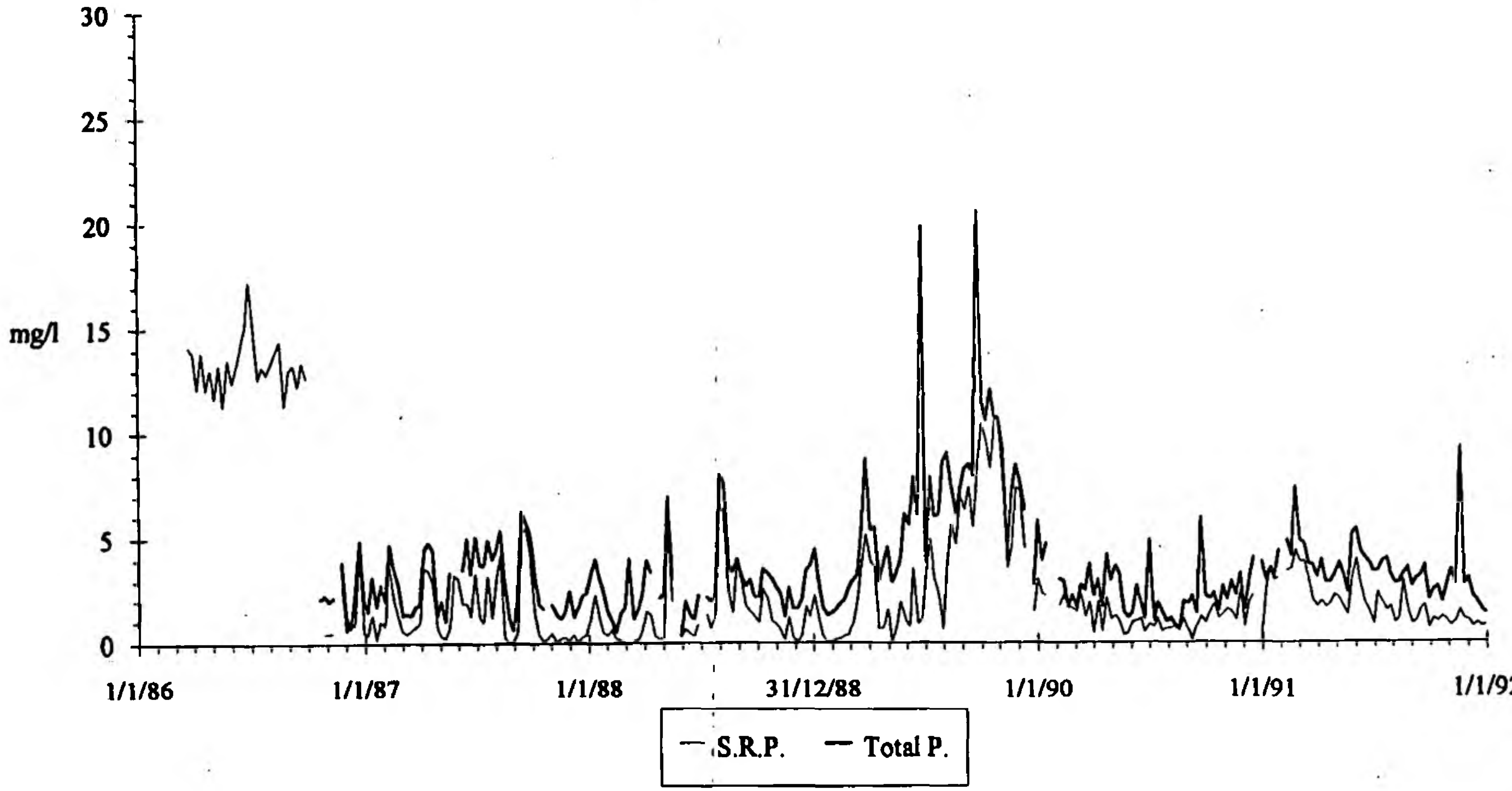


Figure 2.2 Aylsham Sewage Treatment Works: final effluent phosphorus concentration.

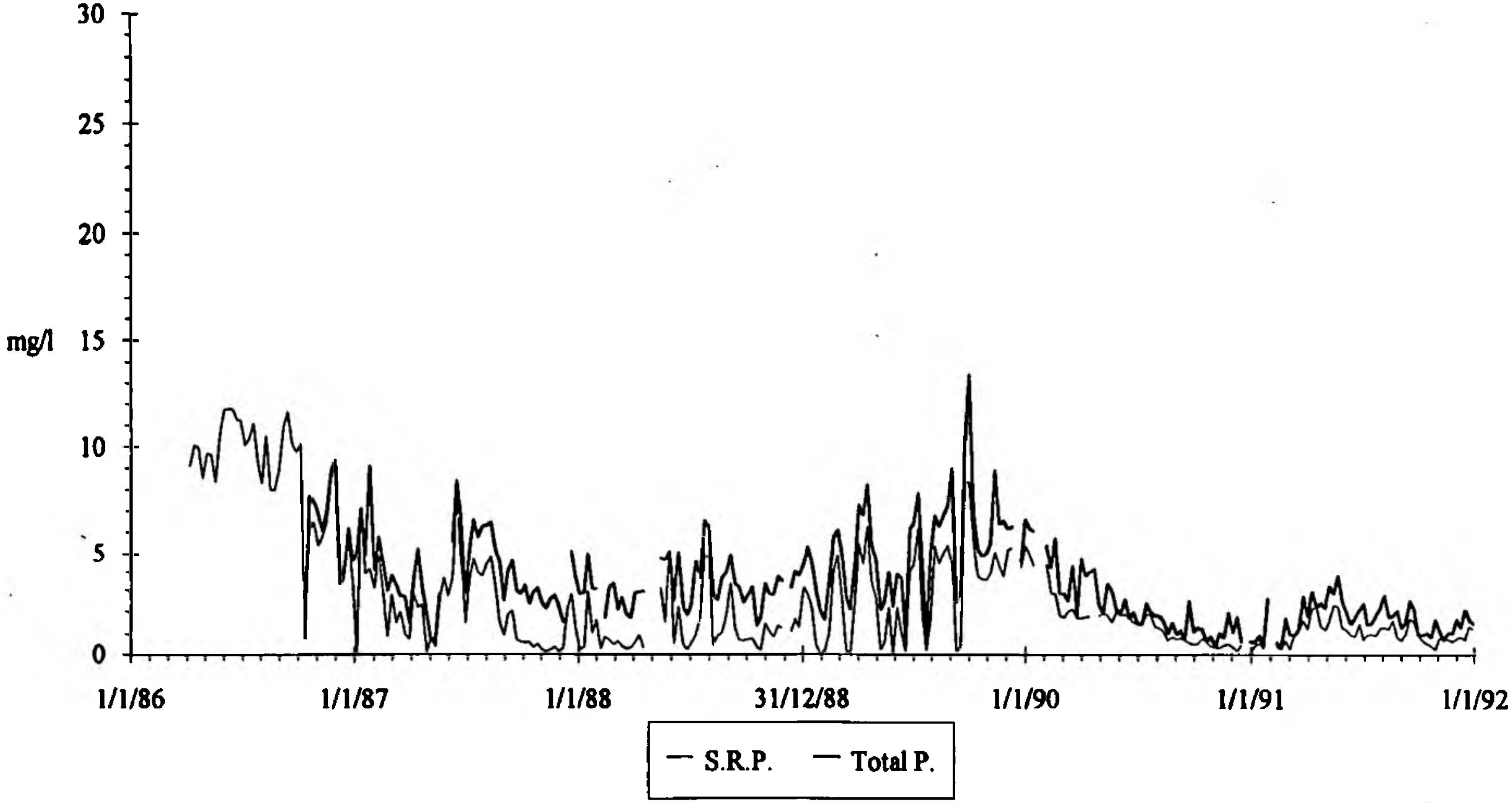


Figure 2.3 Belaugh Sewage Treatment Works: final effluent phosphorus concentration.

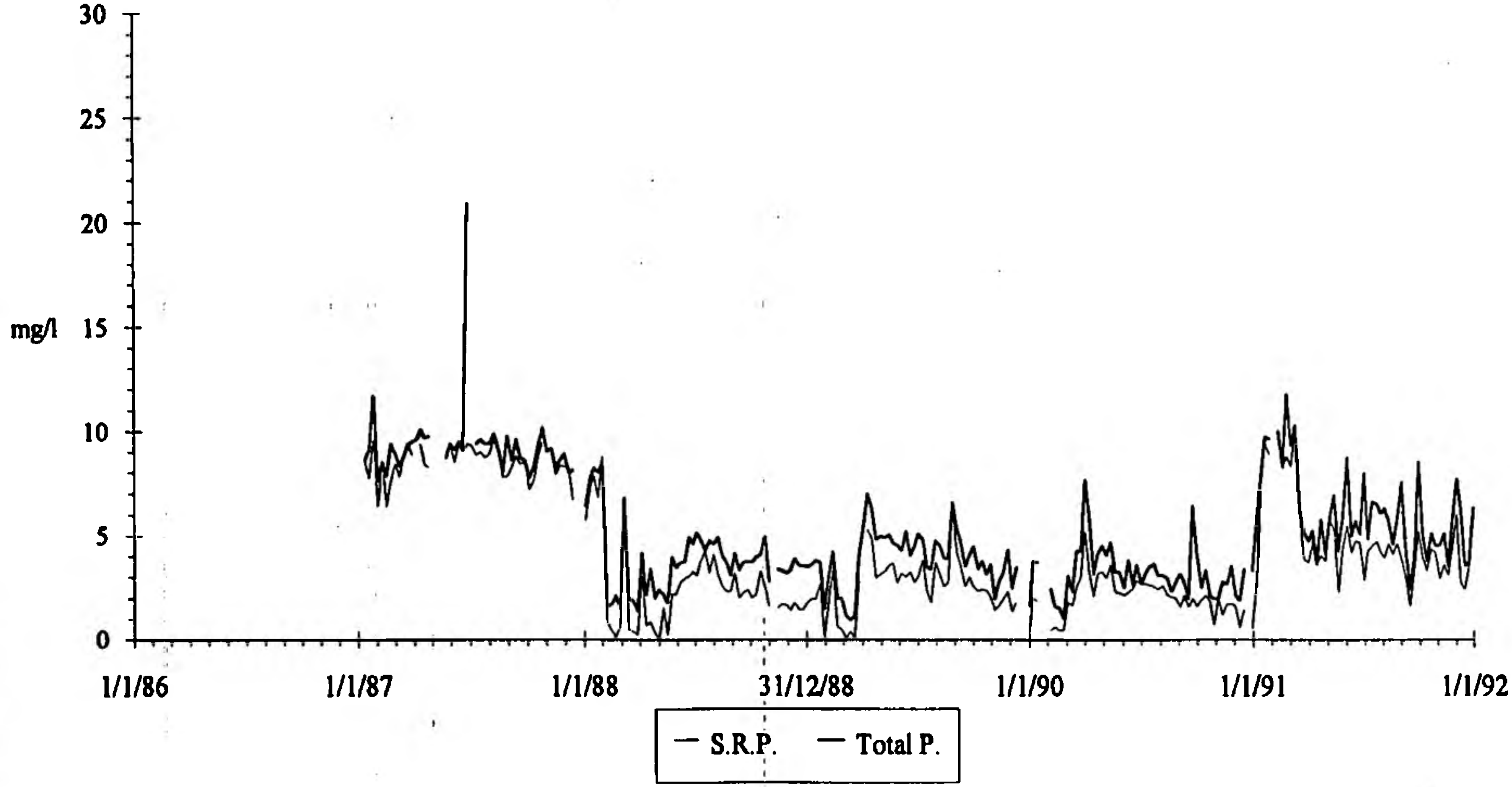


Figure 2.4 R.A.F. Coltishall Sewage Treatment Works: final effluent phosphorus concentration.

Table 2.3 River Ant, sewage treatment works soluble reactive phosphorus concentrations

		mg/l phosphate -P			
		Horning	Stalham	Worstead	South Repps
Quarterly					
April-June	1986	1.64	1.12	-	1.65
July-Sept	1986	4.09	0.72	-	1.16
Oct-Dec	1986	1.56	0.54	-	1.18
Jan-March	1987	1.32	0.49	-	4.42
April-June	1987	2.17	1.20	-	2.90
July-Sept	1987	2.68	0.63	-	1.35
Oct-Dec	1987	1.14	0.30	-	3.40
Jan-March	1988	1.02	0.64	0.50	2.01
April-June	1988	1.81	0.95	2.50	2.32
July-Sept	1988	1.79	1.17	6.02	6.63
Oct-Dec	1988	2.14	0.90	0.50	3.36
Jan-March	1989	1.43	1.29	4.80	3.17
April-June	1989	2.49	0.35	5.82	4.78
July-Sept	1989	3.33	0.72	6.88	6.53
Oct-Dec	1989	1.83	3.29	12.72	6.47
Jan-March	1990	0.44	3.22	5.90	2.32
April-June	1990	1.51	1.69	1.23	1.04
July-Sept	1990	1.09	0.46	0.91	0.94
Oct-Dec	1990	0.57	1.51	0.45	2.32
Jan-March	1991	0.18	0.25	0.72	1.21
April-June	1991	0.44	1.07	3.17	0.83
July-Sept	1991	0.57	0.57	0.38	0.06
Oct-Dec	1991	1.53	0.36	1.78	0.47
Annually					
1986 mean		2.19	0.91	-	1.77
1987 mean		1.84	0.66	-	2.93
1988 mean		2.04	0.92	2.38	3.58
1989 mean		2.28	1.40	7.49	5.21
1990 mean		0.93	1.66	1.97	1.63
1991 mean		0.38	0.56	1.49	0.43

Table 2.4 River Ant, sewage treatment works total phosphorus concentration

		mg/l Total -P			
		Horning	Stalham	Worstead	South Repps
Quarterly					
April-June	1986	-	-	-	2.49
July-Sept	1986	-	-	-	2.80
Oct-Dec	1986	-	-	-	2.76
Jan-March	1987	-	-	-	-
April-June	1987	-	-	-	6.10
July-Sept	1987	3.20	1.12	-	4.24
Oct-Dec	1987	1.61	0.94	-	5.50
Jan-March	1988	2.38	1.76	0.66	4.11
April-June	1988	2.97	2.24	3.20	6.10
July-Sept	1988	3.23	1.41	6.32	7.42
Oct-Dec	1988	2.25	2.52	1.24	6.10
Jan-March	1989	2.82	2.64	6.40	7.26
April-June	1989	4.28	1.32	8.18	7.27
July-Sept	1989	5.18	1.43	9.13	11.40
Oct-Dec	1989	3.11	4.80	14.43	8.80
Jan-March	1990	1.00	4.07	6.03	4.12
April-June	1990	2.45	3.19	2.78	3.81
July-Sept	1990	2.00	1.05	1.80	4.25
Oct-Dec	1990	1.29	2.24	1.33	4.43
Jan-March	1991	0.79	0.68	1.42	3.19
April-June	1991	0.96	1.40	4.49	0.70
July-Sept	1991	1.26	1.33	1.31	0.43
Oct-Dec	1991	2.25	0.97	3.08	1.17
Annually					
1986 mean		-	-	-	3.14
1987 mean		1.77	0.96	-	5.46
1988 mean		3.25	1.98	2.91	6.03
1989 mean		3.86	2.52	9.47	8.71
1990 mean		1.71	2.58	2.66	4.16
1991 mean		1.03	1.09	2.46	1.30

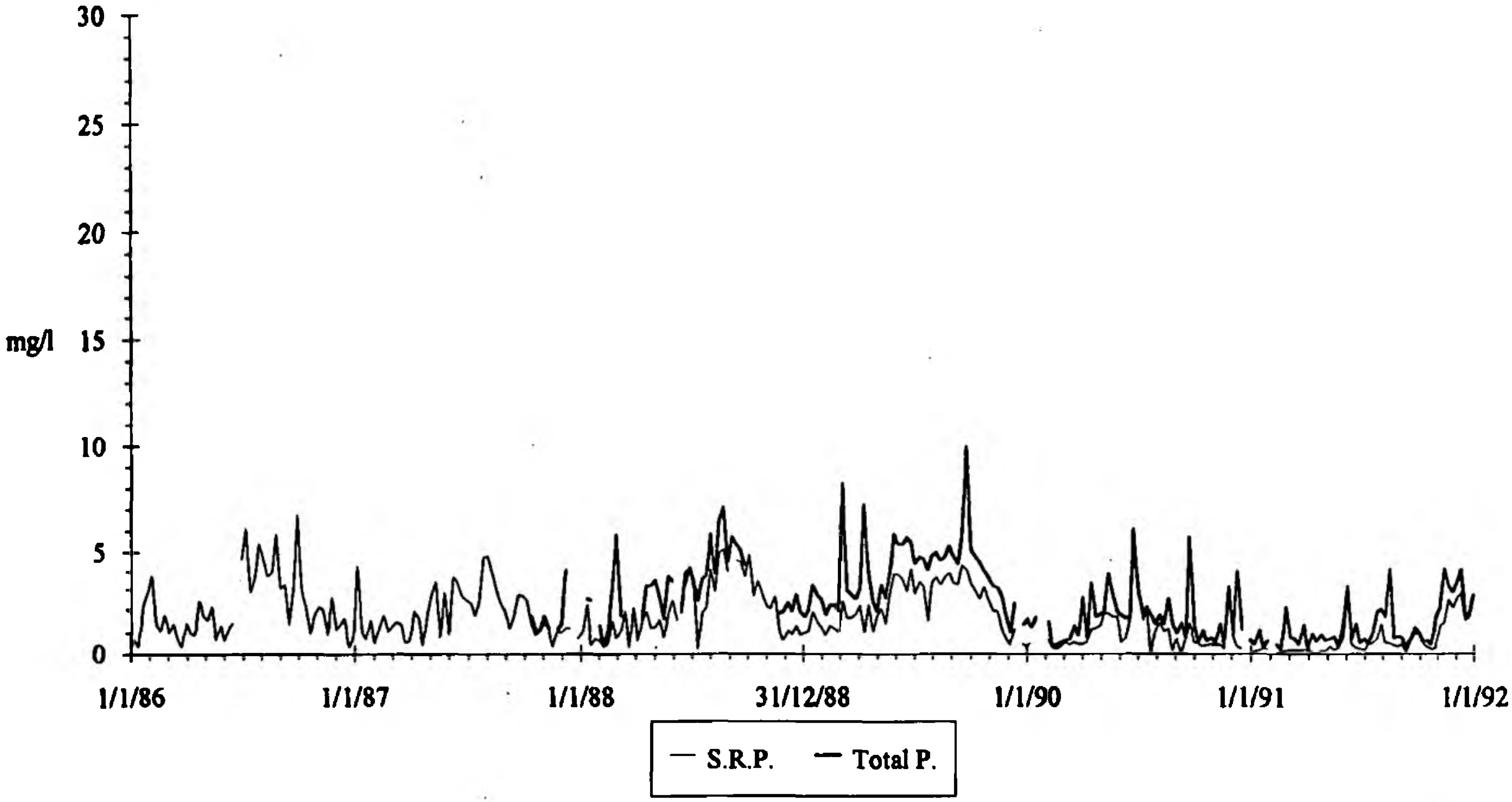


Figure 2.5 Horning Sewage Treatment Works: final effluent phosphorus concentration.

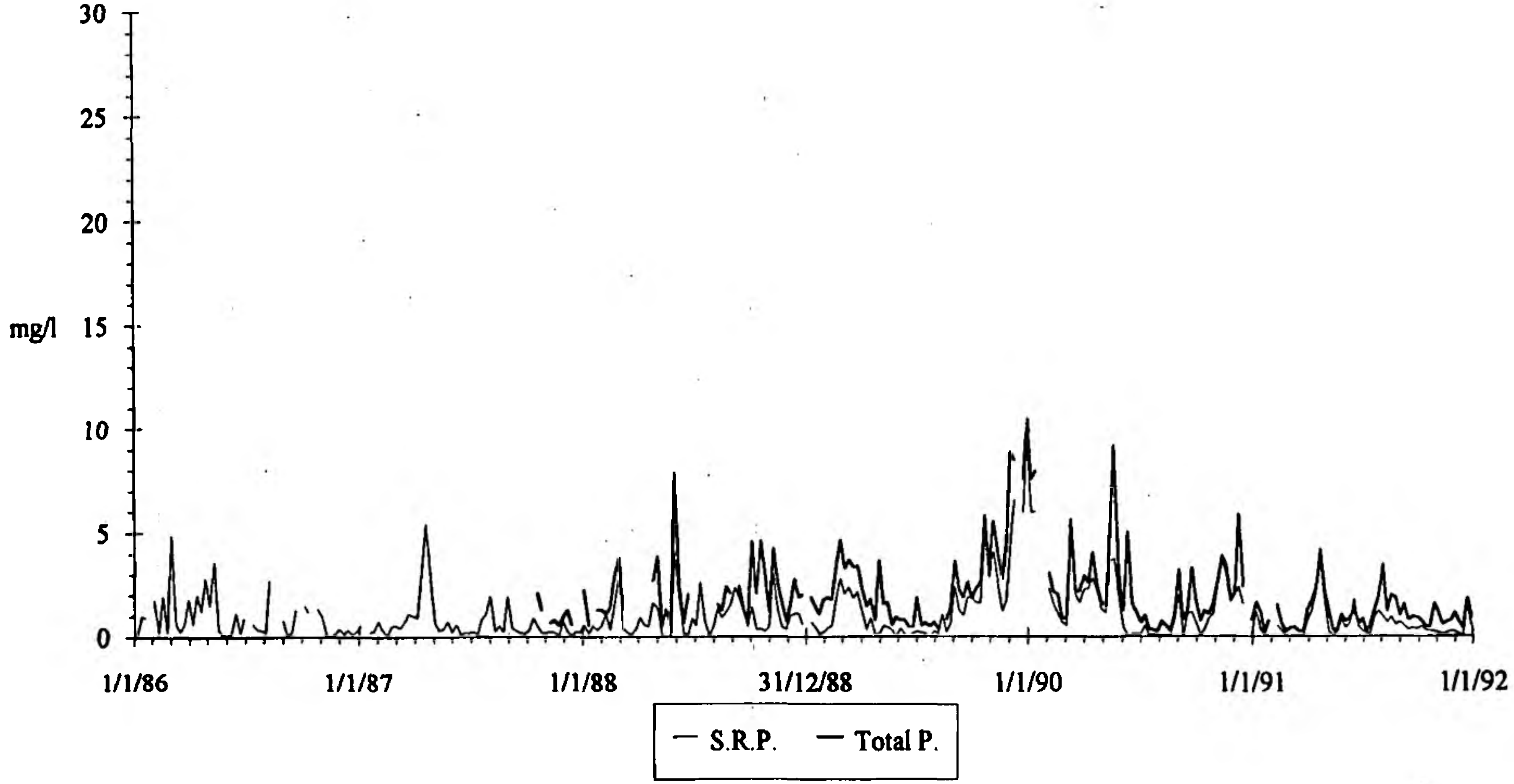


Figure 2.6 Stalham Sewage Treatment Works: final effluent phosphorus concentration.

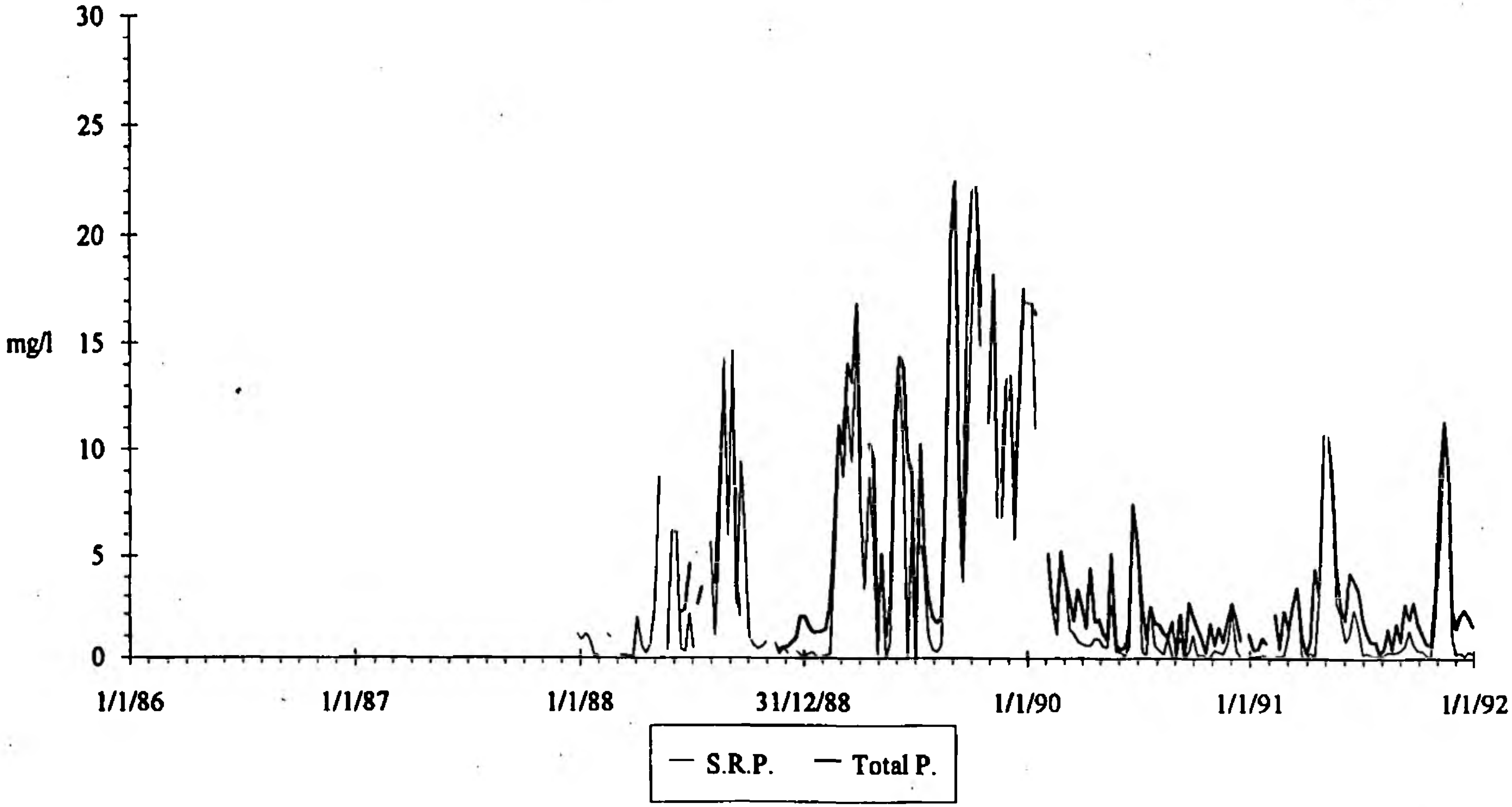


Figure 2.7 Worstead Sewage Treatment Works: final effluent phosphorus concentration.

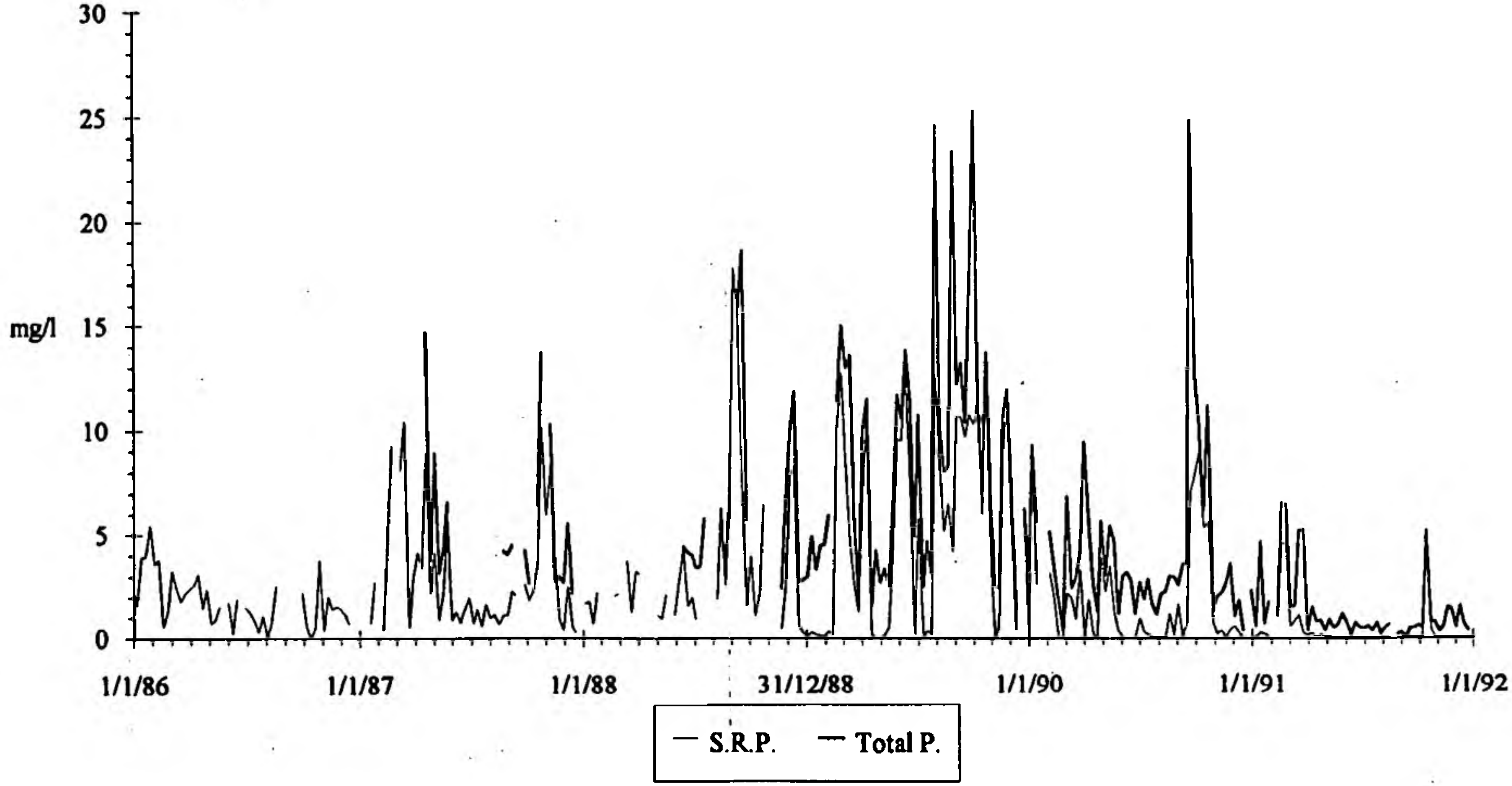


Figure 2.8 South Repps Sewage Treatment Works: final effluent phosphorus concentration.

in the concentration of Particulate Phosphorus during 1991 from a spring quarterly average of 1.02mg/l to a winter average of 1.98mg/l.

An increase in the annual average concentration of phosphorus was also recorded at R.A.F. Coltishall S.T.W.. The spring quarterly average for both S.R.P. and Total P. concentration equalled pre-stripping levels. Figure 2.4 shows this increase in phosphorus concentration and the subsequent decrease during the late spring. However, the concentration does not fall to the levels recorded for previous years. The sudden increase in phosphorus concentration at the beginning of 1991 was due to problems with a faulty pump.

2.2.2 River Ant Sewage Treatment Works - Concentration of Phosphorous Discharged to the River

Further improvement in the reduction of phosphorus concentration was recorded at all of the Anglian Water S.T.W.'s monitored on the River Ant (Figures 2.5-8). The annual averages for S.R.P. and Total P. concentration are lower in 1991 than for any year since stripping was initiated. Tables 2.3-4 show the quarterly and annual averages for S.R.P. and Total P. phosphorus concentration.

At Horning S.T.W. the phosphorus concentration remained below previously recorded levels for the first three quarters of 1991 but increased during the latter winter quarter. This is in contrast to previous years when the concentration of phosphorus at Horning increased during the summer period. The concentration of S.R.P. and Total P., at Stalham S.T.W., increased in the late spring and summer then decreased during the latter part of the year. Phosphorus reduction was largely maintained at Worstead S.T.W. throughout 1991 with the exception of two large fluctuations; one in the late spring and the other in the late autumn. Levels of phosphorus on these occasions reached concentrations obtained during the period of ineffective stripping in 1989. The phosphorus concentration at South Repps S.T.W. has remained low throughout 1991. The weekly fluctuations in concentration are also less pronounced in comparison to those recorded for the years 1986 - 1990.

2.2.3 Phosphorus Load Discharged to the River Bure and Ant

The changes previously described in S.R.P. and Total P. concentration are reflected in the loading values (Tables 2.5-8). The phosphorus loads are calculated using discharge values obtained from Anglian Water, for each of the S.T.W.

The efficiency of the phosphorus stripping process at reducing the load discharged to the River Bure during 1991, in comparison to previous years, is shown in Table 2.9. At Briston and Aylsham between 90-99% of the potential S.R.P. load was removed from the final effluent during the later part of 1990. By the first quarter of 1991 this was reduced to only 75%, however, during the year the percentage removal improved to greater than 90% at both S.T.W.. The percentage of potential S.R.P. removed at Belaugh remained fairly constant throughout 1991 with a decrease to 86% in the spring quarter. At R.A.F. Coltishall the percentage removal fell to just 11% for the first quarter of 1991 and then only rose to just over 50% for the remainder of the year. In previous years up to 80% of the potential S.R.P. loading had been removed.

The phosphorus load entering the River Ant, from the S.T.W.'s monitored, has been maintained at a reduced level during 1991. At Stalham and South Repps the annual

Table 2.5 River Bure, sewage treatment works soluble phosphorus loads

		kg/d phosphate -P			
		Briston	Aylsham	Belaugh	RAF Colt.
Quarterly					
April-June	1986	3.28	14.75	13.01	-
July-Sept	1986	3.35	15.40	12.29	-
Oct-Dec	1986	0.26	1.78	7.62	-
Jan-March	1987	1.65	1.13	3.23	5.59
April-June	1987	0.74	2.51	3.66	5.58
July-Sept	1987	0.63	2.85	4.28	5.45
Oct-Dec	1987	0.21	0.46	1.22	5.25
Jan-March	1988	0.17	0.85	1.69	2.11
April-June	1988	1.97	1.52	2.18	1.06
July-Sept	1988	1.62	3.47	2.27	1.88
Oct-Dec	1988	0.36	1.19	1.21	1.21
Jan-March	1989	0.36	1.65	2.38	0.88
April-June	1989	0.54	2.22	4.41	2.20
July-Sept	1989	3.32	6.85	5.34	2.11
Oct-Dec	1989	3.66	11.35	6.44	1.25
Jan-March	1990	2.70	2.58	4.06	0.94
April-June	1990	0.35	1.15	2.25	1.72
July-Sept	1990	0.21	0.88	1.51	1.40
Oct-Dec	1990	0.05	1.59	0.68	0.98
Jan-March	1991	0.95	3.41	0.96	4.86
April-June	1991	0.71	2.13	1.73	2.74
July-Sept	1991	0.31	1.34	1.25	2.64
Oct-Dec	1991	0.63	0.92	0.87	2.46
Annually					
1986 mean		2.49	11.52	11.02	-
1987 mean		0.79	1.74	3.10	5.37
1988 mean		1.01	1.74	1.84	1.58
1989 mean		1.94	5.40	4.61	1.63
1990 mean		0.75	1.51	2.04	1.33
1991 mean		0.63	1.88	1.21	3.10

Table 2.6 River Bure, sewage treatment works total phosphorus loads

		kg/d Total -P			
		Briston	Aylsham	Belaugh	RAF Colt.
Quarterly					
April-June	1986	-	-	-	-
July-Sept	1986	-	-	-	-
Oct-Dec	1986	0.47	2.59	8.30	-
Jan-March	1987	2.81	2.46	4.94	6.04
April-June	1987	1.06	4.26	5.20	6.72
July-Sept	1987	1.04	4.93	8.13	5.84
Oct-Dec	1987	0.91	2.55	5.08	5.59
Jan-March	1988	0.63	3.21	5.96	2.73
April-June	1988	2.77	3.54	4.37	2.03
July-Sept	1988	1.93	4.97	4.94	2.60
Oct-Dec	1988	0.80	2.98	3.61	2.31
Jan-March	1989	0.84	3.96	4.84	1.66
April-June	1989	0.96	9.08	7.29	3.19
July-Sept	1989	4.19	11.71	8.46	2.80
Oct-Dec	1989	4.09	13.31	9.07	2.04
Jan-March	1990	3.27	4.08	5.83	1.77
April-June	1990	0.75	2.78	3.66	2.40
July-Sept	1990	0.45	2.39	2.08	2.06
Oct-Dec	1990	0.26	2.94	1.59	1.61
Jan-March	1991	1.47	4.39	1.42	5.45
April-June	1991	1.12	3.73	2.67	3.46
July-Sept	1991	0.56	2.87	2.10	3.87
Oct-Dec	1991	0.89	2.90	1.53	3.16
Annually					
1986 mean		-	-	-	-
1987 mean		1.44	3.54	5.88	5.92
1988 mean		1.55	3.66	4.73	2.44
1989 mean		2.48	9.44	7.38	2.48
1990 mean		1.01	3.00	3.19	2.04
1991 mean		0.98	3.42	1.95	3.92

Table 2.7 River Ant, sewage treatment works soluble reactive phosphorus loads

		kg/d phosphate -P			
		Horning	Stalham	Worstead	South Repps
Quarterly					
April-June	1986	0.29	1.11	-	0.22
July-Sept	1986	0.63	0.64	-	0.16
Oct-Dec	1986	0.37	0.51	-	0.16
Jan-March	1987	0.30	0.46	-	0.60
April-June	1987	0.63	1.24	-	0.39
July-Sept	1987	1.79	0.84	-	0.18
Oct-Dec	1987	0.81	0.45	-	0.46
Jan-March	1988	0.75	1.41	0.02	0.34
April-June	1988	0.76	1.28	0.10	0.30
July-Sept	1988	1.86	1.35	0.21	0.97
Oct-Dec	1988	1.51	0.84	0.02	0.51
Jan-March	1989	0.52	1.23	0.21	0.48
April-June	1989	0.81	0.33	0.27	0.72
July-Sept	1989	1.02	0.60	0.31	0.98
Oct-Dec	1989	0.94	2.61	0.67	0.97
Jan-March	1990	0.29	3.14	0.27	0.35
April-June	1990	0.49	1.82	0.06	0.16
July-Sept	1990	0.40	0.48	0.05	0.14
Oct-Dec	1990	0.24	1.53	0.02	0.35
Jan-March	1991	0.06	0.25	0.04	0.13
April-June	1991	0.17	1.05	0.17	0.01
July-Sept	1991	0.22	0.59	0.02	0.01
Oct-Dec	1991	0.62	0.32	0.12	0.07
Annually					
1986 mean		0.42	0.87	-	0.24
1987 mean		0.88	0.75	-	0.40
1988 mean		1.22	1.22	0.09	0.53
1989 mean		0.81	1.16	0.36	0.78
1990 mean		0.36	1.69	0.09	0.25
1991 mean		0.27	0.67	0.08	0.07

Table 2.8 River Ant, sewage treatment works total phosphorus loads

		kg/d Total -P.			
		Horning	Stalham	Worstead	South Repps
Quarterly					
April-June	1986	-	-	-	0.34
July-Sept	1986	-	-	-	0.38
Oct-Dec	1986	-	-	-	0.37
Jan-March	1987	-	-	-	-
April-June	1987	-	-	-	0.82
July-Sept	1987	1.87	1.87	-	0.57
Oct-Dec	1987	1.15	1.25	-	0.74
Jan-March	1988	2.56	3.45	0.03	0.75
April-June	1988	1.23	2.87	0.13	0.71
July-Sept	1988	2.58	1.64	0.22	1.08
Oct-Dec	1988	2.08	2.41	0.04	0.92
Jan-March	1989	1.12	2.44	0.28	1.09
April-June	1989	1.39	1.24	0.37	1.09
July-Sept	1989	1.58	1.21	0.41	1.64
Oct-Dec	1989	1.70	3.71	0.76	1.22
Jan-March	1990	0.61	3.97	0.28	0.62
April-June	1990	1.23	3.36	0.10	0.57
July-Sept	1990	0.87	1.10	0.09	0.64
Oct-Dec	1990	0.53	2.27	0.07	0.67
Jan-March	1991	0.27	0.69	0.08	0.37
April-June	1991	0.36	1.37	0.25	0.11
July-Sept	1991	0.47	1.38	0.07	0.07
Oct-Dec	1991	0.89	0.91	0.20	0.16
Annually					
1986 mean		-	-	-	0.42
1987 mean		1.22	1.32	-	0.74
1988 mean		2.10	2.58	0.11	0.90
1989 mean		1.41	2.13	0.45	1.27
1990 mean		0.74	2.62	0.13	0.63
1991 mean		0.50	1.19	0.14	0.21

Table 2.9 River Bure, sewage treatment works quarterly % phosphate-P removal

		% removal			
		Briston	Aylsham	Belaugh	RAF Colt.
Quarter					
Oct-Dec	1986	82	88	40	-
Jan-March	1987	51	93	75	-
April-June	1987	78	83	71	-
July-Sept	1987	81	81	66	-
Oct-Dec	1987	94	93	91	-
Jan-March	1988	95	94	87	62
April-June	1988	41	90	94	80
July-Sept	1988	52	77	82	66
Oct-Dec	1988	89	92	91	78
Jan-March	1989	89	89	81	84
April-June	1989	84	85	65	60
July-Sept	1989	1	55	58	62
Oct-Dec	1989	0	25	53	76
Jan-March	1990	19	82	68	83
April-June	1990	90	92	82	69
July-Sept	1990	94	94	88	75
Oct-Dec	1990	99	90	95	82
Jan-March	1991	71	77	92	11
April-June	1991	79	86	86	50
July-Sept	1991	91	91	90	52
Oct-Dec	1991	81	94	93	55

loadings of both S.R.P. and Total P. are approximately half those recorded for previous years.

2.3 Phosphorus Load In The River Bure At Key Points.

2.3.1 Phosphorus Concentration

The annual average concentration of phosphorus (S.R.P., Total P. and Particulate P.) and suspended solids are shown in Tables 2.10-11. The concentration of Total P. recorded in the River Bure at all three sites is lower in 1991 than in any other year since phosphate stripping was initiated in 1986. This lowering of phosphorus concentration may well be artificially created by the change in sampling method. However, the concentration of S.R.P. at Ingworth Guaging Station has remained fairly consistent with the levels recorded in previous years. The mean annual S.R.P. concentrations at Horstead Mill and Wroxham Rail Bridge appear to reflect the performance of the phosphate stripping at the S.T.W.'s. The concentrations of S.R.P. at both sampling stations are below those recorded for late 1989 when there was an extended period of ineffective phosphate removal at S.T.W.'s; but are still not as low as those recorded in 1988. A significant reduction in concentration is apparent in the particulate P. fraction and T.S.S. at all three sites. This, again, may well be due to the change in sampling regime. The spot sampling method will not record all the spate occasions that have occurred. It is during such a spate period that there would be an increase in the concentration of the Particulate P. and T.S.S. carried by the river.

Figure 2.9 illustrates the monthly fluctuations in phosphorus concentration at all three river sites. The change in sampling method and frequency clearly removes many of the peaks in Total P. concentration, however, the overall picture is still much the same as previous years. Both Horstead and Wroxham Rail Bridge show a spring decline in S.R.P. concentration due to uptake by an increase in algal productivity.

At both Horstead and Wroxham Rail Bridge the S.R.P. concentration in the river fell after phosphate stripping was initiated in 1986. This lower level was maintained until 1989 when the concentration rose due to a period of ineffective phosphorus stripping at the S.T.W.'s. Following the rise in available phosphorus in the river the chlorophyll a concentrations also increased. The relationship between phosphorus concentration and the change in algal biomass (chlorophyll a) is illustrated in figures 2.10-11. This increase in chlorophyll a concentration is probably also due to the reduction in river flow increasing the retention time. During 1990 the effectiveness of the stripping process at the S.T.W.'s improved and this was followed by a reduction in the river phosphorus concentration. This reduction in the level of available phosphorus is in turn followed by a lowering of the chlorophyll a concentration during 1991. A summary of the chlorophyll a quarterly and annual concentrations can be seen in Table 2.12.

2.3.2 Phosphorus Loads

The mean annual loadings of phosphorus and solids (Tables 2.13-14) also show a significant decline at all three sites. This is probably due to a combination of both the lowering of particulate P. and T.S.S. concentration, as described, and the effect of the reduced river flows in 1991 (Table 2.15). The S.R.P. load in the river at Ingworth Guaging Station and Horstead Mill has remained unaltered in comparison to 1990, whereas, the load at Wroxham Rail Bridge has decreased.

Table 2.10 River Bure, concentration of total and dissolved phosphorus

	mg/l phosphorus -P			mg/l Total -P		
	Ingworth	Horstead	Wroxham	Ingworth	Horstead	Wroxham
Quarterly						
April-June 1986	0.047	0.095	0.092	0.103	0.135	0.170
July-Sept 1986	0.051	0.128	0.158	0.128	0.169	0.262
Oct-Dec 1986	0.036	0.068	0.076	0.114	0.105	0.123
Jan-March 1987	0.037	0.048	0.037	0.125	0.142	0.151
April-June 1987	0.018	0.031	0.027	0.142	0.097	0.094
July-Sept 1987	0.051	0.059	0.050	0.133	0.094	0.163
Oct-Dec 1987	0.044	0.049	0.050	0.159	0.108	0.135
Jan-March 1988	0.028	0.022	0.031	0.224	0.180	0.189
April-June 1988	0.042	0.018	0.020	0.220	0.096	0.098
July-Sept 1988	0.042	0.032	0.012	0.116	0.089	0.150
Oct-Dec 1988	0.041	0.035	0.022	0.132	0.071	0.088
Jan-March 1989	0.035	0.036	0.043	0.203	0.092	0.118
April-June 1989	0.012	0.019	0.035	0.127	0.103	0.120
July-Sept 1989	0.057	0.058	0.106	0.014	0.117	0.151
Oct-Dec 1989	0.044	0.112	0.073	0.167	0.136	0.250
Jan-March 1990	0.050	0.034	0.052	0.188	0.104	0.211
April-June 1990	0.015	0.016	0.013	0.194	0.147	0.113
July-Sept 1990	0.032	0.051	0.057	0.176	0.101	0.285
Oct-Dec 1990	0.046	0.053	0.059	0.145	0.080	0.211
Jan-March 1991	0.031	0.040	0.021	0.091	0.088	0.085
April-June 1991	0.029	0.027	0.019	0.083	0.065	0.101
July-Sept 1991	0.035	0.032	0.046	0.075	0.066	0.132
Oct-Dec 1991	0.050	0.047	0.048	0.109	0.075	0.093
Annually						
1986 mean	0.043	0.097	0.111	0.116	0.139	0.191
1987 mean	0.037	0.047	0.042	0.143	0.108	0.136
1988 mean	0.038	0.029	0.021	0.170	0.111	0.132
1989 mean	0.037	0.055	0.062	0.154	0.112	0.154
1990 mean	0.034	0.038	0.044	0.182	0.109	0.212
1991 mean	0.035	0.035	0.034	0.087	0.071	0.106

Table 2.11 River Bure, concentration of particulate phosphorus and solids

	mg/l Particulate -P.			mg/l T.S.S.		
	Ingworth	Horstead	Wroxham	Ingworth	Horstead	Wroxham
Quarterly						
April-June 1986	0.059	0.043	0.099	9.48	4.26	13.47
July-Sept 1986	0.072	0.044	0.098	6.86	3.56	12.87
Oct-Dec 1986	0.084	0.038	0.047	8.95	3.63	4.62
Jan-March 1987	0.901	0.087	0.115	22.47	11.42	28.33
April-June 1987	0.122	0.068	0.078	30.49	7.81	14.30
July-Sept 1987	0.081	0.030	0.106	9.80	7.14	15.42
Oct-Dec 1987	0.125	0.066	0.084	27.89	13.68	14.19
Jan-March 1988	0.186	0.081	0.163	48.05	33.59	31.93
April-June 1988	0.183	0.076	0.081	44.82	12.17	17.80
July-Sept 1988	0.074	0.054	0.135	10.69	7.07	24.05
Oct-Dec 1988	0.091	0.038	0.063	19.82	4.55	6.93
Jan-March 1989	0.167	0.061	0.081	17.86	6.89	13.52
April-June 1989	0.115	0.083	0.077	10.02	9.31	16.05
July-Sept 1989	0.075	0.055	0.049	7.18	4.83	13.23
Oct-Dec 1989	0.126	0.025	0.174	12.73	3.05	18.92
Jan-March 1990	0.137	0.069	0.163	20.11	11.11	12.64
April-June 1990	0.179	0.131	0.094	9.94	11.19	13.09
July-Sept 1990	0.143	0.049	0.228	8.81	6.84	34.49
Oct-Dec 1990	0.099	0.033	0.152	6.49	2.21	18.71
Jan-March 1991	0.061	0.048	0.064	7.76	5.25	8.12
April-June 1991	0.054	0.038	0.082	5.22	3.09	12.88
July-Sept 1991	0.041	0.034	0.086	3.86	2.16	11.41
Oct-Dec 1991	0.047	0.028	0.030	9.13	2.13	4.19
Annually						
1986 mean	0.073	0.076	0.076	8.41	4.45	9.65
1987 mean	0.106	0.061	0.094	23.49	9.86	17.93
1988 mean	0.131	0.148	0.112	30.89	14.73	20.26
1989 mean	0.116	0.056	0.092	11.23	6.08	15.21
1990 mean	0.149	0.072	0.173	11.96	67.93	21.51
1991 mean	0.051	0.037	0.072	5.95	3.05	9.75

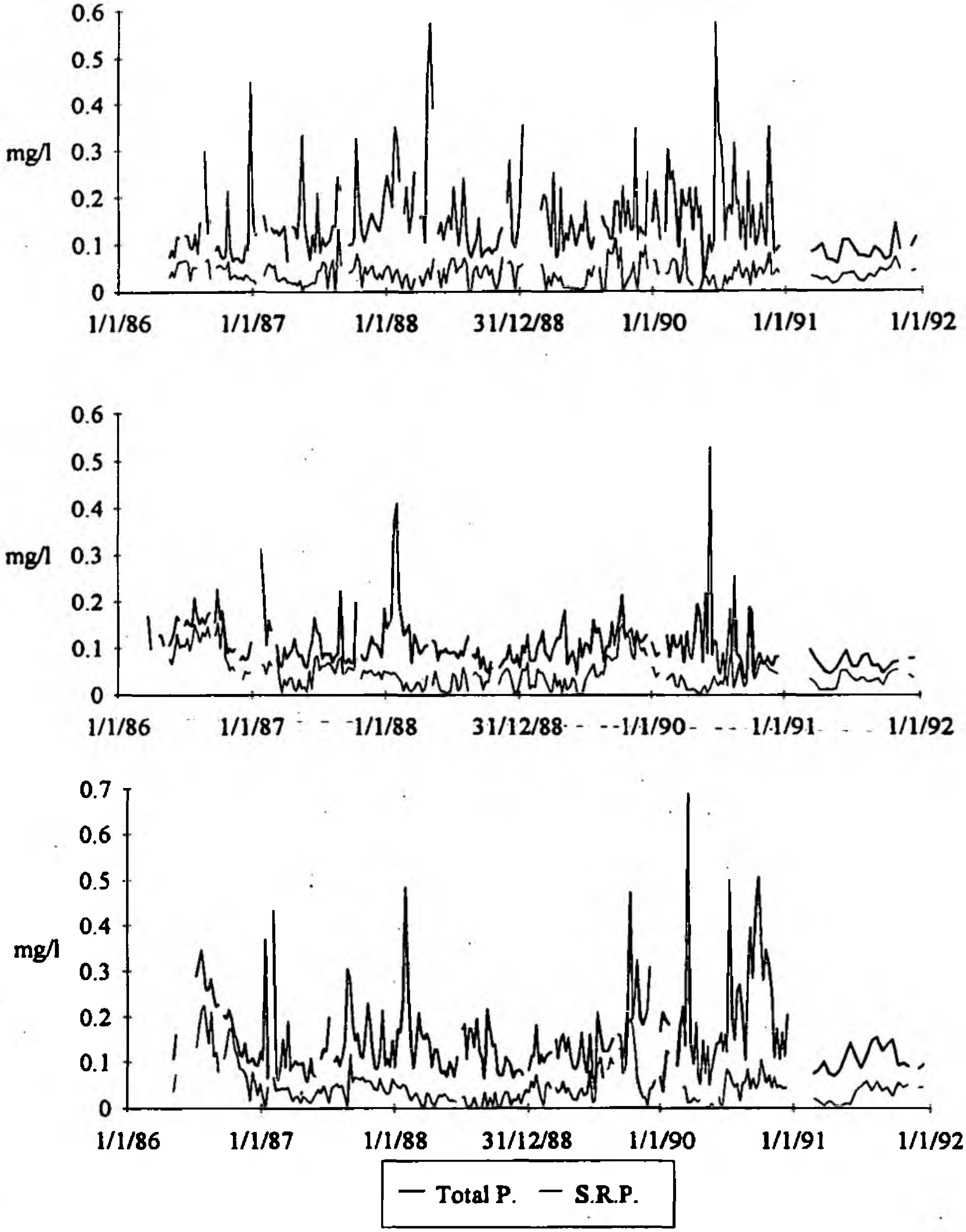


Figure 2.9 Total and soluble phosphorus concentration at (a) Ingworth Guaging Station (b) Horstead Mill (c) Wroxham Rail Bridge.

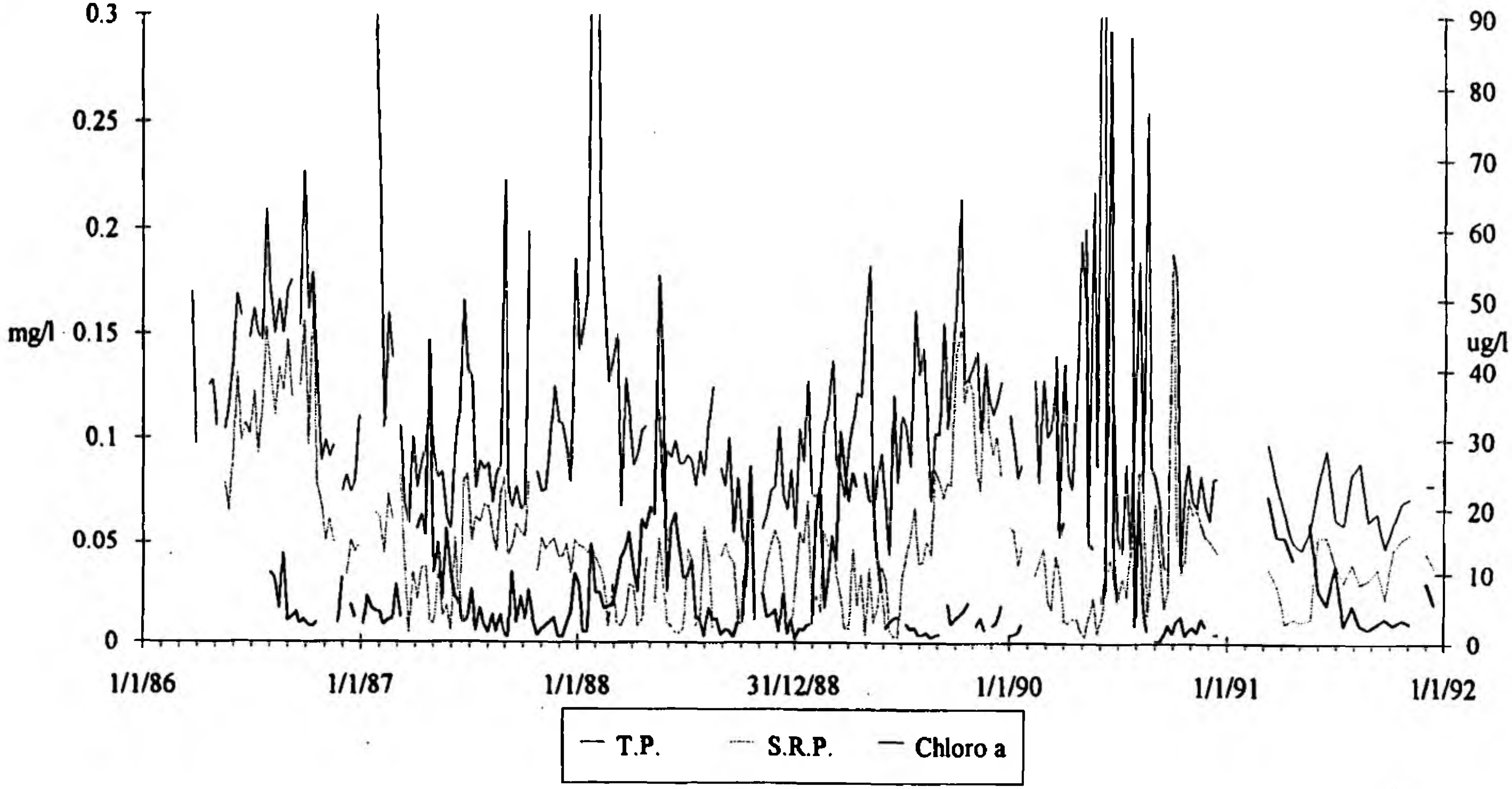


Figure 2.10 Chlorophyll a, total phosphorus and soluble reactive phosphorus concentration at Horstead Mill.

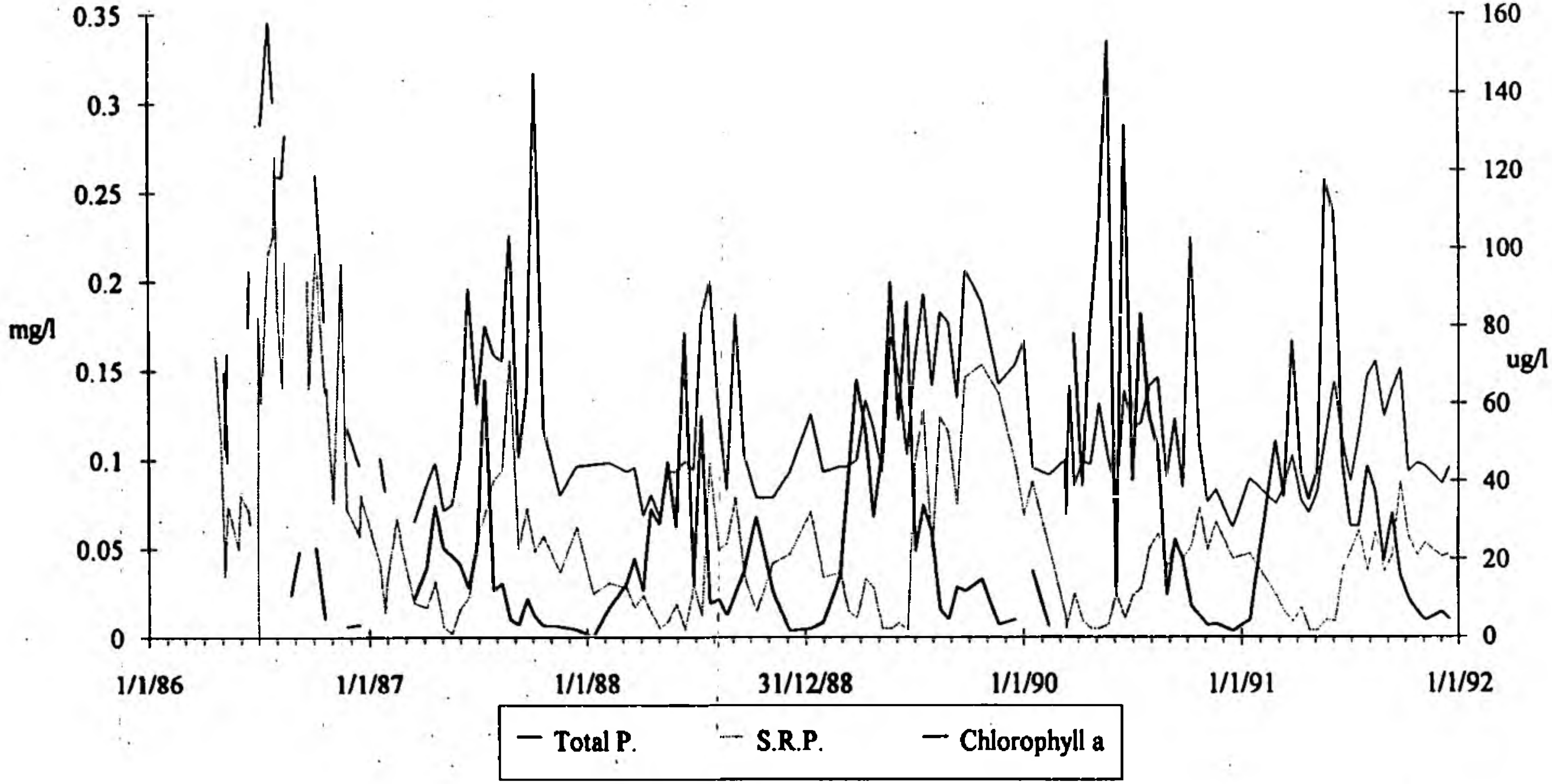


Figure 2.11 Chlorophyll a, total phosphorus and soluble reactive phosphorus concentration at Wroxham Rail Bridge.

Table 2.12 River Bure, concentration of chlorophyll

	ug/l Chlorophyll a	
	Horstead	Wroxham
Quarterly		
April-June 1986	2.41	42.00
July-Sept 1986	5.63	27.60
Oct-Dec 1986	3.98	7.34
Jan-March 1987	6.04	5.97
April-June 1987	12.47	21.57
July-Sept 1987	3.45	18.38
Oct-Dec 1987	3.74	3.15
Jan-March 1988	8.35	14.03
April-June 1988	19.22	34.16
July-Sept 1988	3.35	18.37
Oct-Dec 1988	6.23	14.87
Jan-March 1989	13.13	25.62
April-June 1989	13.34	60.83
July-Sept 1989	2.13	17.11
Oct-Dec 1989	3.46	7.53
Jan-March 1990	12.94	53.90
April-June 1990	25.88	86.73
July-Sept 1990	12.49	40.71
Oct-Dec 1990	2.30	4.34
Jan-March 1991	11.63	41.73
April-June 1991	11.63	65.18
July-Sept 1991	3.16	29.44
Oct-Dec 1991	5.17	6.56
Annually		
1986 mean	4.76	27.48
1987 mean	6.44	14.59
1988 mean	9.35	22.95
1989 mean	8.21	30.26
1990 mean	13.09	47.40
1991 mean	7.58	36.21

Table 2.13 River Bure, loadings of total and dissolved phosphorus

	kg/d phosphate-P			kg/d Total-P		
	Ingworth	Horstead	Wroxham	Ingworth	Horstead	Wroxham
Quarterly						
April-June 1986	3.67	14.34	18.69	8.85	22.18	39.09
July-Sept 1986	3.92	16.20	23.28	9.63	22.76	38.36
Oct-Dec 1986	4.01	10.79	15.57	15.61	17.26	27.47
Jan-March 1987	4.13	12.03	9.85	14.41	36.19	41.77
April-June 1987	1.87	6.65	5.84	12.97	19.94	24.06
July-Sept 1987	6.41	13.45	17.17	18.81	24.43	47.31
Oct-Dec 1987	7.79	13.80	18.71	29.95	33.31	53.69
Jan-March 1988	5.47	10.49	13.67	43.13	71.15	90.87
April-June 1988	4.15	4.22	5.01	28.83	20.02	25.52
July-Sept 1988	3.74	5.22	2.59	10.28	14.54	29.35
Oct-Dec 1988	4.18	6.17	5.42	13.74	12.48	16.96
Jan-March 1989	3.28	6.13	9.09	18.82	17.01	26.08
April-June 1989	1.04	3.21	6.00	10.02	16.17	19.80
July-Sept 1989	3.44	7.29	13.87	7.93	12.85	19.30
Oct-Dec 1989	3.45	15.48	12.53	13.52	20.00	41.23
Jan-March 1990	5.69	7.29	12.57	21.78	22.12	51.57
April-June 1990	0.96	1.96	1.89	11.72	18.39	15.61
July-Sept 1990	1.53	4.51	5.98	8.36	8.24	29.36
Oct-Dec 1990	3.08	7.39	9.66	9.96	10.95	33.25
Jan-March 1991	2.75	7.80	4.85	8.17	16.72	17.89
April-June 1991	1.61	3.52	3.08	4.59	8.93	15.80
July-Sept 1991	1.47	2.62	4.24	3.21	5.14	12.07
Oct-Dec 1991	3.58	7.47	9.01	7.60	11.74	17.61
Annually						
1986 mean	3.88	14.05	19.16	11.27	22.33	33.76
1987 mean	5.03	11.40	12.95	19.39	27.88	41.95
1988 mean	4.37	6.61	6.48	22.06	30.77	41.03
1989 mean	2.71	7.88	10.27	11.89	16.44	25.72
1990 mean	2.69	4.88	7.16	13.31	16.33	36.42
1991 mean	2.18	4.73	4.93	5.51	9.65	15.42

Table 2.14 River Bure, loadings of particulate phosphorus and solids

	kg/d Particulate P.			kg/d Solids		
	Ingworth	Horstead	Wroxham	Ingworth	Horstead	Wroxham
Quarterly						
April-June 1986	5.25	6.62	19.57	0.86	0.72	2.52
July-Sept 1986	5.50	5.56	15.08	0.52	0.45	1.90
Oct-Dec 1986	11.60	6.80	11.90	1.15	0.72	1.19
Jan-March 1987	8.98	24.16	31.93	2.60	3.18	7.72
April-June 1987	11.19	13.28	18.38	2.87	1.51	3.18
July-Sept 1987	12.39	10.98	29.27	2.06	2.42	5.13
Oct-Dec 1987	22.16	19.51	34.98	5.15	4.06	6.55
Jan-March 1988	37.25	60.66	77.19	9.83	14.40	15.45
April-June 1988	19.68	15.80	20.51	4.94	2.64	4.35
July-Sept 1988	6.54	9.32	26.76	0.93	1.14	4.83
Oct-Dec 1988	9.56	6.72	12.76	2.15	0.81	1.41
Jan-March 1989	15.54	10.88	16.99	1.71	1.25	2.85
April-June 1989	8.97	12.96	13.80	0.79	1.37	2.70
July-Sept 1989	4.48	5.56	6.29	0.43	0.51	1.61
Oct-Dec 1989	10.06	4.52	29.08	1.11	0.53	3.13
Jan-March 1990	16.10	14.83	35.51	2.19	2.54	3.09
April-June 1990	10.27	16.42	13.73	0.68	1.52	2.00
July-Sept 1990	6.83	4.04	23.38	0.41	0.53	3.88
Oct-Dec 1990	6.88	4.32	23.59	0.46	0.29	1.15
Jan-March 1991	5.42	8.92	13.04	0.75	1.00	1.65
April-June 1991	2.98	5.41	12.72	0.29	0.43	2.04
July-Sept 1991	1.73	2.52	7.82	0.16	0.17	1.05
Oct-Dec 1991	4.02	4.27	8.61	0.63	0.25	0.80
Annually						
1986 mean	7.91	6.45	14.45	0.85	0.83	1.75
1987 mean	13.99	16.48	28.77	3.27	2.76	5.65
1988 mean	17.62	24.25	35.01	4.49	4.94	6.48
1989 mean	9.20	8.56	16.16	0.92	0.94	2.55
1990 mean	10.51	10.06	24.71	0.96	1.43	3.19
1991 mean	3.33	4.92	10.49	0.42	0.42	1.41

Table 2.15 River Bure flow

		Flow cumecs		
		Ingworth	Horstead	Wroxham
Quarterly				
April-June	1986	1.15	2.14	2.55
July-Sept	1986	0.86	1.49	1.78
Oct-Dec	1986	1.27	2.05	2.45
Jan-March	1987	1.47	2.80	3.34
April-June	1987	1.54	2.36	2.82
July-Sept	1987	1.40	2.47	2.94
Oct-Dec	1987	1.81	3.44	4.10
Jan-March	1988	2.10	4.06	4.84
April-June	1988	1.25	2.58	3.07
July-Sept	1988	1.02	1.95	2.32
Oct-Dec	1988	1.14	1.98	2.36
Jan-March	1989	1.05	2.06	2.44
April-June	1989	0.91	1.71	1.98
July-Sept	1989	0.68	1.23	1.46
Oct-Dec	1989	1.00	1.93	2.36
Jan-March	1990	1.33	2.93	3.49
April-June	1990	0.73	1.50	1.79
July-Sept	1990	0.56	0.99	1.17
Oct-Dec	1990	0.83	1.67	2.02
Jan-March	1991	1.00	2.13	2.54
April-June	1991	0.67	1.52	1.81
July-Sept	1991	0.53	0.93	1.11
Oct-Dec	1991	0.81	1.78	2.13
Annually				
1986 mean		1.21	2.09	2.50
1987 mean		1.46	3.30	3.30
1988 mean		1.36	2.61	3.11
1989 mean		0.91	1.73	2.07
1990 mean		0.86	1.77	2.11
1991 mean		0.75	1.59	1.89

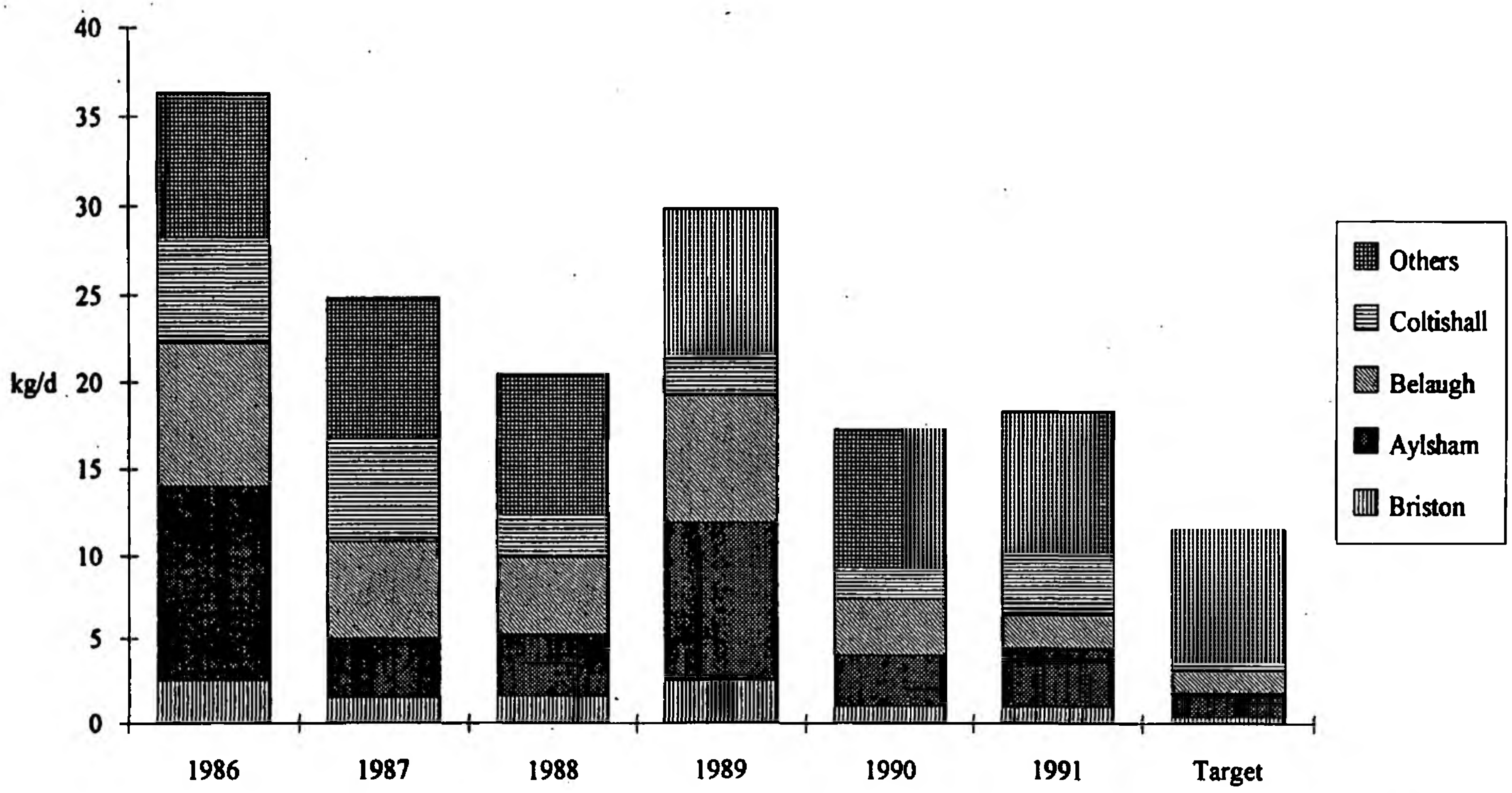


Figure 2.12 **Total phosphorus discharged to the River Bure from the sewage treatment works.**

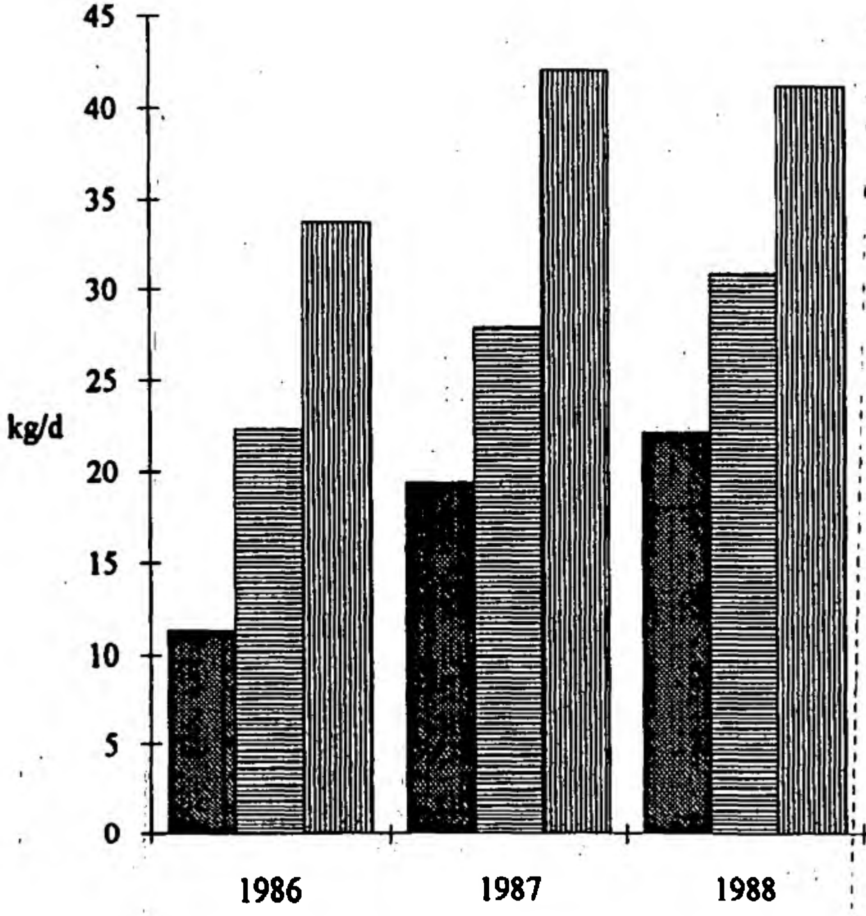
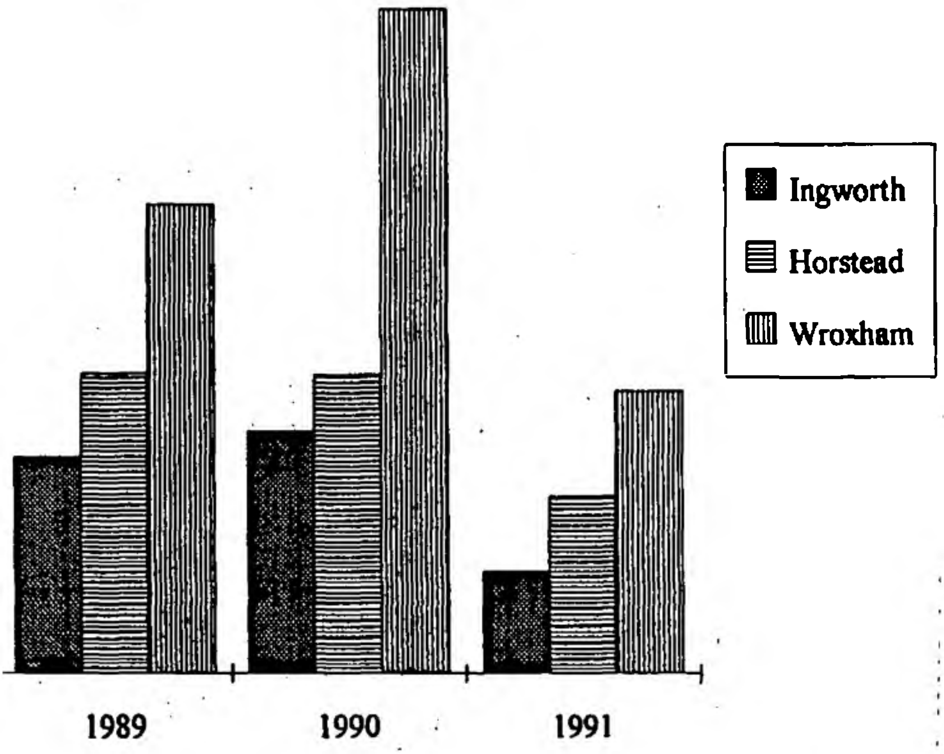


Figure 2.13 Total phosphorus load in the River Bure.


Ingworth
Horstead
Wroxham
1989
1990
1991

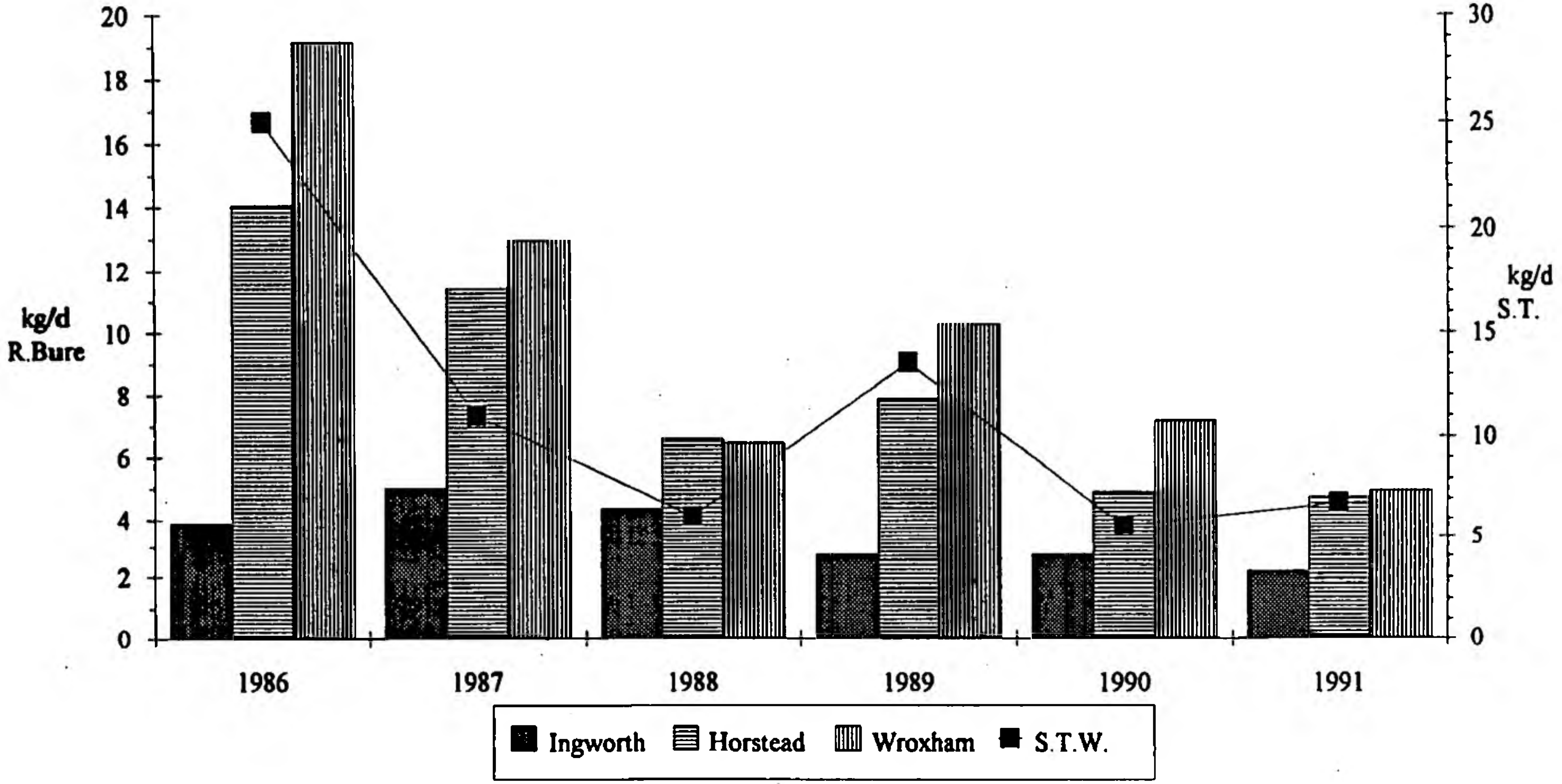


Figure 2.14 **Soluble reactive phosphorus load in the River Bure and soluble reactive phosphorus input from the sewage treatment works.**

Figure 2.12 shows the Total P. discharged into the river Bure from the S.T.W.'s. The levels achieved are still above the target level, however, the effectiveness of phosphorus stripping has been maintained. Assessment of the impact of phosphate stripping is complicated by the effects of river discharge. Total P. loadings in the river appear to more closely reflect the changes in river flow than the changes in Total P. discharged from the S.T.W.'s (Figure 2.13). This is largely due to the close relationship between flow rate and Particulate P. loads i.e. in periods of high river flow there is an increase in particulate P.

A closer relationship can be seen between the S.R.P. discharged from the S.T.W's and the changes in river phosphorus loads (Figure 2.14). The S.R.P. fraction is not so influenced by periods of high flow and therefore enables a more direct assessment to be made of the impact of phosphorus removal from the S.T.W.'s. At both Horstead Mill and Wroxham Rail Bridge the S.R.P. loadings closely follow the changes in phosphorus discharged from the S.T.W's since stripping was initiated in 1986. The S.R.P. load at Horstead has remained unchanged in comparison with 1990, whereas, the loadings at Wroxham Rail Bridge have decreased. This may be due to an increase in the level of algal productivity, which will be enhanced by the low river flows and reduced flushing rate. The S.R.P. loadings at Ingworth have remained unchanged since 1989.

2.4 The Effect Of Phosphate Removal On The Limnology Of The River Bure Broads

The effect of continued phosphorus removal from the S.T.W.'s on the River Bure Broads is difficult to interpret. Figures 2.15-18 illustrate the long term seasonal trends in phosphorus cycling at four of the Bure Broads; Belaugh, Wroxham, Ranworth and South Walsham.

2.4.1 Belaugh Broad

Belaugh Broad is an upper river broad which was mud-pumped in 1987. Monitoring of the broad was not initiated until the summer of 1987. The Total P. concentration has remained fairly constant between 0.1 and 0.2 mgP /l. The S.R.P. concentration undergoes seasonal cycling which is characterised by a decrease in the spring and summer due to uptake by algae. Chlorophyll a concentration change is shown in Figure 2.19. The high and sudden increases in chlorophyll concentration in 1988 are coincident with periods of heavy rain, suggesting an input of phosphorus from the surrounding carr (Third Annual Report). The changes in concentration during 1989 - 1991 have been less erratic showing the expected seasonal cycling of a spring rise followed by a decline towards the autumn. A lowering of algal numbers during the summer of 1990 may well be due to the appearance of *Daphnia* (Figure 2.23) which graze upon the algae. The chlorophyll levels did not fall during the summer of 1991 after the normal spring peak and *Daphnia* were not recorded as a component of the Broad's zooplankton. The winter increase in the concentration of S.R.P. in 1989 was higher than for previous years, this may be a reflection of the decrease in the effectiveness of the phosphate stripping process at the S.T.W.'s.

2.4.2 Wroxham Broad

Wroxham Broad is also an upper river broad which is well flushed by the river. The pre-phosphate stripping, 1986, Total P. concentration was higher than the levels recorded for the following years (Figure 2.16). The decrease in phosphorus supply to the broad is

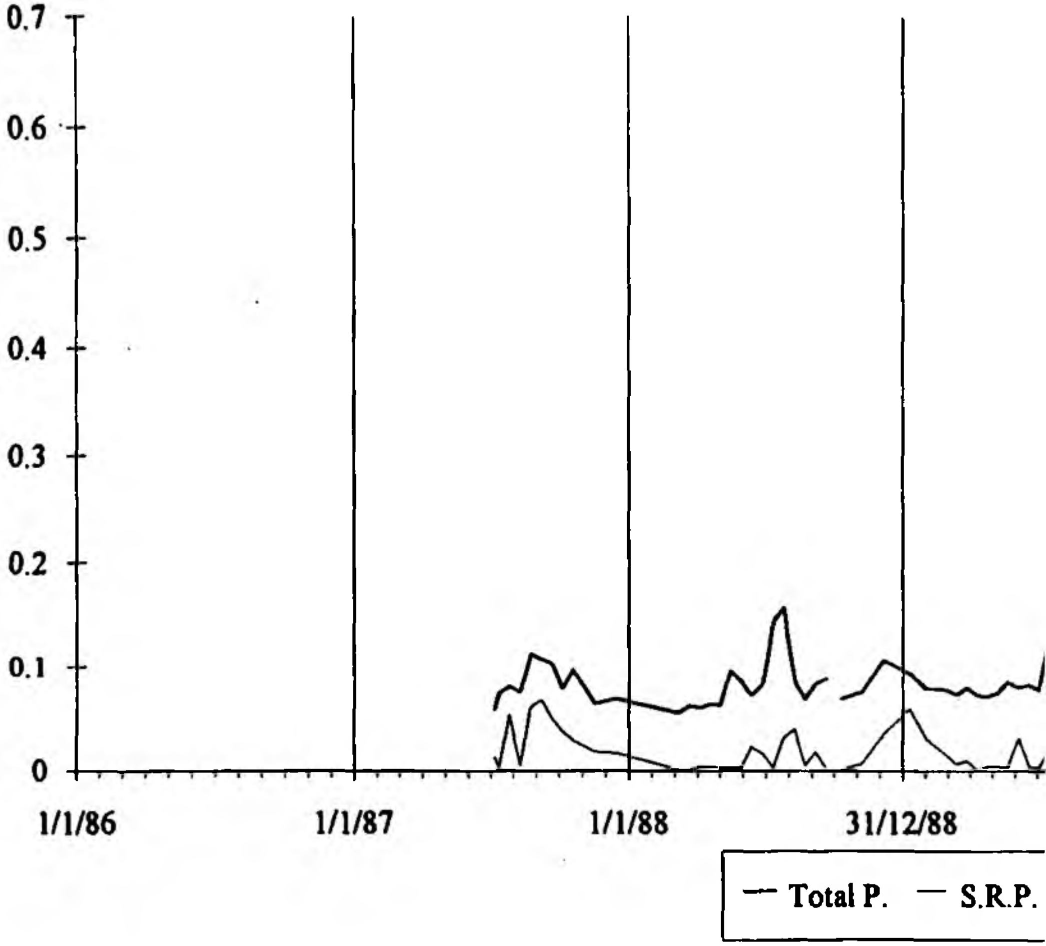
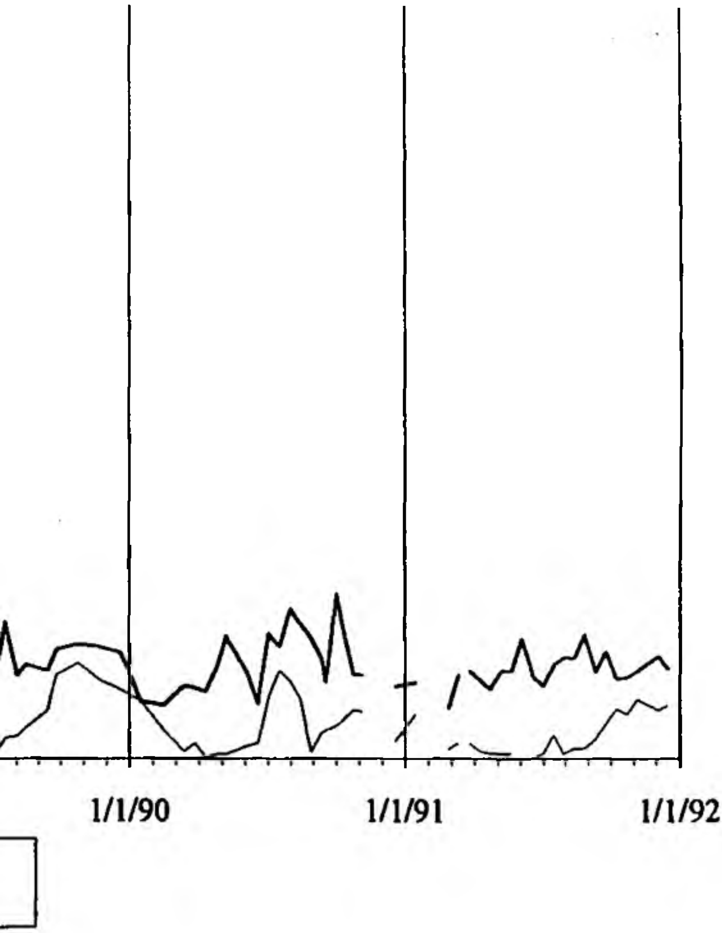


Figure 2.15 Belaugh Broad: phosphorus concentration (mg/l).



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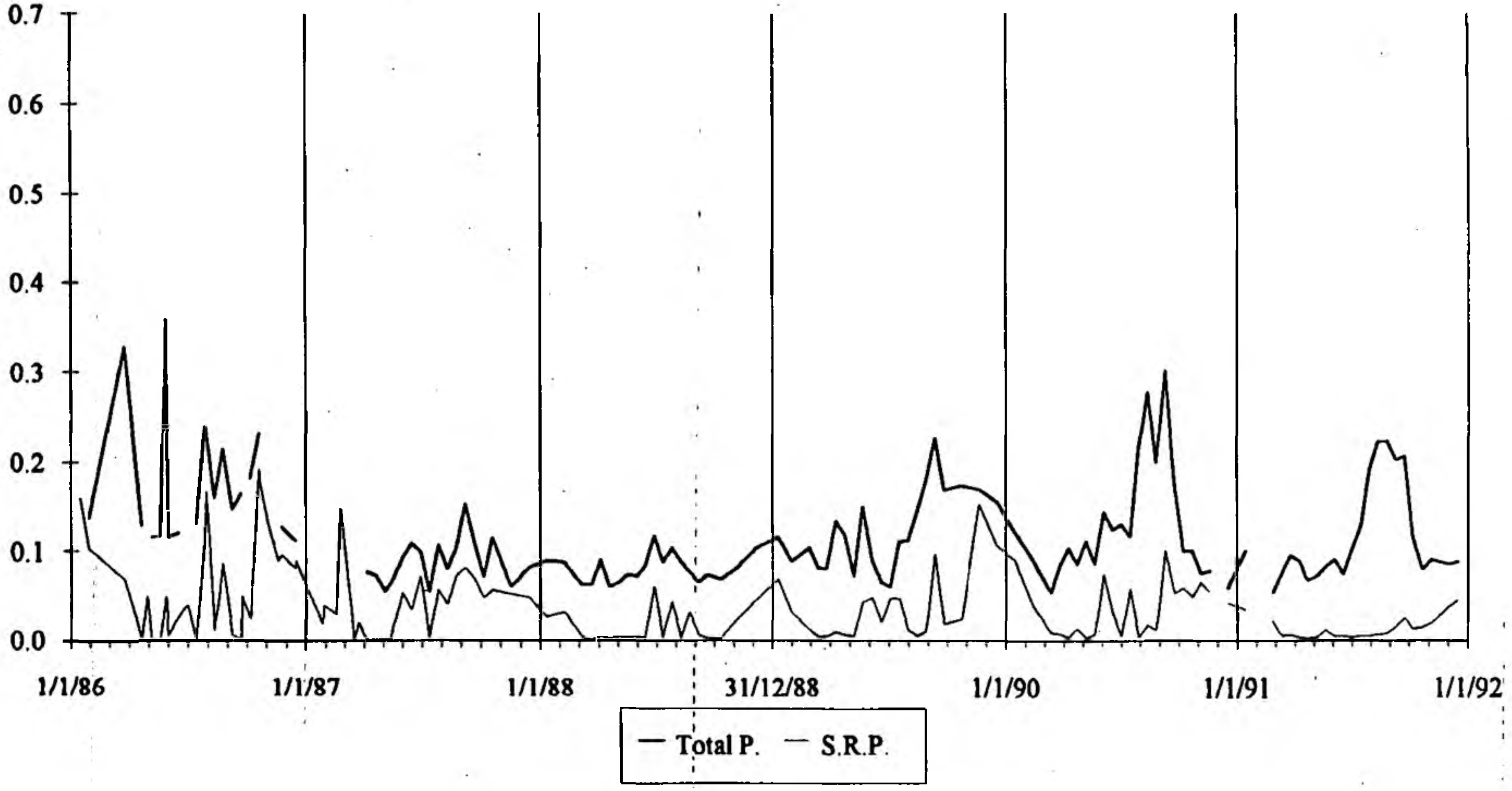


Figure 2.16 Wroxham Broad: phosphorus concentration (mg/l).

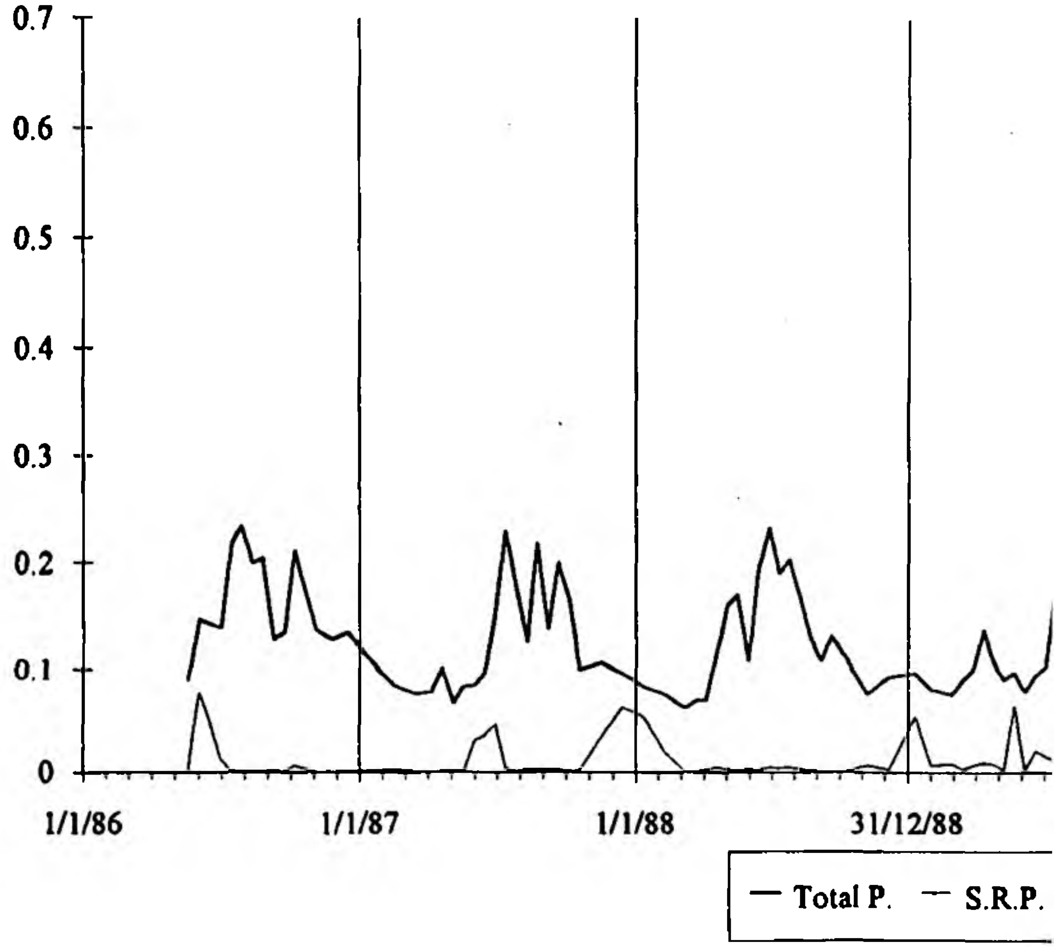
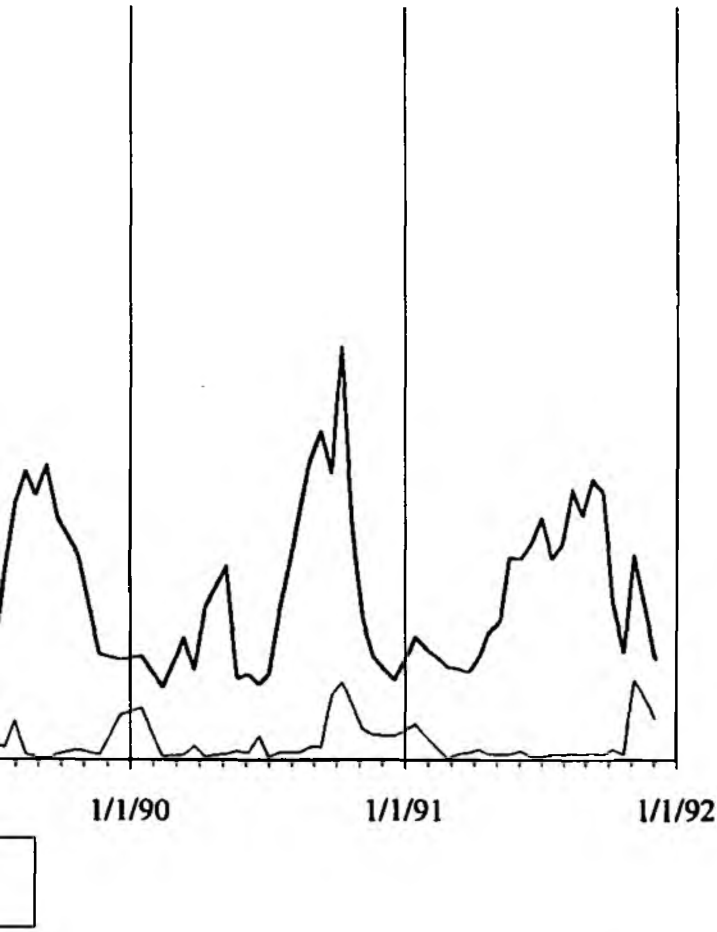


Figure 2.17 Ranworth Broad: phosphorus concentration (mg/l).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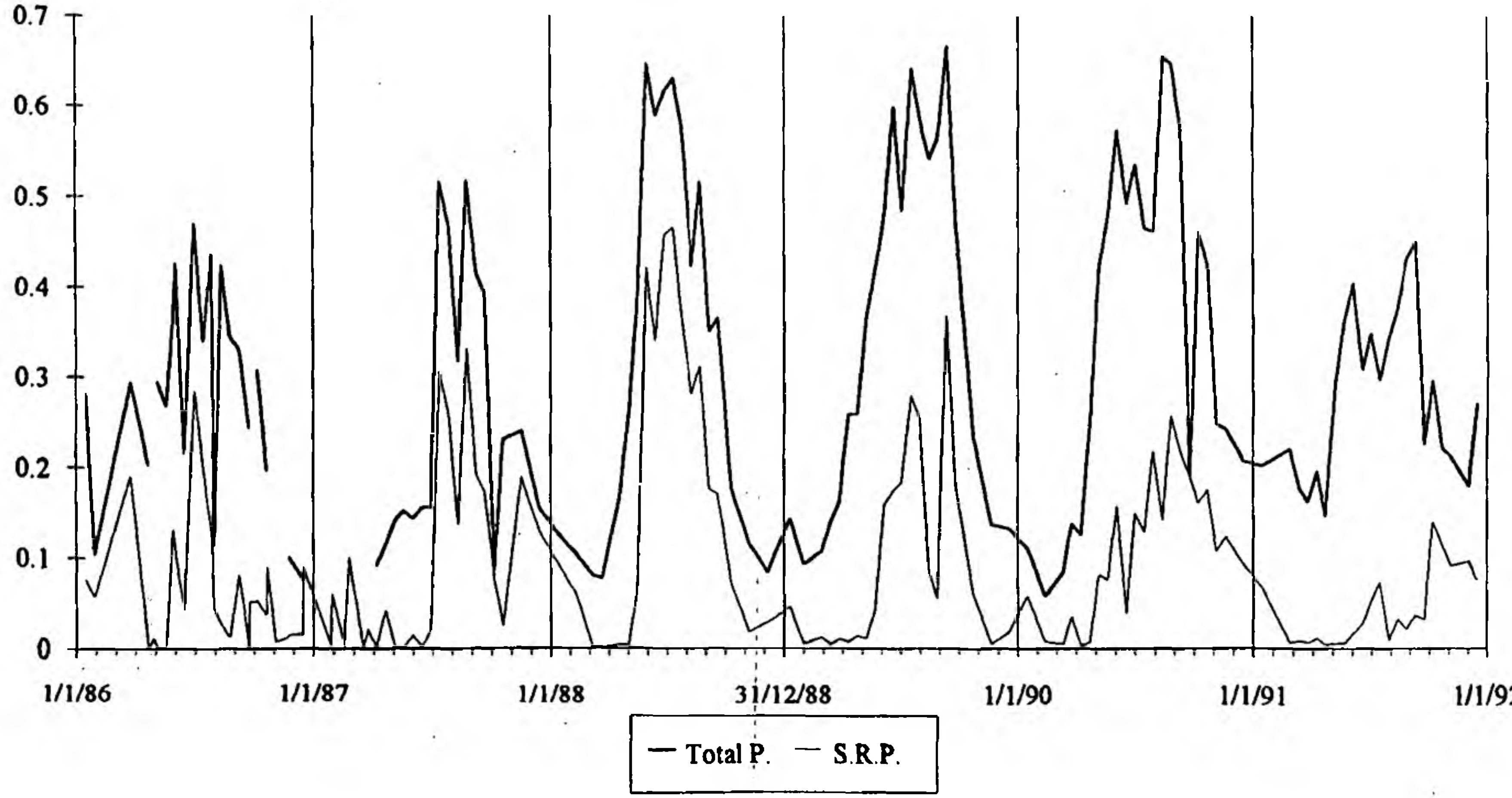


Figure 2.18 South Walsham Broad: phosphorus concentration (mg/l).

associated with a decrease in chlorophyll a concentration during 1987 (Figure 2.20). Due to the reduction in algal uptake over the summer months the available phosphorus concentration remained high. The Total P. concentration in 1988 remained at the lower level whereas the S.R.P. level fell in the spring due to an increase in algal productivity. This seasonal cycling of a decline in available phosphorus in the spring associated with an increase in chlorophyll has continued until 1991. However, during the late summer an increase in the amount of available phosphorus and more importantly, an increase in the Total P. concentration has also been recorded. This pattern suggests that there is sediment release occurring. Also complicating this seasonal cycle is the grazing effect of the cladocerans upon the phytoplankton community. In 1989 and 1990 a fall in chlorophyll a concentration is associated with an increase in the numbers of *Daphnia* and other cladocera (Figure 2.24). The extent of the summer decrease in algal numbers is affected by both the actual numbers of zooplankton and the influence of phosphorus release from the sediments. The numbers of zooplankton are lower in 1990 and therefore the chlorophyll a concentration only fell briefly. Also, in 1989, when the numbers of zooplankton were higher the increase in phosphorus concentration in the late summer brought about an increase in chlorophyll a concentration. Another complication is the effectiveness of the phosphorus removal process at the S.T.W.'s. There is an increase in both the concentration of Total P. and the S.R.P. in 1989 which can be associated with the poor performance of phosphate stripping at that time. During 1991 the Total P. levels are as high as previous years, especially in the late summer. However, the available phosphorus does not appear to have increased at this time. The increase in Total P. is related to an increase in chlorophyll a concentration. It is probable that tidal mixing is having a greater effect, due to the low river flows, and that algal rich water from the middle river is being brought up into the upper broads. Furthermore, the reduction in river flow will lead to an increase in retention time which is favourable to algal growth.

2.4.3 Ranworth Broad

Ranworth Broad is situated in the middle region of the River Bure and is characterised by summer peaks in Total P. (Figure 2.17). The Total P. concentration has changed very little since 1986; most years show a spring increase followed by a early summer fall and then the Total P. peak in the late summer. The lowering in Total P. concentration is associated with a decline in the chlorophyll a concentration i.e. a fall in the number of algae (Figure 2.21). The extent of the decline in the algal numbers is dependant upon the *Daphnia* population (Figure 2.25). The spring / summer decline is more pronounced in 1989 and 1990 when the numbers of cladocera were higher. No *Daphnia* were collected during 1991 and there was no associated fall in chlorophyll a concentration. During these periods of low phytoplankton numbers there are small, short-lived peaks in S.R.P.. The increases in S.R.P. that are recorded later in the year, associated with an increase in Total P., are likely to be due to the release of phosphorus from the sediment to the overlying water.

2.4.4 South Walsham Broad

South Walsham Broad is located in the lower reaches of the River Bure. The phosphorus cycling within this broad reflects the long retention time and the effects of sediment release (Figure 2.18). Also, this site receives direct inputs of phosphorus from a small S.T.W.. The concentration of Total P. in the summer is high (up to 0.7mgP/l) and has increased from 1986 to 1989. This increase is reflected in the chlorophyll a concentration (Figure 2.22). The high level of Total P. is supported by a very high concentration of available phosphorus (between 0.2 - 0.5 mgP/l). These concentrations

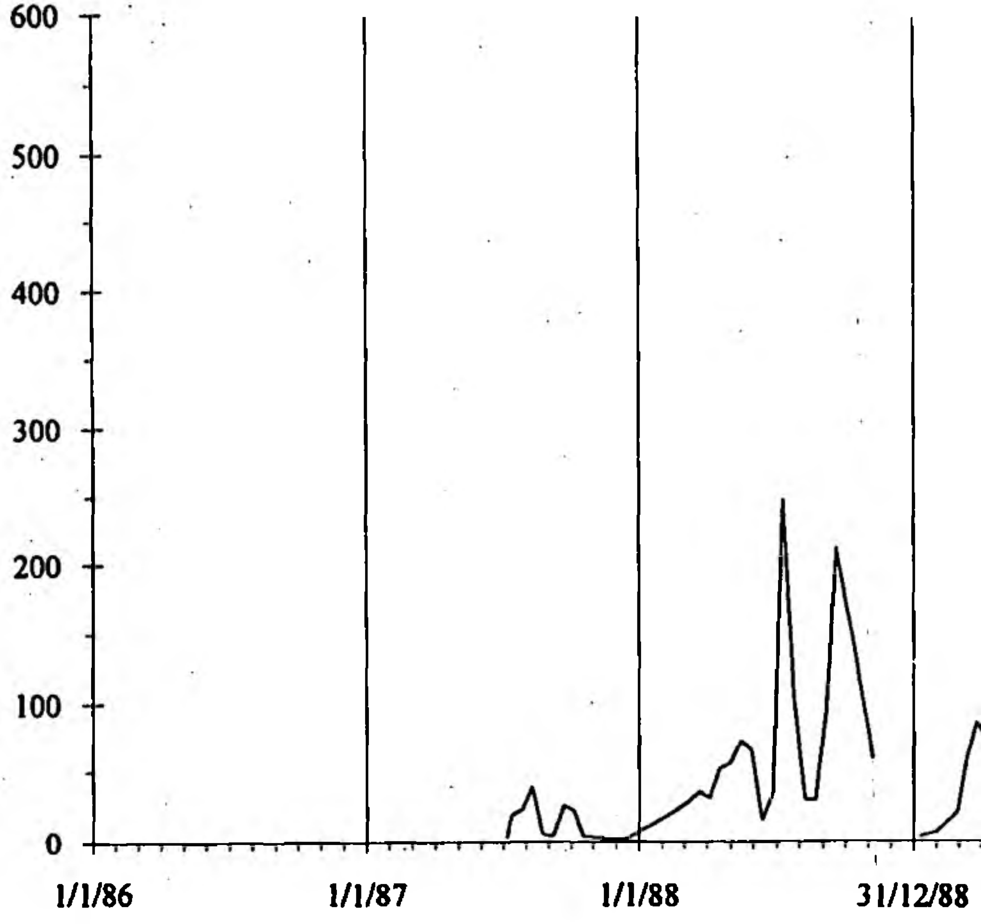
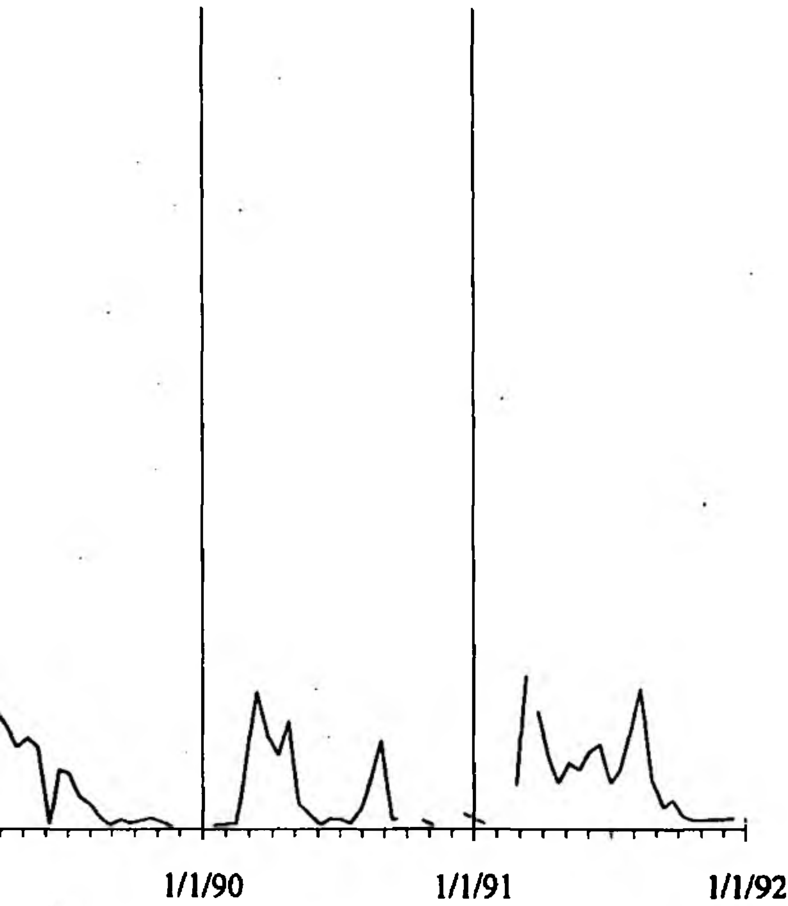


Figure 2.19 Belaugh Broad: chlorophyll a concentration (ug/l).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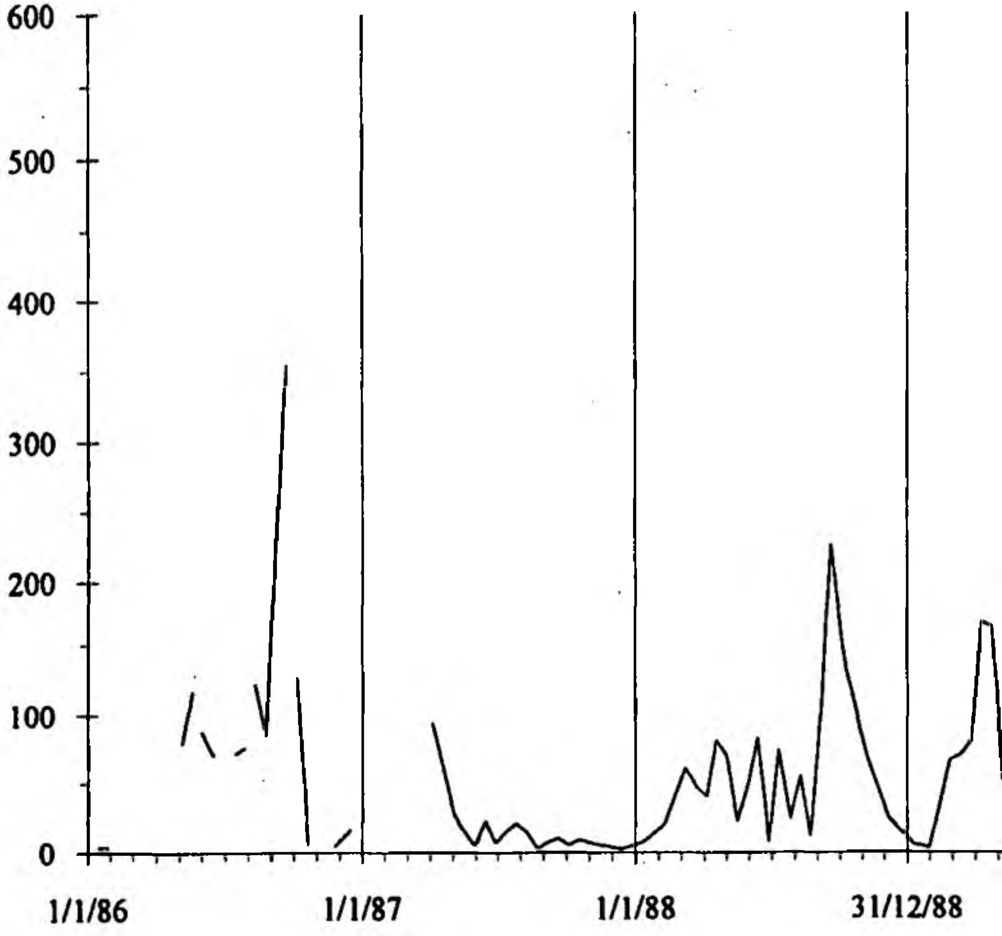
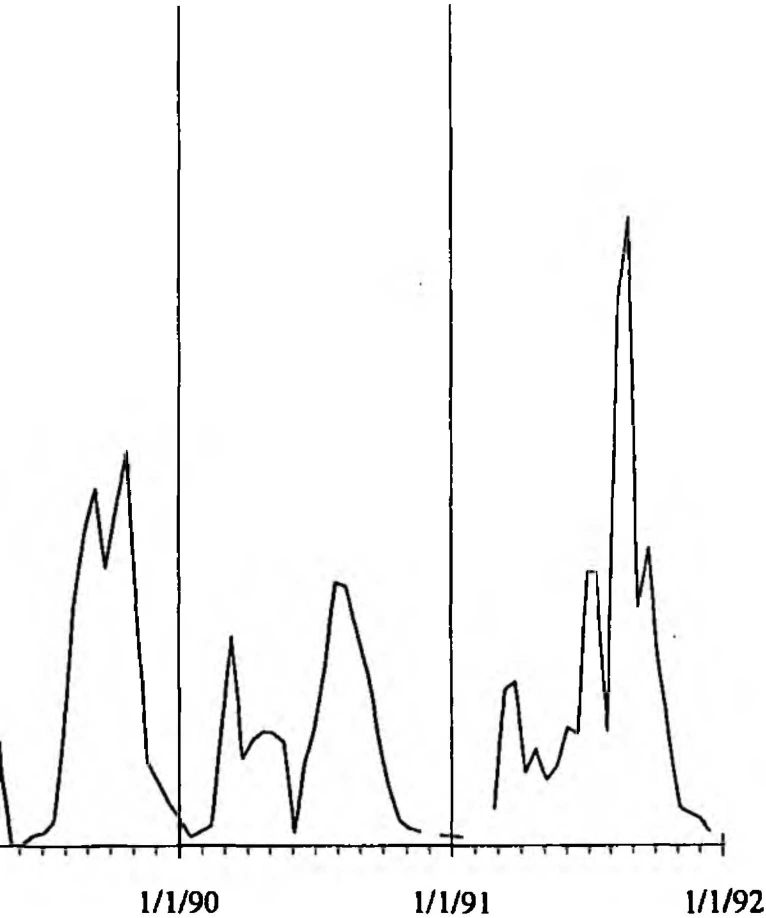


Figure 2.20 Wroxham Broad: chlorophyll a concentration (ug/l).



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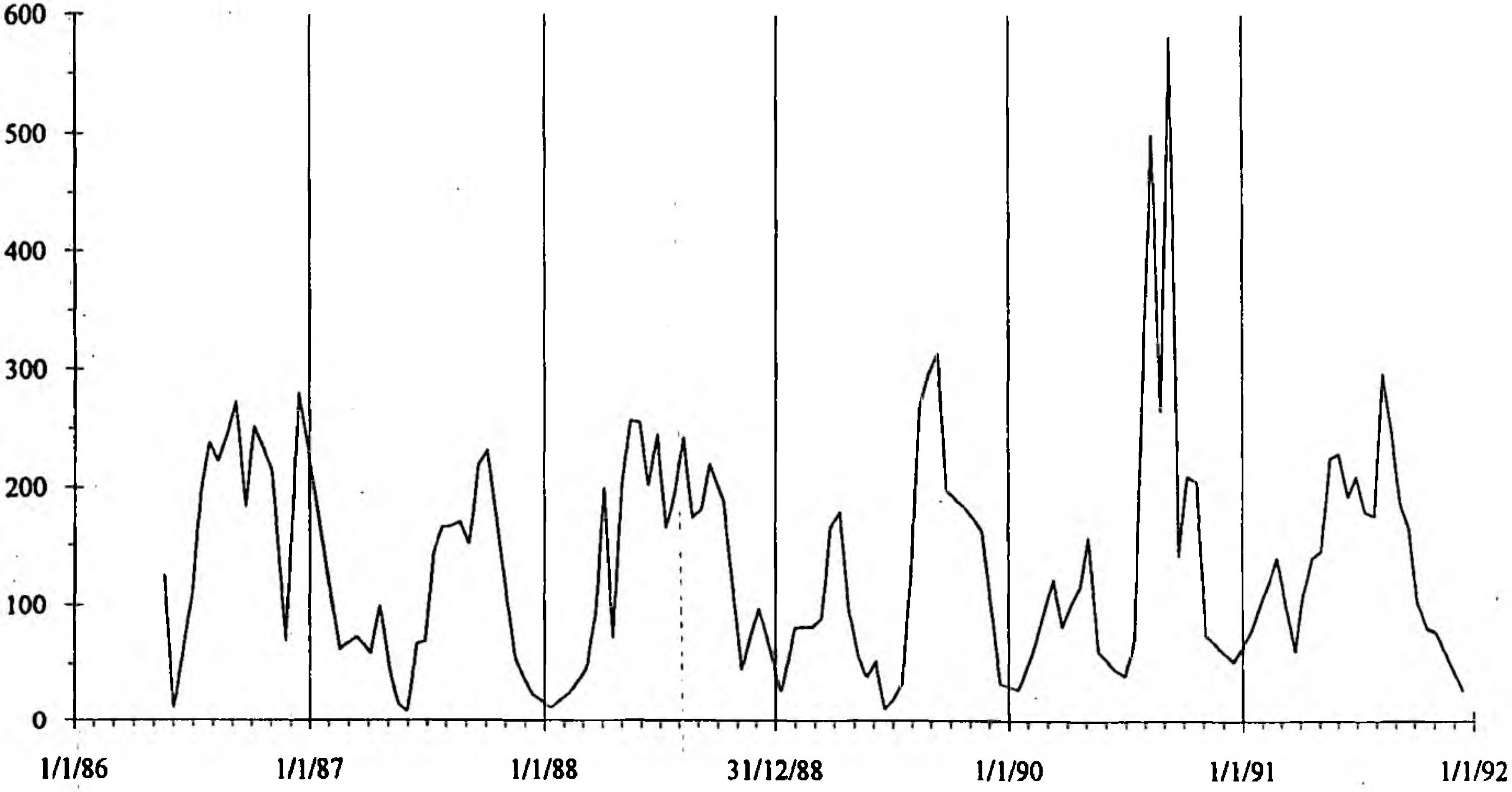


Figure 2.21 Ranworth Broad: chlorophyll a concentration (ug/l).

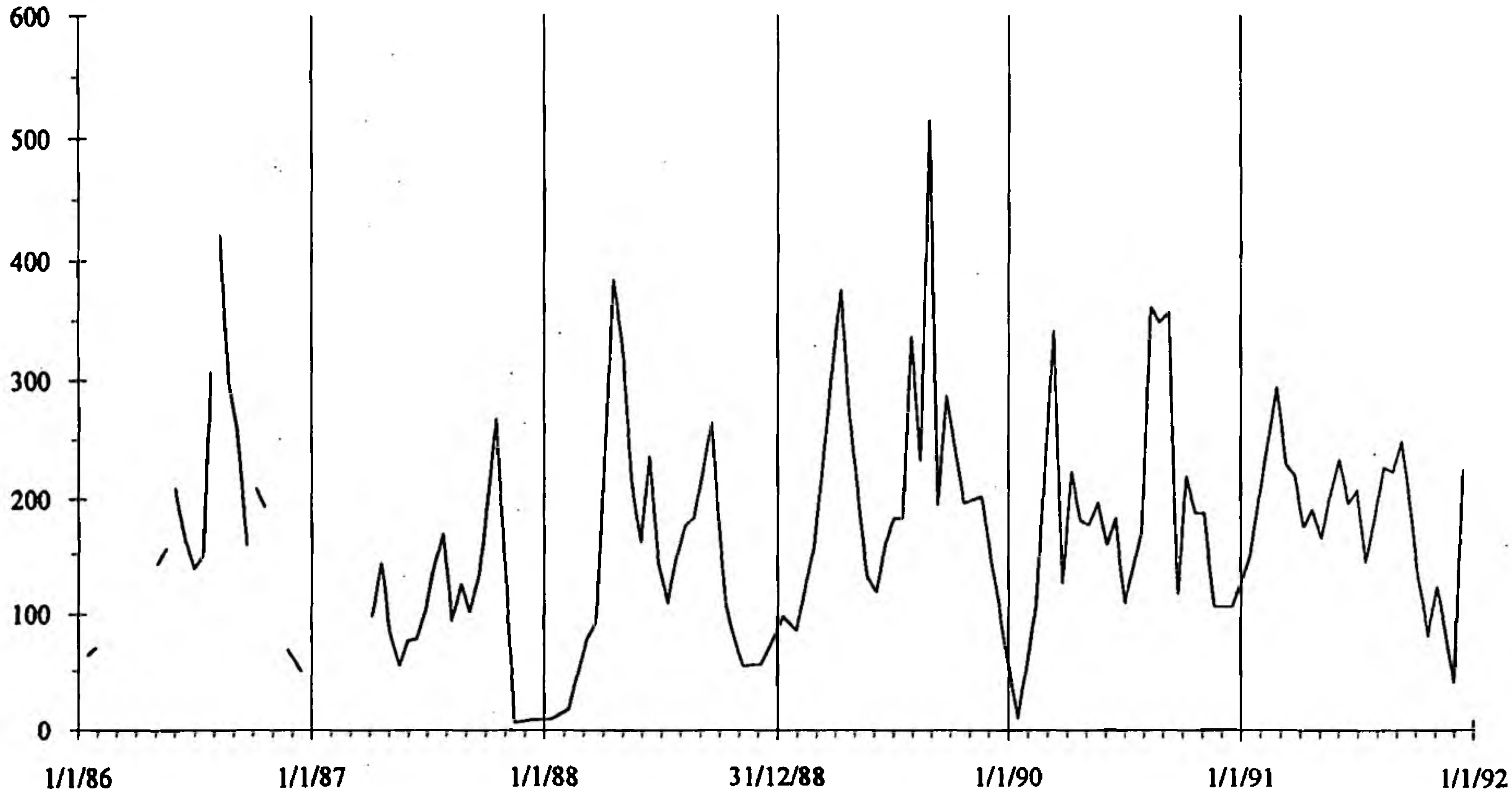


Figure 2.22 South Walsham Broad: chlorophyll a concentration (ug/l).

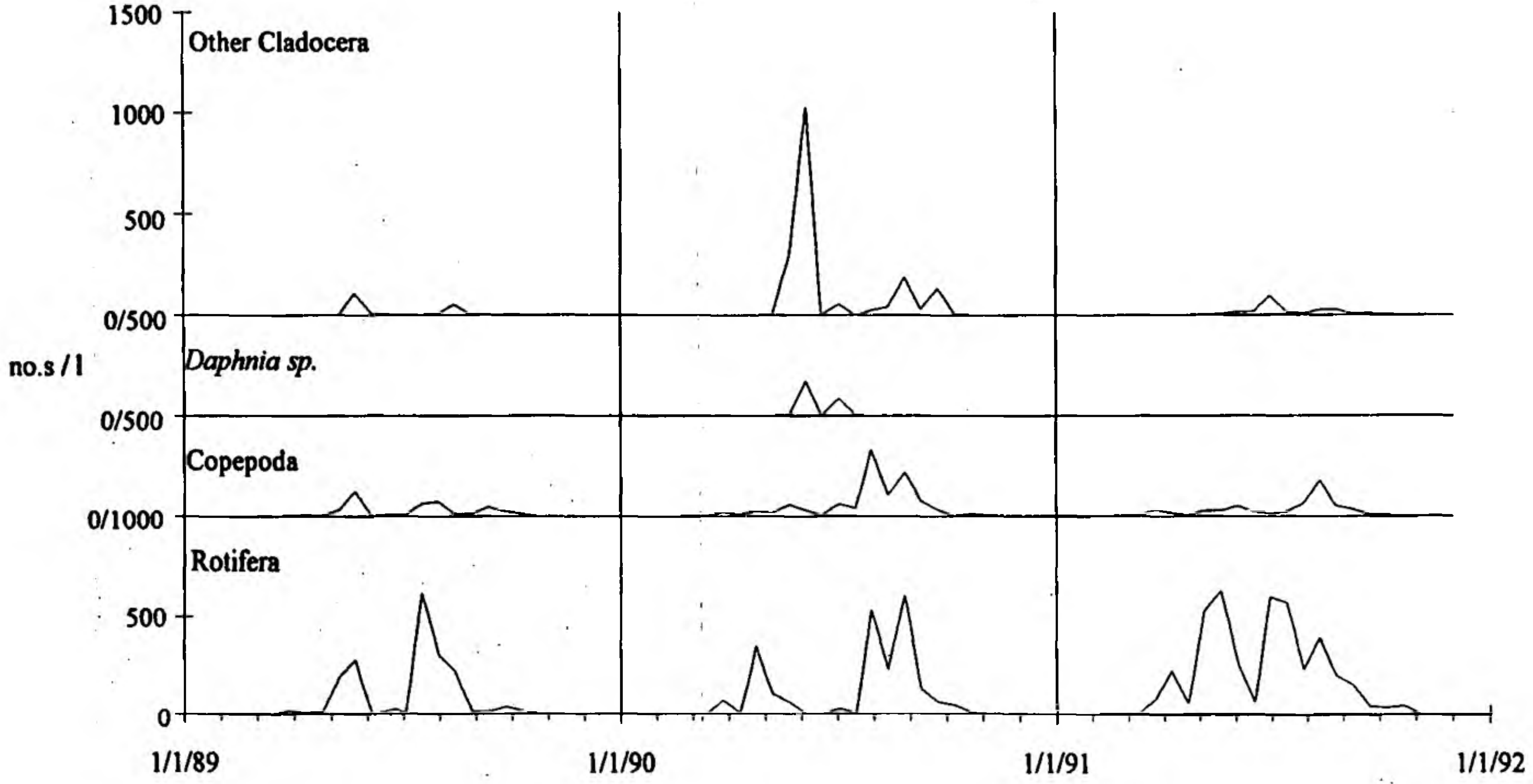


Figure 2.23 Belaugh Broad: zooplankton population.

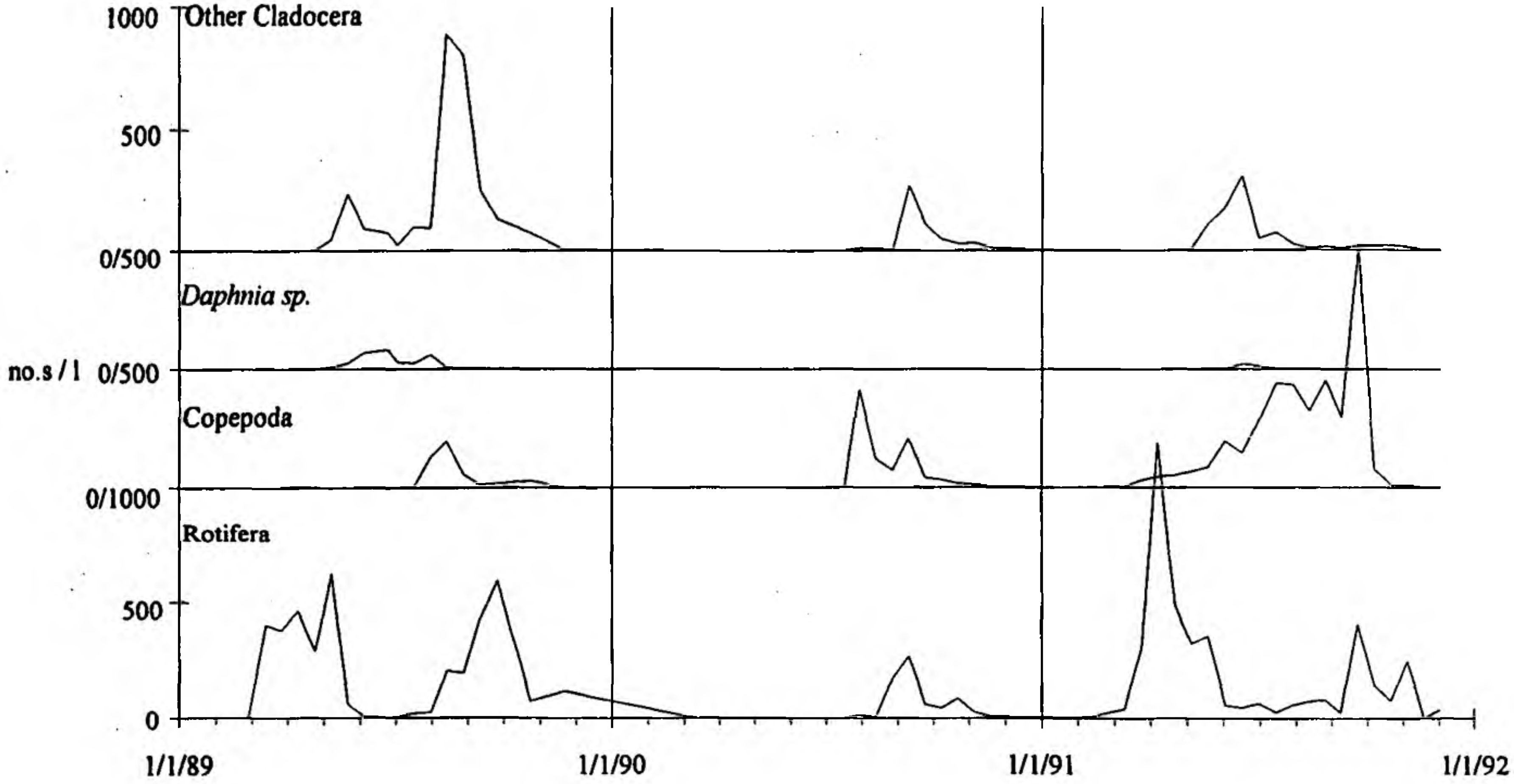


Figure 2.24 Wroxham Broad: zooplankton population.

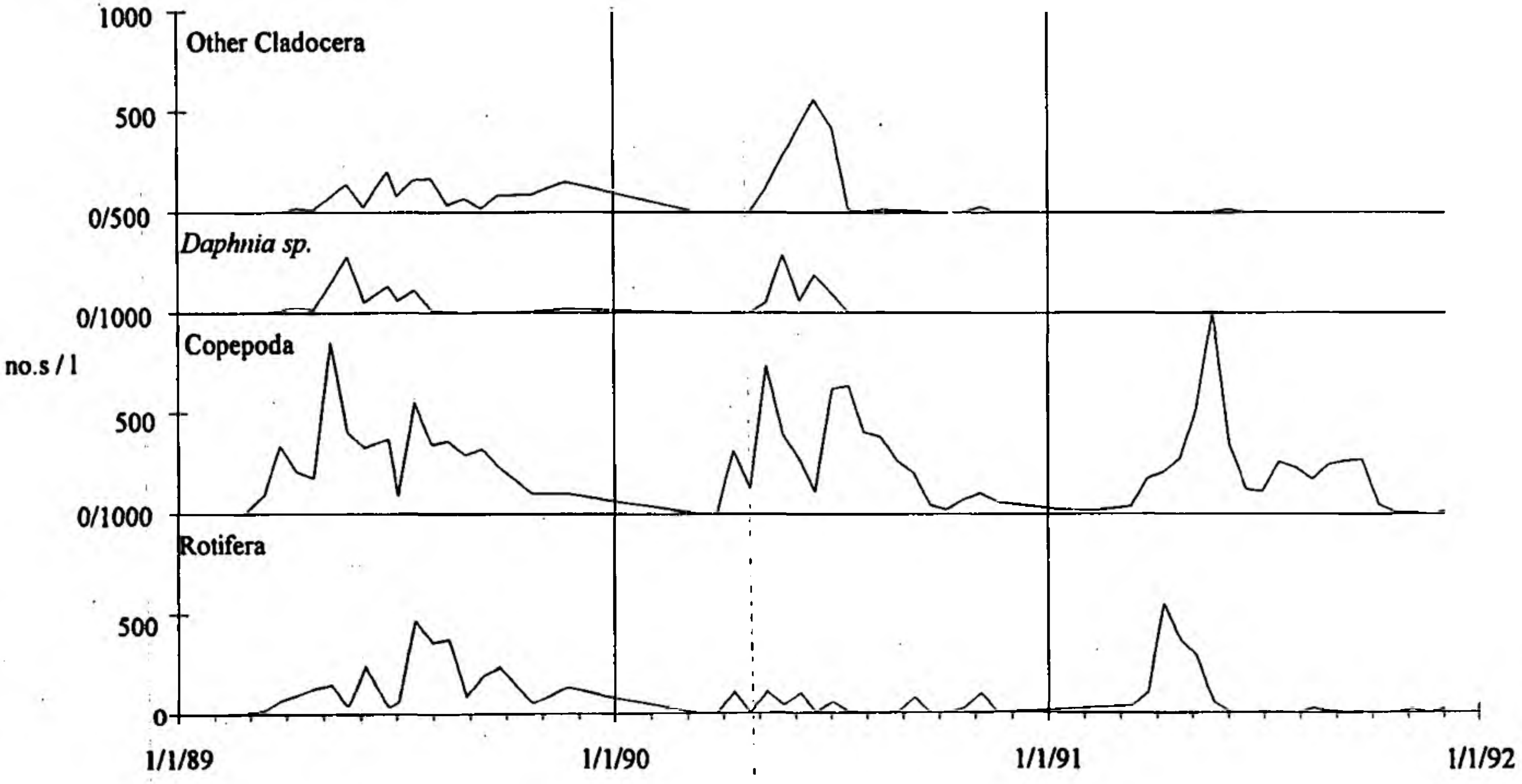


Figure 2.25 Ranworth Broad: zooplankton population.

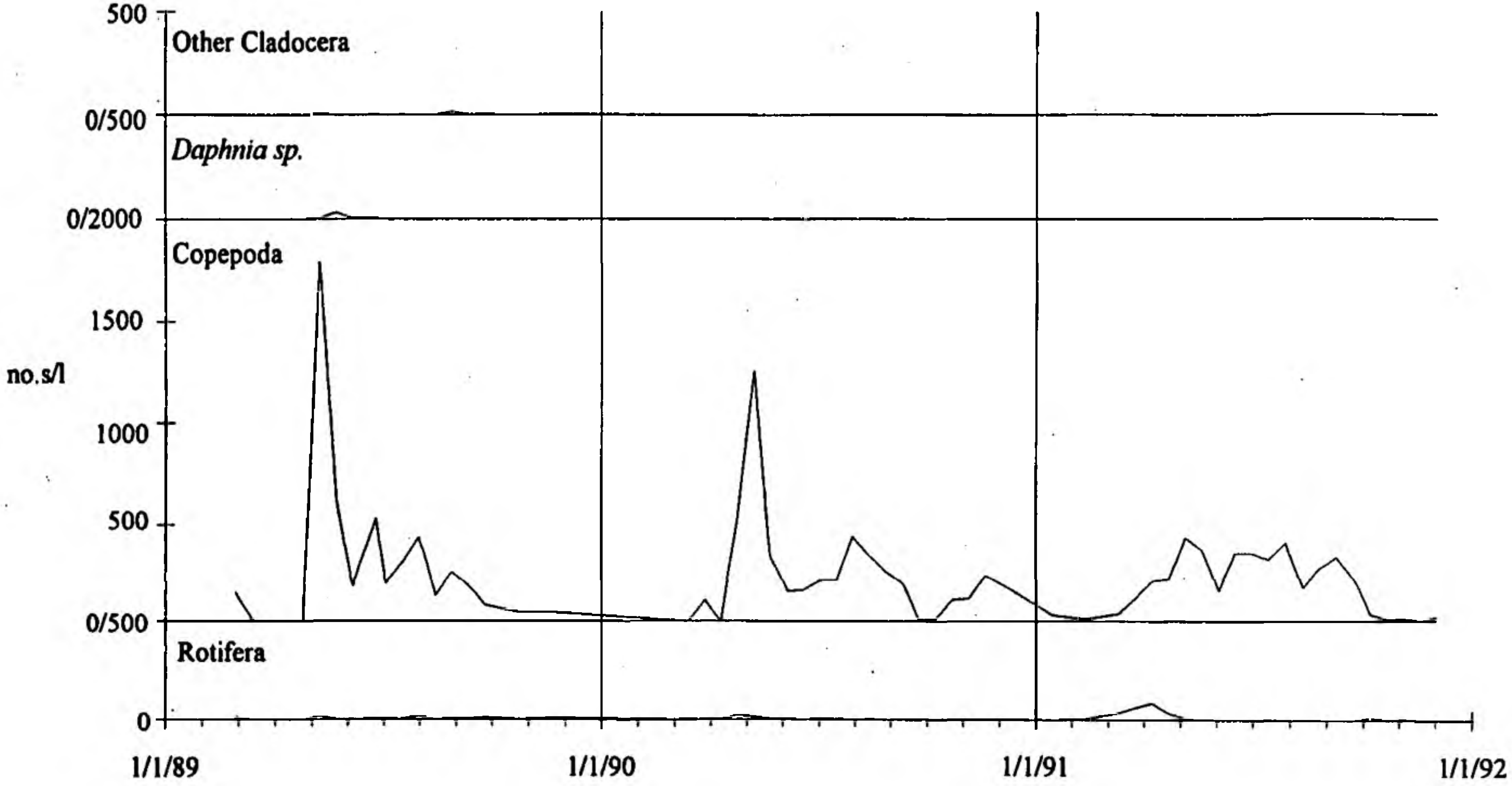


Figure 2.26 South Walsham: zooplankton population.

are probably due to the combination of the release of phosphorus from the sediment and the inputs from the S.T.W.. During 1989 and 1990 the level of S.R.P. detected in the water column fell, however, the chlorophyll levels remained high throughout most of the year. The reduced river flows during these years will increase the retention time of the broad and thus increase algal productivity which will cause a depletion of the available phosphorus. The concentration of S.R.P. continued to decline during 1991, causing an overall lowering of the amount of Total P.. The concentration of chlorophyll a was maintained at a high level throughout the year. Cladoceran numbers are low and therefore have little effect upon the level of chlorophyll (Figure 2.26).

2.5 Conclusions

The effect of phosphorus removal at the S.T.W.'s upon the River Bure and its Broads is difficult to assess. In the non-tidal stretch of the river there is a relationship between the phosphorus load discharged from the S.T.W.'s and the nutrient load in the river. The reduction in phosphorus concentration at Wroxham Broad after the initiation of phosphorus stripping in 1986 is clear. This reduction is also reflected in the lowering of the phytoplankton numbers. However, further down the River Bure these changes are difficult to assess due to such factors as the release of phosphorus from the sediment and tidal mixing. The continued reduction in the supply of phosphorus to the River Bure and associated Broads is important if any major changes are to take place.

3. LONG TERM EXPERIMENTS AT ALDERFEN AND COCKSHOOT BROADS

3.1 Introduction

We have continued to moniter the long term experiments at Cockshoot and Alderfen Broads. Alderfen Broad was dammed off from its single, effluent-rich inflow stream in 1979 but no other treatment was applied. Cockshoot Broad was dammed off from the eutrophicated River Bure in 1982 and is now fed by a good quality inflow stream which passes first through an alder wetland. The Broad discharges through a one-way sluice. Sediment was removed from most of Cockshoot Broad to create a 1m water depth at the time of damming and fish have been removed from the Broad since the winter of 1988. For both Broads, changes in water chemistry, phytoplankton and zooplankton have been followed almost continuously throughout the period since treatments began.

3.2 Alderfen Broad

Since damming, Alderfen Broad has shown a rise and fall and then a rise again in the biomass of its submerged plant community (largely *Ceratophyllum demersum*). At present it is probably just past the peak of the cycle. The previous peak was in 1982. For much of the period since 1979 the Broad has had very low inorganic nitrogen concentrations owing to its isolation from its catchment and very high soluble reactive and total phosphorus concentrations due to internal release from the sediment. It is a very shallow Broad and its fish stock has dwindled, perhaps due to periodic overnight deoxygenation which led in 1991 to an obvious fish-kill.

In 1991 the Broad behaved in a consistent way to that in immediately previous years. Phosphorus concentrations were relatively high early in the year but reached very high values from May onwards followed by little dilution in the autumn of what proved to be a meteorologically dry year. A rather similar pattern in ammonium concentrations testifies to the similar sedimentary origin of this ion (Fig. 3.1).

After a small spring peak of phytoplankton, algal crops stayed low in very clear water for the rest of the year whilst submerged plant biomass at its maximum in late summer was a little lower than that in 1990. The relative abundances of two *Daphnia* species (*D.magna* and *D.hyalina*)(Fig. 3.2) explain the low algal crops despite high nutrient availability and are undoubtedly related to the reduced fish stock. Reasons for the cycle in plant abundance are not clear but probably related to physiology of *Ceratophyllum* itself as they do not correlate well with any extraeous biotic or non-biotic factors. As it is not possible to carry out a detailed programme to investigate this phenomenon, and because the water depth in the Broad has become very low, it is recomended that a programme of rehabilitation by sediment removal be now begun but that the isolation of the Broad be retained.

3.3 Cockshoot Broad

A very similar pattern has been maintained in Cockshoot Broad to that in 1990, with the effects of the dry weather superimposed. More saline water may occasionally enter Cockshoot Broad by overtopping of the dams at very high tides. This has only negligable effects on nutrient loadings but does increase the chloride concentrations to several hundred mg per litre for a period. Inputs of chloride in winter 1990 were only slowly diluted by the stream water and remained at about double the normal background

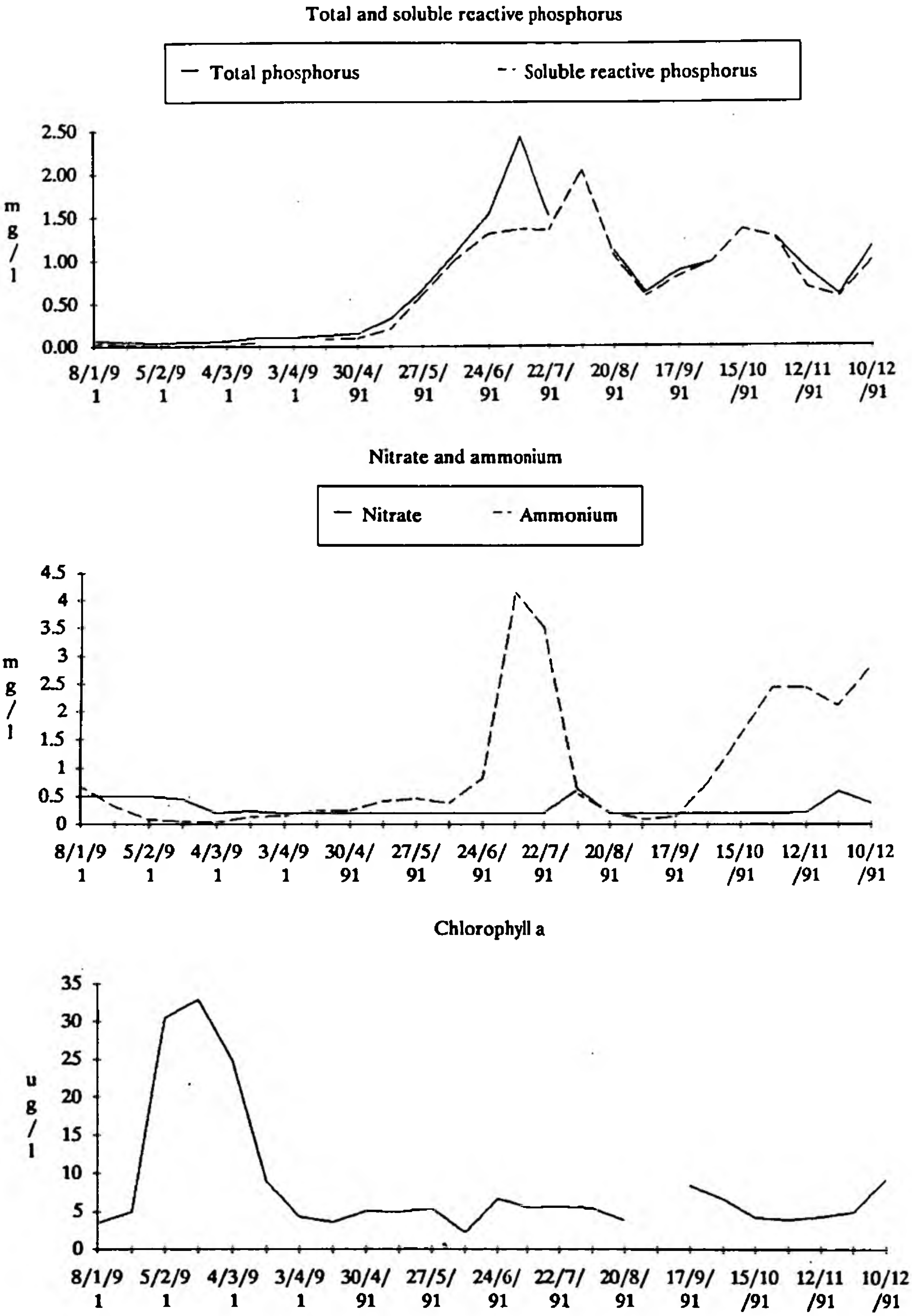


Fig 3.1 Concentrations of total and soluble reactive phosphorus, nitrate, ammonium and chlorophyll in Alderfen Broad 1991

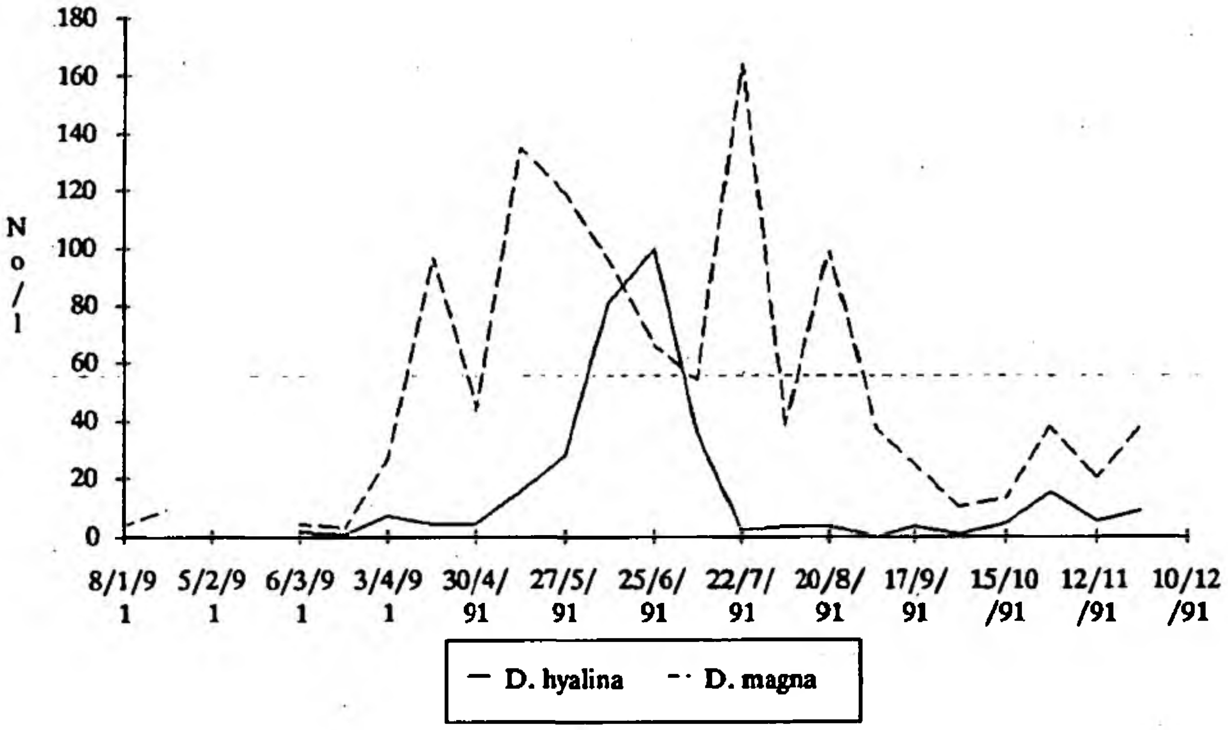


Fig 3.2 Daphnia populations in Alderfen Broad 1991

values for the Broad during the rest of the year. They increased again in the late autumn (Fig. 3.3).

Phosphorus concentrations showed a pattern, which has been established over the past four years, of major internally-derived increases in concentration, largely of soluble reactive phosporus. These appear to originate in the long thin dyke section of the Broad, which has had well established communities of aquatic plants for many years, but are reflected in concentrations in the main Broad as the summer progresses (Fig. 3.4). Ammonium-N concentrations rose a little in the dyke during the summer, but only to very modest peaks and inorganic nitrogen was scarce in the Broad and dyke throughout the period. This was not due to any shortage of nitrate input from the catchment to the feeder stream, but largely to denitrification of the stream water as it passed through the alder carr on its passage to the Broad.

Phytoplankton populations were high in the spring prior to the temperature related development of the large *Daphnia* populations but fell to negligable values (<5ug per l)for the summer period and the water was extremely clear. The large species *D.pulex* was present, perhaps supported by the refuges provided by the water lilies. In autumn chlorophyll a concentrations rose, as is normal as *Daphnia* seasonally declined (Fig. 3.5) The high *Daphnia* populations are related to the very low fish stock maintained in the Broad and create a light climate that is favourable for submerged plant development.

Submerged plants, however, have been slow to develop in the main Broad and experiments elsewhwere suggest that this may be due to bird grazing of the initially sparse colonizers. In the dyke an early (1983) development of plants was aided perhaps by the sheltered conditions there despite grazing. The flora in the dyke was, however, less well developed in 1991. The water lilies continued to spread but populations of *Elodea canadensis* and *Ceratophyllum demersum* were unusually sparse. Reasons for this are unclear as otherwise conditions were similar to the previous years in which these species thrived. It is possible that a cold, late spring delayed their growth such that the balance between growth and grazing was displaced in favour of the latter at a critically early time.

It is recommended that the Cockshoot experiment be continued but with establishment of plant inocula in the main Broad protected in the early years from grazing by coot and geese.

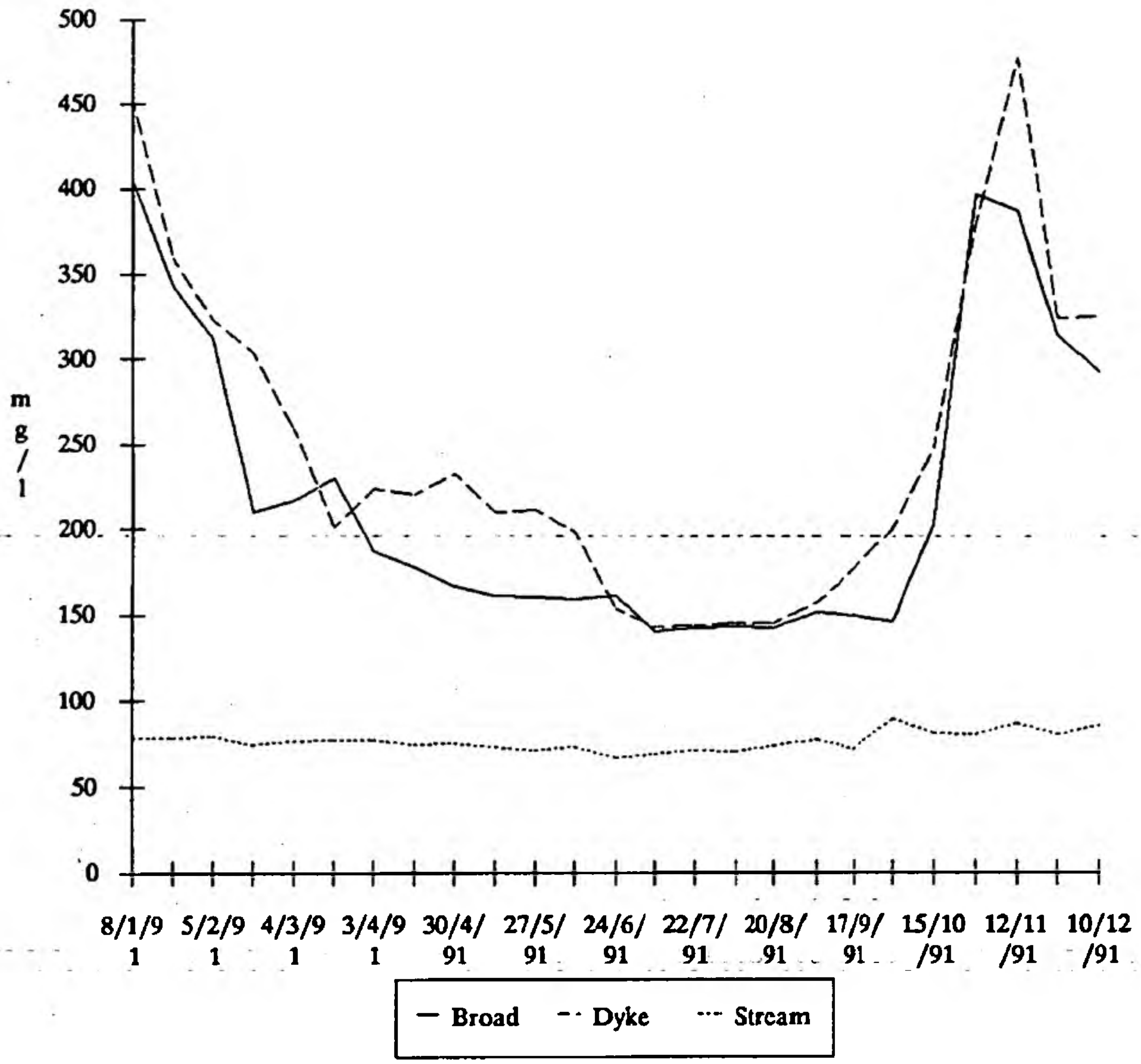


Fig 3.3 Chloride concentrations in Cockshoot Broad, Dyke and Stream 1991

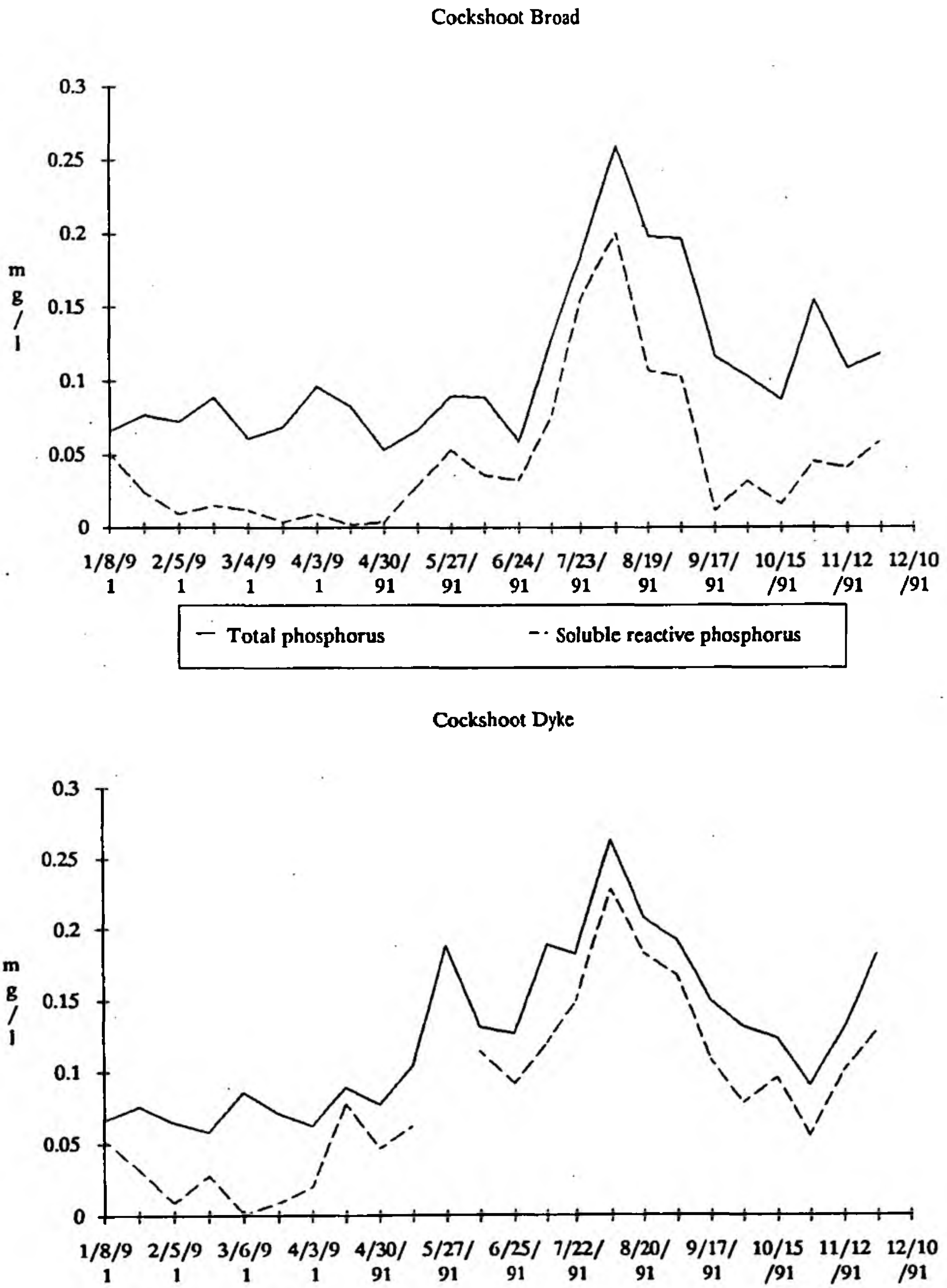
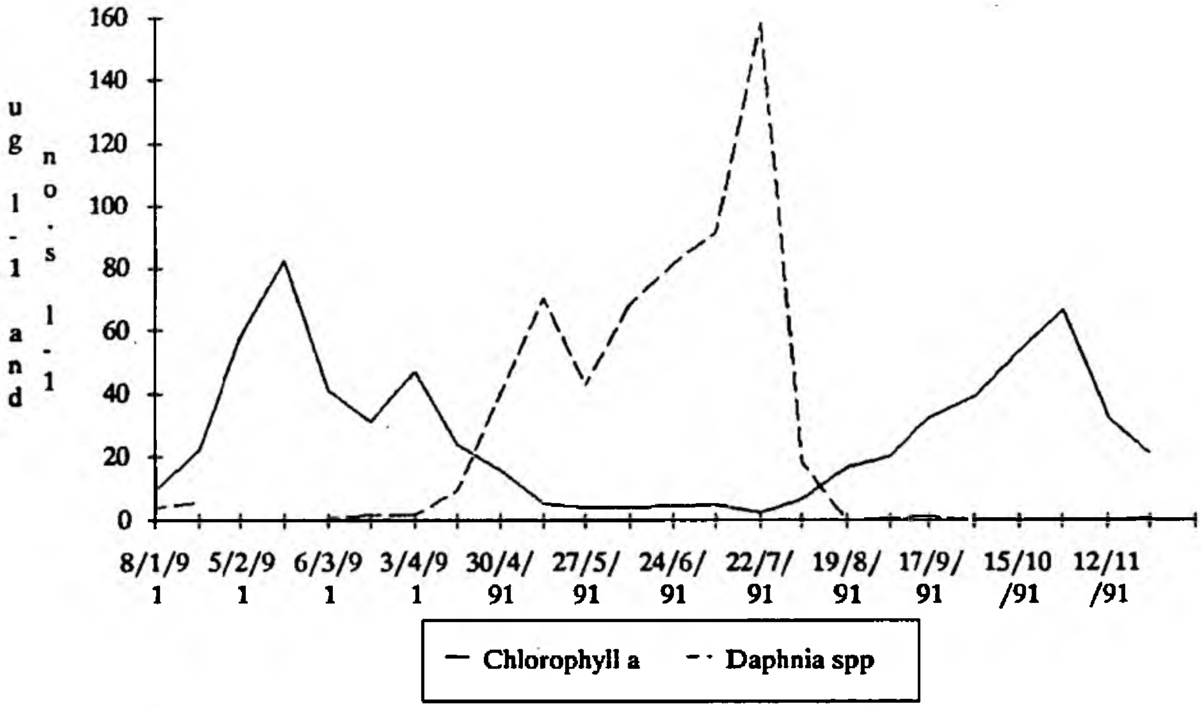


Fig 3.4 Total and soluble reactive phosphorus in Cockshoot Broad and Dyke 1991

Cockshoot Broad



Cockshoot Dyke

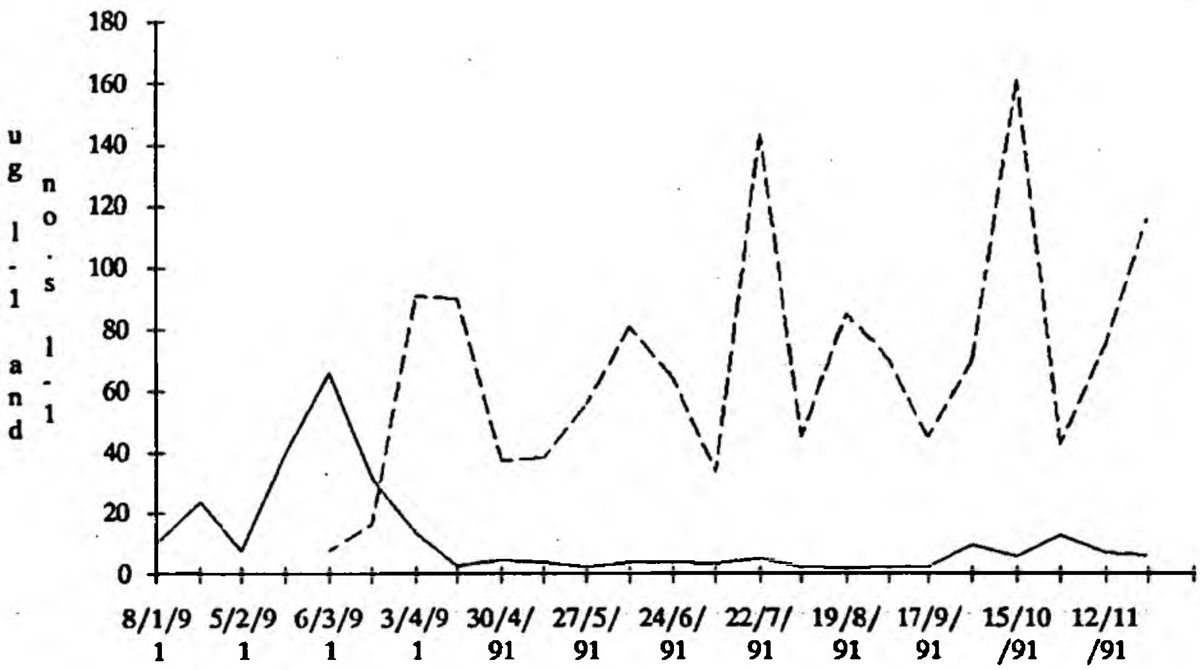


Fig 3.5 Chlorophyll concentrations and Daphnia populations in Cockshoot Broad and Dyke 1991

4. HOVETON GREAT BROAD FISH EXCLOSURE EXPERIMENT

4.1 Introduction

The engineering works in Hoveton Great Broad to construct a fish-proof barrier were completed in May 1990 (see Figs 4.1 and 4.2). Because fish had spawned early that year, it was not possible to remove them until winter 1990/1, allowing 1990 to act as a control year. The removal of fish in January and Febuary 1991 marked the resumption of the experimental programme and should have produced clear water in the exclosure during the summer. However, *Daphnia* failed to multiply as expected and chlorophyll levels were similar to the main Broad.

At first it was thought there had been an incomplete removal of fish, leaving a few adults which had produced large numbers of zooplanktivorous fry. Fish were occassionally spotted within the exclosure and in July two dead adults, one roach and one bream, were found. In early September, an operation to assess the fish stock was carried out by Dr. Martin Perrow. He found large numbers of fish, both adults and young. An experiment was also carried out at this time in which two small fish-proof cages were constructed in the exclosure and stocked with zooplankton. Large numbers of *Daphnia* grew in both, illustrating that food supply and other conditions were adequate.

In late September, an examination of the base of the exclosure wall was made and it was found that a hole had been scoured underneath over a short section. This was obviously the source of the large numbers of fish found in Dr. Perrows survey. Plans were made to take professional advice and repair the hole ready for a fish removal operation in time for the next years spawning season. It transpired that the panels used in this section of the wall had had to be shortened because the sediment was harder here and they could not be pushed all the way down. In Febuary 1992 an engineering firm, Wetland Works, installed boards the full 3m length alongside the original panels over the length of the breach.

4.2 Results

4.2.1 Water chemistry

The results of analyses of soluble reactive phosphorus (SRP), total phosphorus (TP), nitrate and ammonium for 1990 and 1991 are shown in Fig 4.3. Monitoring inside the exclosure was discountinued after the hole was discovered in September. The results from inside and outside the exclosure are very similar in 1991.

Concentrations of SRP were generally low, with no apparent sediment release, although immediate uptake by algae can prevent the detection of SRP release. In the summer months the concentration was mostly near or below the limit of detection ($0.004 mgl^{-1}$). In 1990 release was detected on one date in July.

The pattern for total phosphorus closely follows that of chlorophyll, algal cells making up the greater part of particulate phosphorus in the broad. Values were similar this year to last, rising to a peak in late summer and low during the winter.

The detection limit of nitrate and ammonium analyses has changed during the period of this experiment. In 1990 the analyses were carried out at the NRA laboratory at Peterborough with detection limits of 0.5 and 0.04 mgl^{-1} for nitrate and ammonium

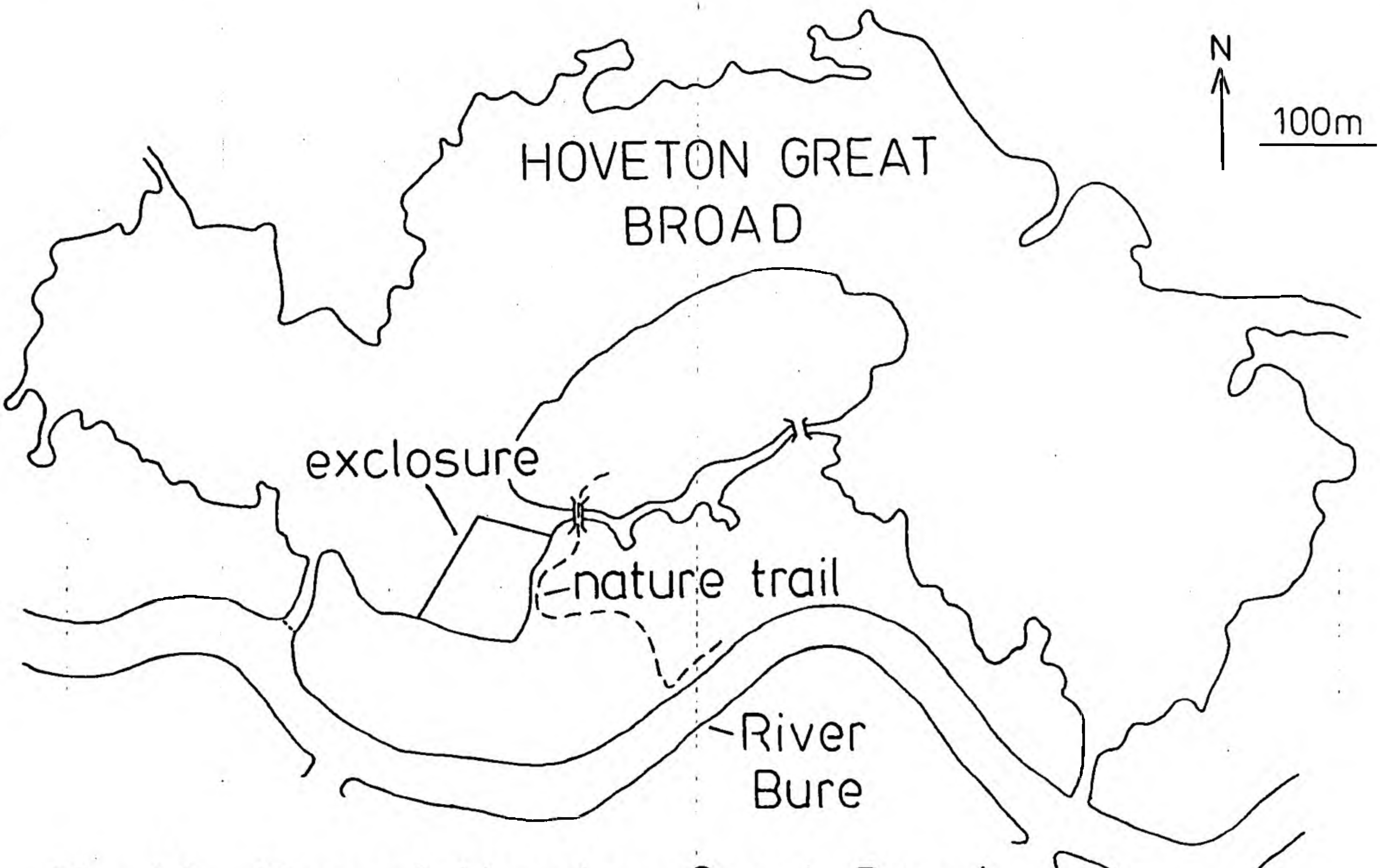


Fig 4.1 Map of Hoveton Great Broad

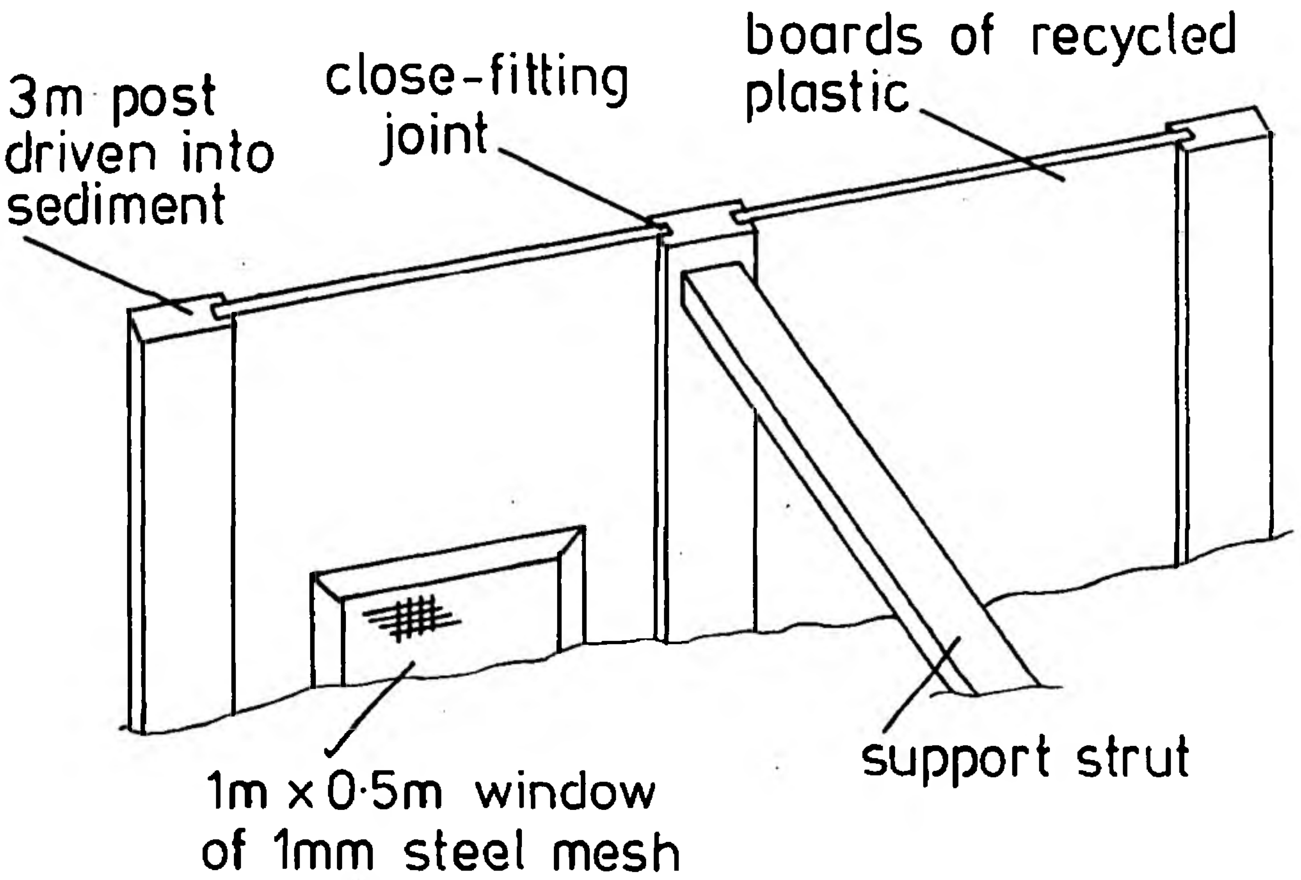


Fig 4.2 Simplified diagram of exclosure wall

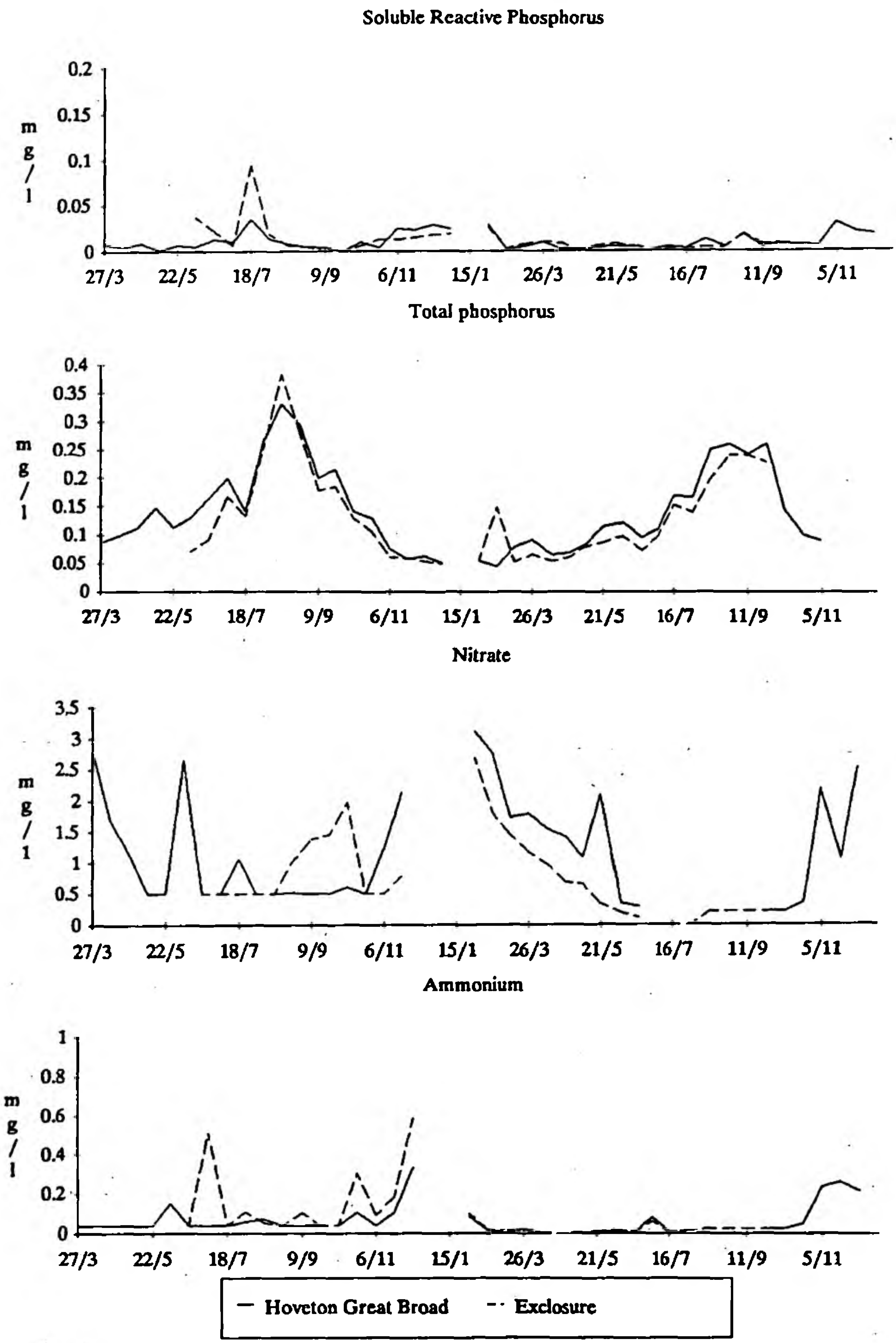


Fig 4.3 Concentrations of total and soluble reactive phosphorus, nitrate and ammonium in Hoveton Great Broad and exclosure 1990-91

respectively. Between January and August 1991 the work was carried out at Haddiscoe lab with detection limits of 0.1 and 0.01 mgl^{-1}. When Peterborough took over again detection limits had been revised to 0.2 and 0.023 mgl^{-1} respectively. Results below the limit of detection are represented at the limit of detection.

Nitrate concentrations are highest during winter, falling gradually to below the limit of detection in April or May and rising again in November. In 1990 the pattern in the exclosure was slightly different with excess nitrate present in August-October. In 1991 concentrations were more similar between the two sites, with values below the limit of detection in the summer and early autumn.

Ammonium shows a similar annual pattern though the period of higher winter concentrations ends earlier, in Febuary. Concentrations in the exclosure in the summer of 1990 were eratic and somewhat higher than the main broad. This year concentrations were at or below the limit of detection in both sites from Febuary to November.

The results of chlorophyll analysis are shown on Fig 4.4. The annual trend for both sites is a winter minimum from November to January, values of roughly 50-100 ugl^{-1} from Febuary to July and a peak of 200+ ugl^{-1} in August and September. Results from the two sites were very similar in 1991, with a lower peak than 1990 (approx. 300 ugl^{-1} compared to approx. 500 ugl^{-1}). There was a period of clear water in the exclosure in June 1990.

4.2.2 Phytoplankton

Phytoplankton data has been summarised by grouping the species into blue-greens, all other species and within these, diatoms. Fig 4.5 shows the results for 1990 and 1991. Numbers of blue-green algae were lower this year than last and numbers of other algae, particularly diatoms, higher during the same period. Between the two sites, results were again more similar in 1991 than 1990. In 1990 populations were lower in the exclosure during the period of clear water, and the number and proportion of blue-greens in the late summer was higher than the main broad. In spring this year phytoplankton populations were somewhat higher in the main broad than the exclosure, though chlorophyll concetrations were very similar.

4.2.3 Zooplankton

Zooplankton species have also been divided into groups: *Daphnia* species, copepods (comprising adults and immature copepodites), and small species (nauplii, *Bosmina* and Rotifers). The significance of these groups is in their susceptibility to predation by fish and their effect on phytoplankton populations. *Daphnia* species are highly susceptible to predation and highly effective at reducing phytoplankton numbers. Copepods are somewhat resistant to predation due to their fast swimming speed and are predatory rather than herbivorous. The diminutive body size of small species makes them both unattractive as fish food and ineffective as phytoplankton grazers. Therefore in the presence of fish we expect a community dominated by copepods and small species, in which phytoplankton can flourish. In the absence of fish predation *Daphnia* outcompete the other species due to their greater efficiency and can reduce phytoplankton to low levels.

Results for 1990 and 1991 are plotted in Fig 4.6. The zooplankton communities of the broad and exclosure are very similar during 1991, with copepods and small species dominant over *Daphnia*. In both sites there is a small peak of *Daphnia* in June,

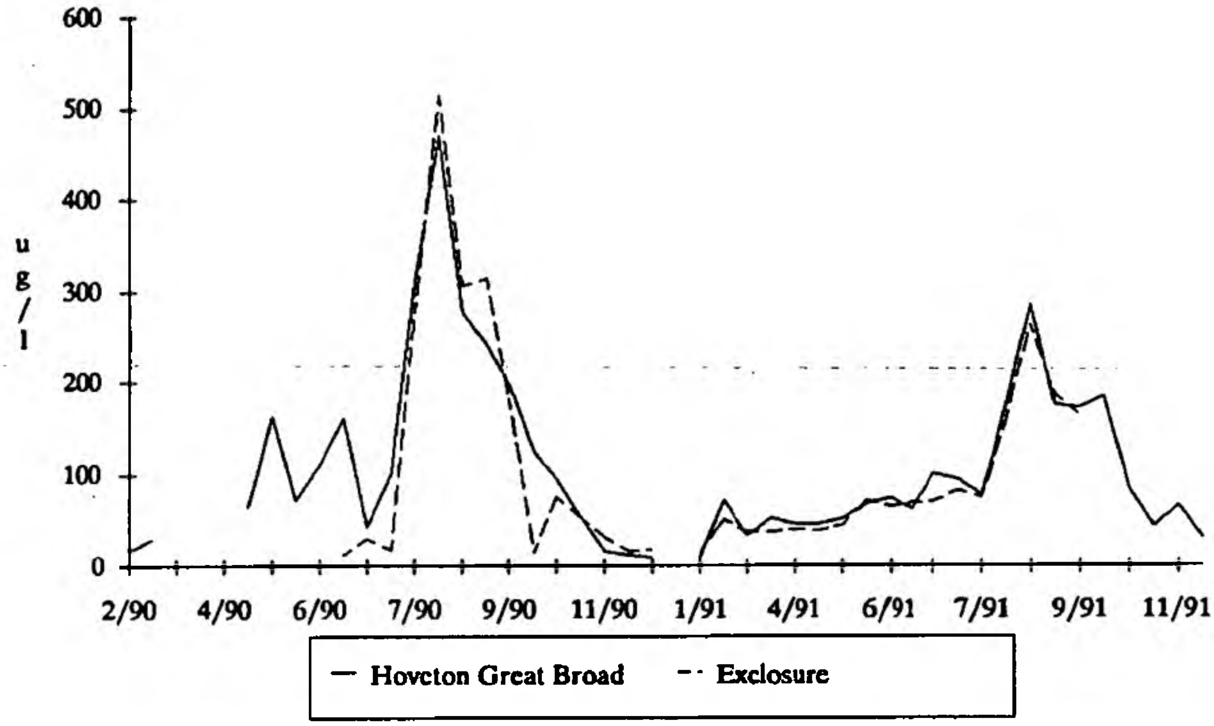
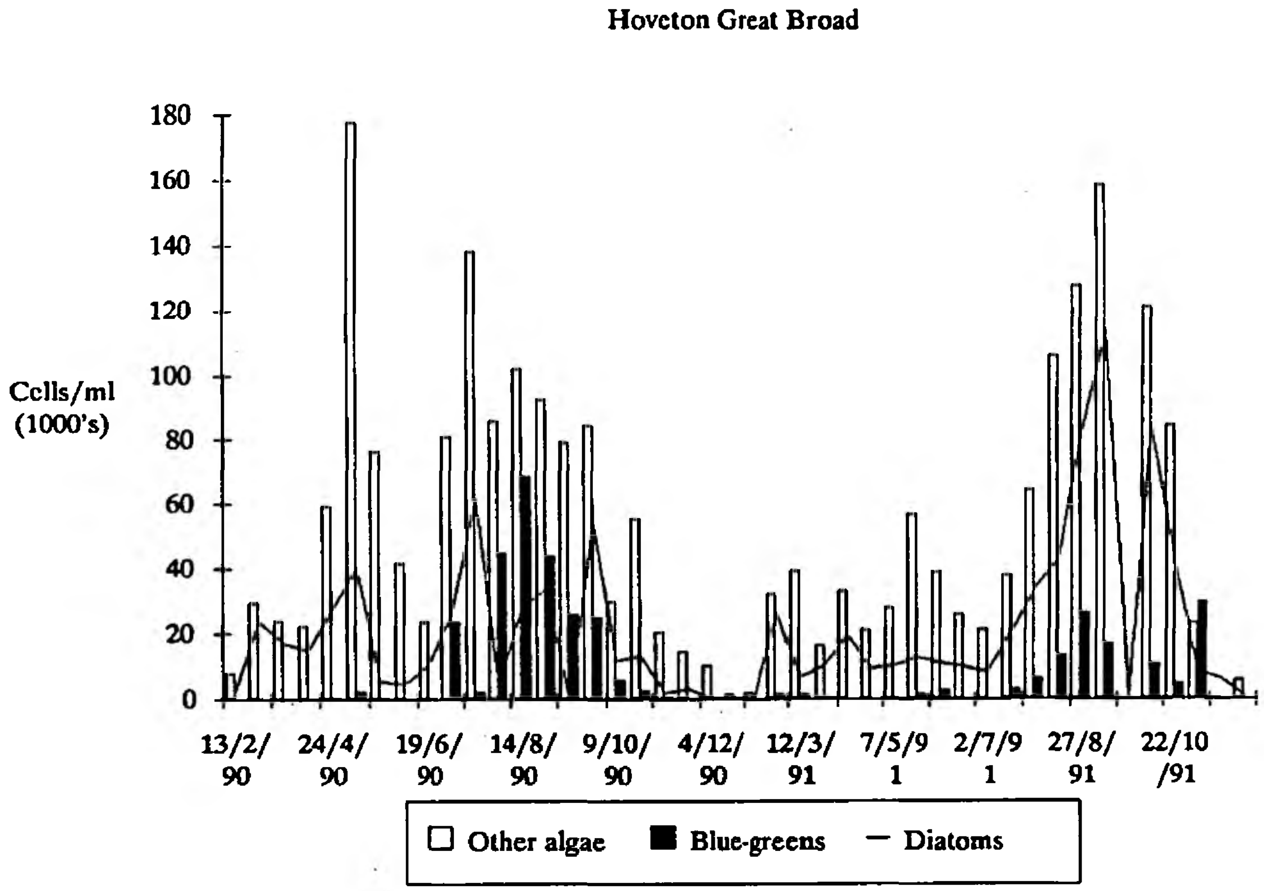


Fig 4.4 Chlorophyll concentrations in Hoveton Great Broad and exclosure 1990- 91



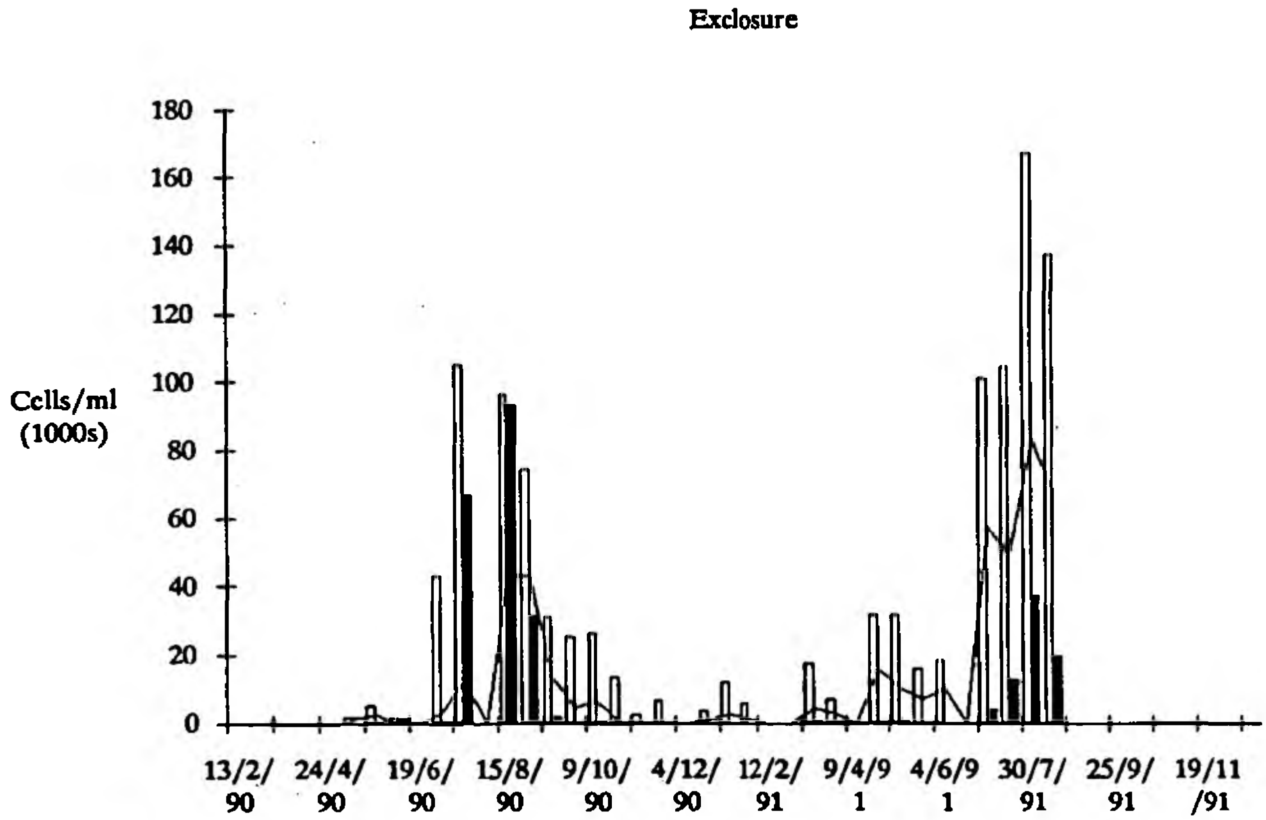


Fig 4.5 Phytoplankton populations in Hoveton Great Broad and exclosure 1990- 91

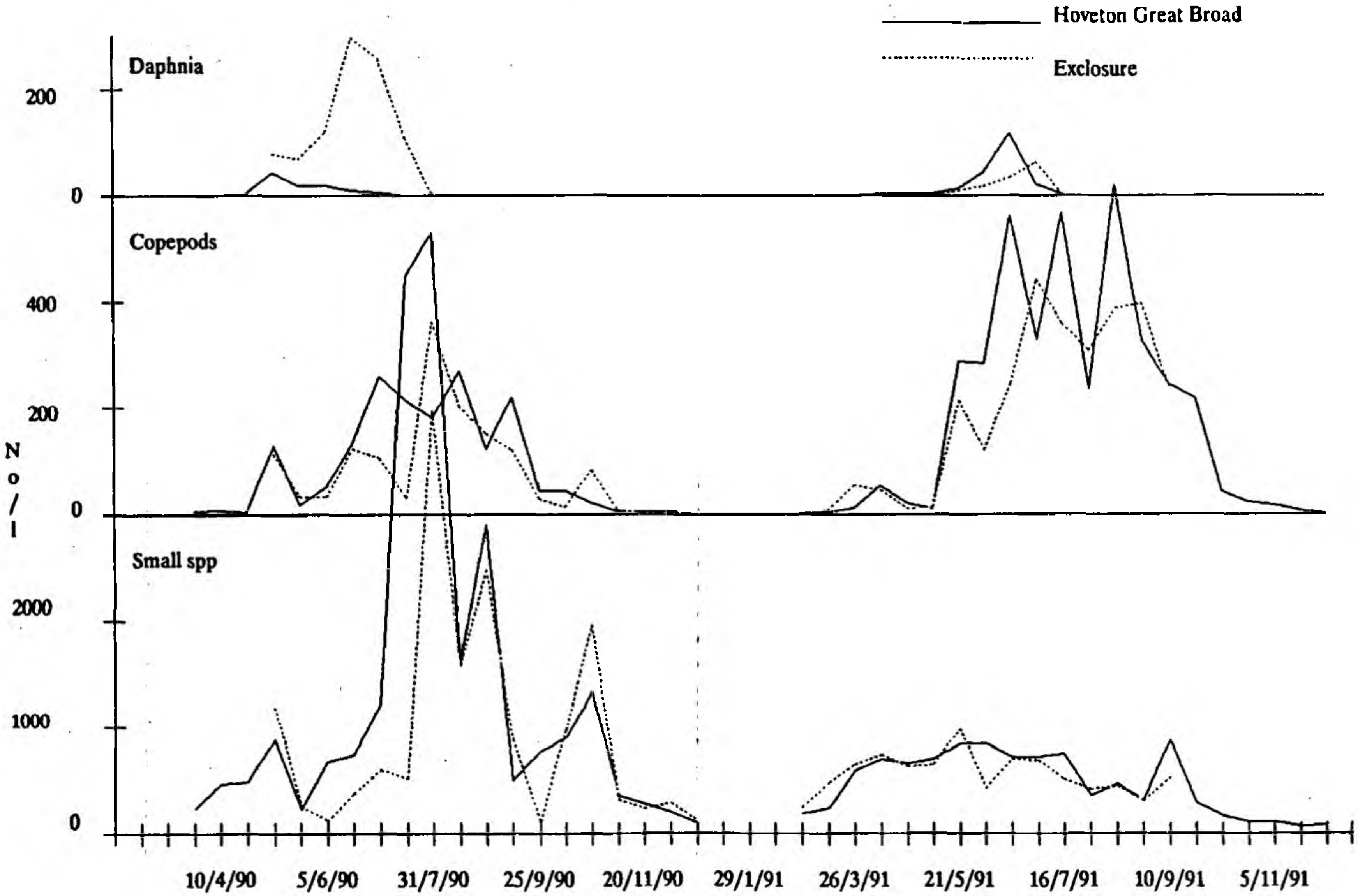


Fig 4.6 Zooplankton populations in Hoveton Great Broad and exclosure 1990- 91

contrasting to last summer when there was an extended period of high numbers in the exclosure. Copepods show the typical pattern of summer abundance, with somewhat higher numbers than 1990. The population of small species showed a peak in 1990 during July and August which was not present in 1991.

4.3 Barrier breaching

We do not know precisely when the barrier was breached, but an examination of the chemical and biological data can provide some clues. We might expect greater differences between concentrations of chemicals inside and outside the exclosure before the hole formed than after it, due to the change in flushing rate. For SRP, nitrate and ammonium there is generally more discrepency during 1990 than 1991 (see Fig 4.3). Chloride concentrations vary little except during tidal surges, of which there was one in September 1990 (see Fig 4.7). On the sampling date the concentration of chloride in the exclosure was about twice that in the main broad. Such a large difference would probably not be expected if the hole had formed by then.

Zooplankton populations show differences up to the end of July 1990 and thereafter considerable similarity (see Fig 4.6). In particular the disappearance of *Daphnia* in July could indicate the influx of fish. However, as this is before the tidal surge discussed above, it is unlikely the hole formed as early as this.

Benthic animals show two possible dates: In January 1991 numbers of chironomids in the exclosure fall from a level higher than that in the main broad to the same level and thereafter follow the same trends. Numbers of oligochaetes stay higher than the main broad until July. The evidence from the chemical data co-incides with the former of these events, making November 1990-January 1991 the most likely candidate for a period when the barier was breached. There was a second tidal surge in late December or early January (not covered by our sampling) which may have been the final impetus to break through the sediment.

4.4 Conclusions

The effects of the breaching of the barrier can clearly be seen in the results of chemical and biological monitoring. During 1990 flushing rate was slightly reduced in the exclosure compared to the main broad and this allowed small differences in concentrations of nutrients to exist and higher numbers of blue-green algae. In comparison, with a higher flushing rate in 1991, nutrient concentrations and blue-green algae numbers were very similar between the two sites. During 1990 there was a relatively low fish stock trapped inside the barrier in contrast to 1991 when fish were free to move in and out from the main broad. This is responsible for the different zooplankton communities in the exclosure, with Daphnia dominant until mid July in 1990, but very low numbers throughout 1991.

The breach in the barrier has meant that 1991 has essentially been another control year. The results are more valid as a control because the fish population is the same as in the main broad, but less so because flushing rate is higher than when the barrier is intact.

The experience of this year has been valuable in learning about precautions to take in constructing such barriers in tidal broads, which may be very useful in the future if this method is applied on a large scale.

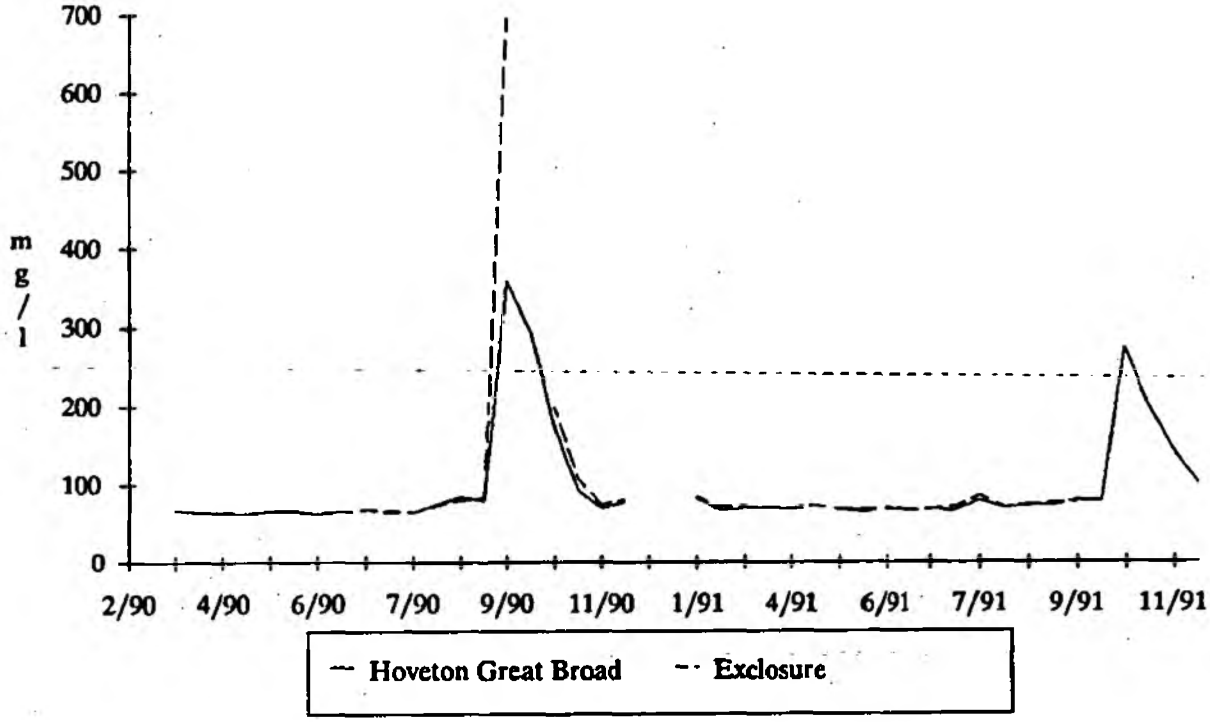


Fig 4.7 Chloride concentrations in Hoveton Great Broad and exclosure 1990-91

5. POUND END

5.1 Introduction

The barrier between Pound End and Hoveton Little Broad (see Fig 5.1) was completed in April 1990 and the fish removed by May. Daphnia numbers increased in the late summer and produced partial clearing of the water. However, in common with Hoveton Great Broad, the water remained cloudy in 1991 and the expected swarms of Daphnia did not materialise. Again the reason was found to be a breach in the barrier, this time discovered in May this year.

Due to the large volume of water that is exchanged each tide through the narrow entrance, it was decided that the existing design would have to be replaced with steel piling. Mesh windows have also been provided in this structure. Piling operations were completed at the end of September.

5.2 Results

Results for soluble reactive phosphorus (SRP), total phosphorus (TP), nitrate and ammonium are shown in Fig 5.2. Concentrations are very similar between Pound End and Hoveton Little Broad for all determinands in 1991, less so in 1990. Soluble reactive phosphorus is low throughout 1991, only rising appreciably above the level of detection in the winter months. In contrast there was excess SRP in Pound End during the period of high Daphnia numbers and in the autumn.

Total phosphorus largely follows the pattern of chlorophyll concetrations, as at Hoveton Great Broad. 1991 results are broadly similar to 1990. Nitrate and ammonium are also similar to Hoveton Great Broad with highest levels in the winter months and low (below limit of detection) during the summer. Concentrations of nitrate are consistently lower in Pound End. Ammonium concentrations were very similar between the two sites in 1991, with somewhat higher values in Pound End in 1990.

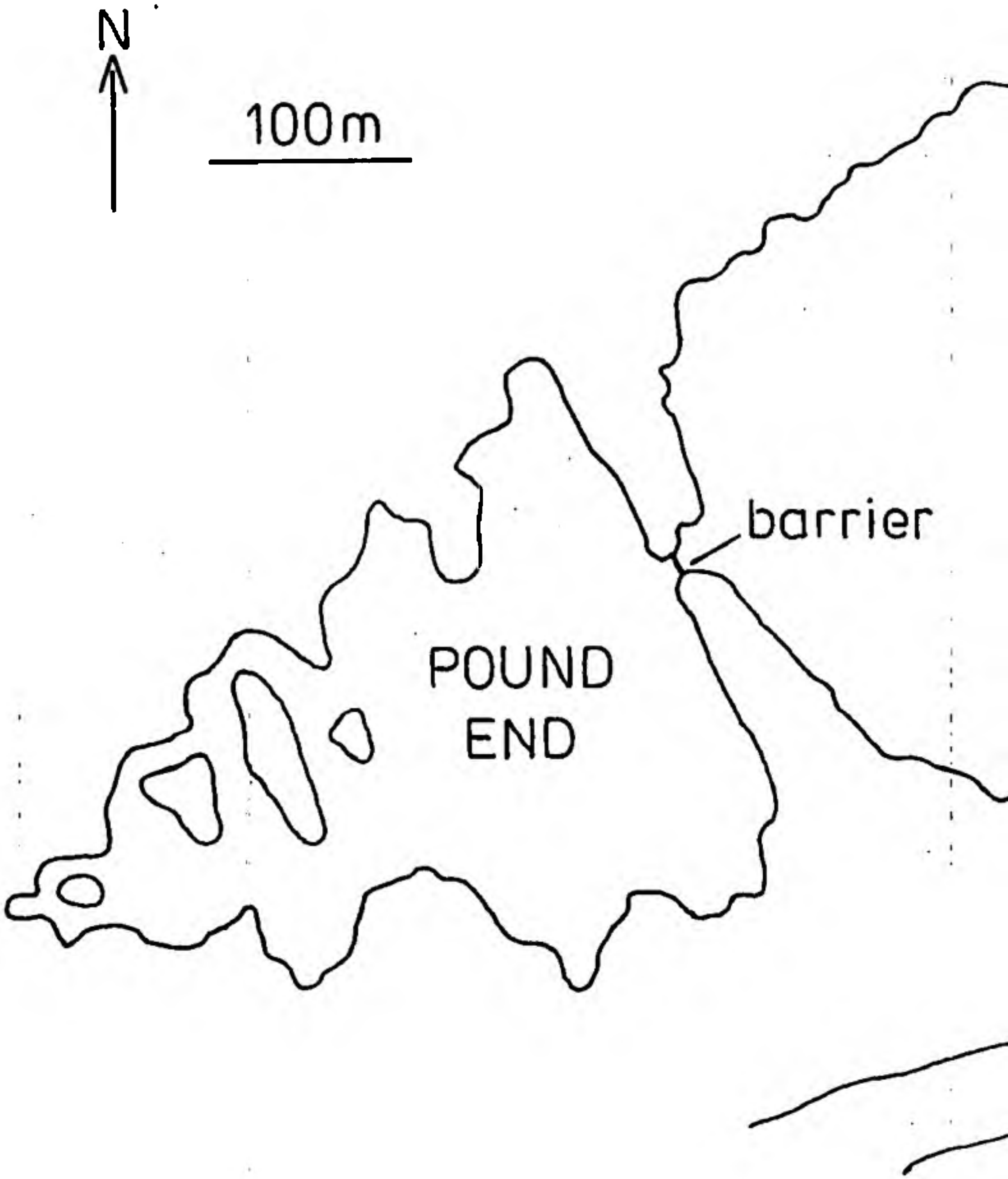
Chlorophyll concentrations show spring and autumn peaks in 1991 (see Fig 5.3), both of just over 100ugl^{-1}. In late June and July values are lower, at around 50ugl^{-1}. This is in contrast to 1990 when the Broad had no definite spring peak or early summer clear period and reached a peak of 270ugl^{-1} in August. Pound End had much lower concentrations than Hoveton Little Broad in 1990 with clear water in July.

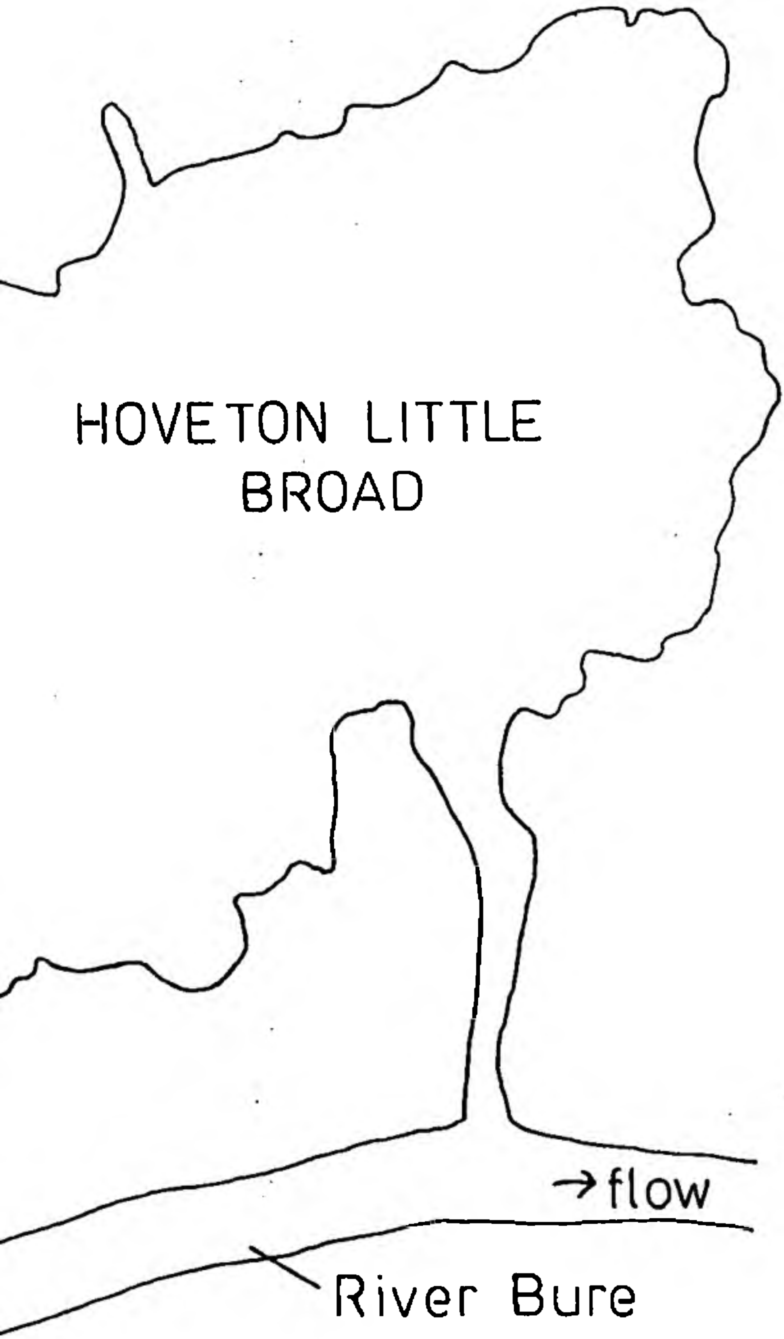
Zooplankton populations, again shown in the groups *Daphnia*, copepods and small species, are presented in Fig 5.4. Identical small peaks of *Daphnia* occur in Pound End and Hoveton Little Broad in June this year. A similar peak occured earlier in 1990 in Hoveton Little Broad, but in Pound End considerable numbers were present from July to October. The pattern for copepods and small species is very similar to Hoveton Great Broad, copepods again occuring in higher numbers than last year and small species lower. The difference in small species is mainly due to Rotifers.

5.3 Conclusions

The situation in Pound End is very similar to that in Hoveton Great Broad. Again 1991 has provided valuable lessons in the design of fish-proof barriers in a tidal system. The results generally show differences between the period when the barrier was intact and while it was breached. When fish stocks were low in 1990, *Daphnia* flourished leading to

Fig 5.1 Map of Pound End and Hoveton Little Broad




HOVETON LITTLE
BROAD
→flow
River Bure

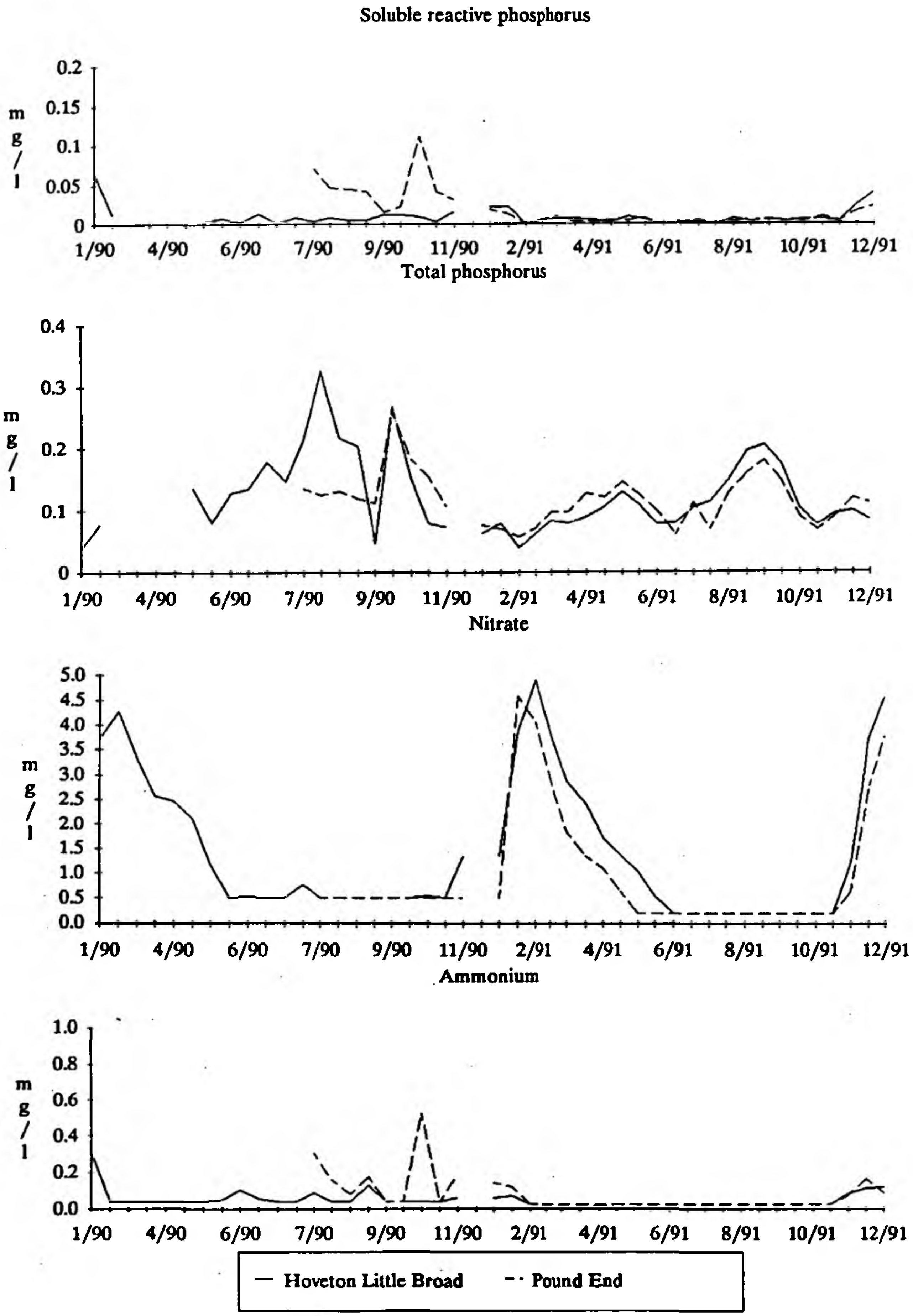


Fig 5.2 Concentrations of total and soluble reactive phosphorus, nitrate and ammonium in Hoveton Little Broad and Pound End 1990-91

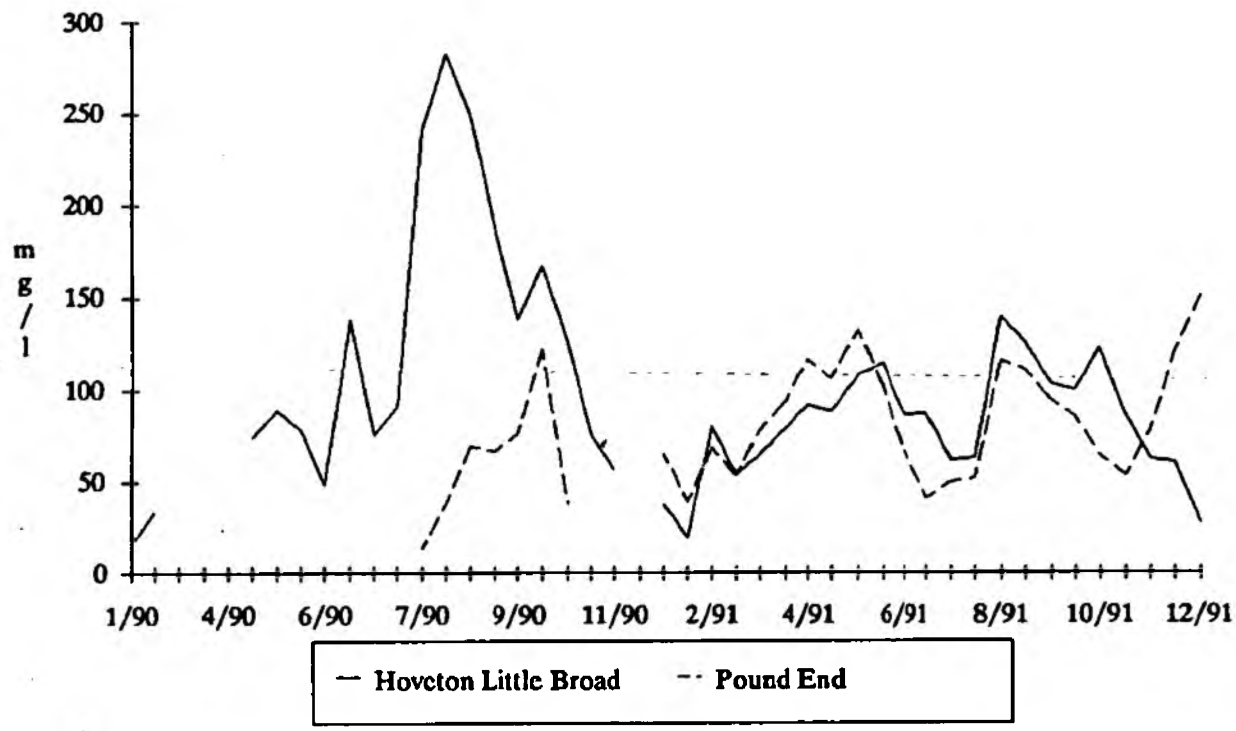


Fig 5.3 Chlorophyll concentrations in Hoveton Little Broad and Pound End 1990-91

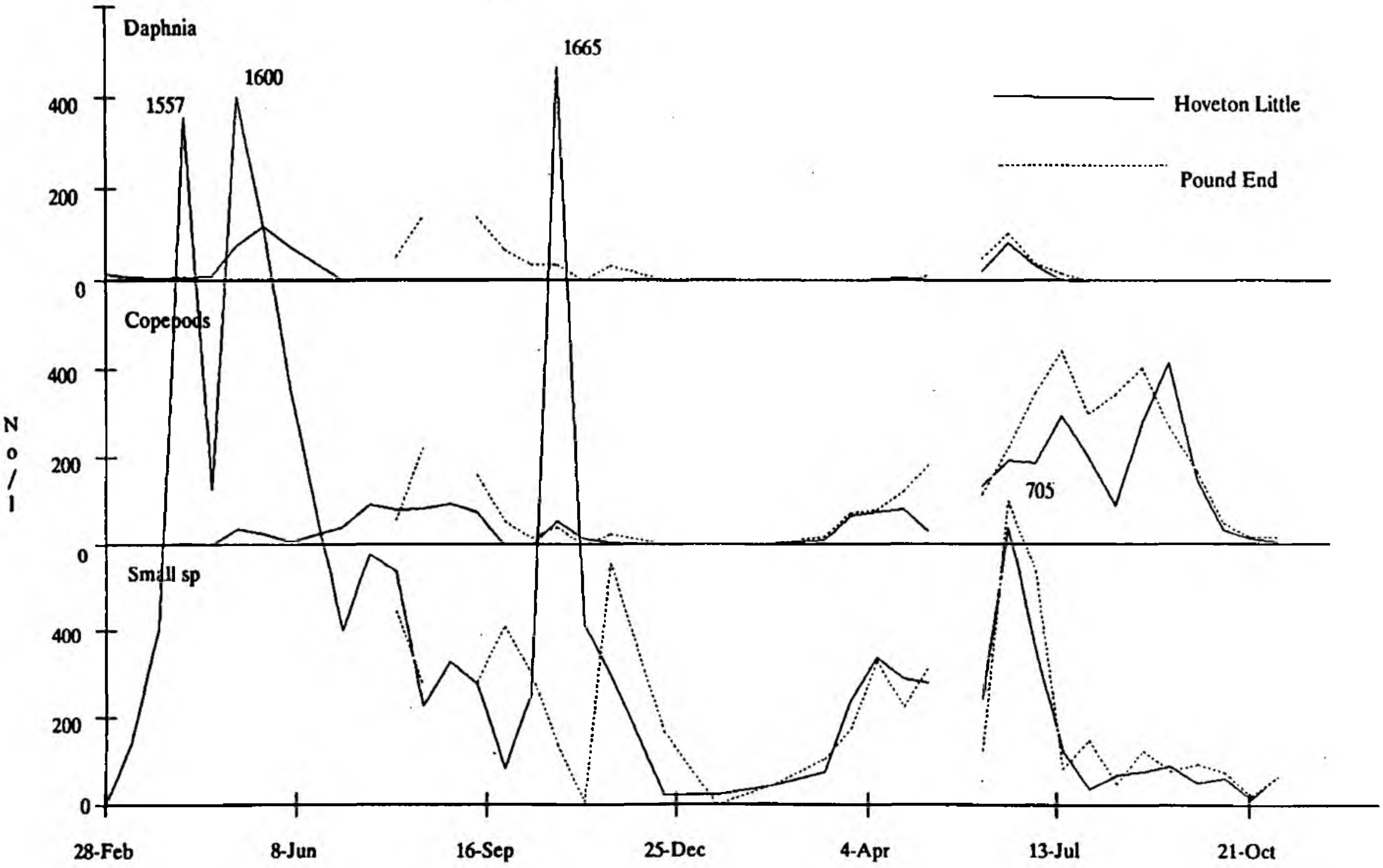


Fig 5.4 Zooplankton populations in Hoveton Little Broad and Pound End 1991

low phytoplankton populations and in turn excess nutrients. In contrast, in 1991 with the high flushing rate between the two sites and free movement of fish, results for chemistry, phytoplankton and zooplankton are very similar. Even after the barrier was replaced in late September, chemical results indicate a good flushing rate, provided by the numerous windows.

With the repair of the barrier and refishing operation (completed Febuary), 1992 should see a resumption of the restoration of this broad.

6. PLANT GROWTH EXPERIMENTS

6.1 Introduction

Two sets of experiments were carried out in 1991 to test the effects of bird grazing and water quality on the survival and growth of plant inoccula. They were designed to run for two or three years in order to test over-wintering survival.

The first experiment involved protective structures called carousels (see Fig 6.1), octagonal wooden structures 8m across. These were used in pairs, in one of which the wooden slats were closely spaced so as to exclude all birds and the top covered with wire mesh. The slats were more widely spaced in the other carousel and the wire mesh ommitted so as to allow entry to coot and other birds. Comparison between these two designs will investigate the relative importance of bird grazing and physical protection.

Pairs of carousels were placed in broads of differing water quality in terms of light penetration due to phtyoplankton populations. Belaugh Broad, Pound End and the Hoveton Great Broad exclosure (see Fig 6.2) should provide the clearest water, in the former because of high flushing rate and in the latter two due to removal of fish. Hoveton Great Broad normally has summer chlorophyll concentrations of 100+ ugl^{-1} while the final site, Ranworth (see Fig 6.2), has 200+ ugl^{-1} of chlorophyll. Several species of plant were used to mimic a Broad community within the resricted availablity.

The second experiment comprised a more detailed investigation of *Elodea* growth, again with and without protection from bird grazing and in two water quality environments - inside the exclosure and in Hoveton Great Broad. This therefore gives four treatments - protected from bird grazing in a good light climate, open to bird grazing in a good light climate, protected from bird grazing in a poor light climate and open to bird grazing in a poor light climate. Other differences, e.g. sediment chemistry, are minimised by comparing between two sites in the same broad. Wood and wire mesh structures 1.5m square (see Fig 6.3) were built to prevent bird grazing, non-protected plants were left in the open water.

6.2 Methods

6.2.1 Carousel experiments

Setup

The structures were built by the Broads Authority in April 1991, and planted up by them in May. The plants comprised approximately 30 rhizomes each of white and yellow water lilies (*Nymphaea alba, Nuphar lutea*), 40 *Elodea canadensis* plants, 60 *Ceratophyllum demersum* and 100 *Stratiotes aloides*. The submerged plants were all weighted down with bricks to allow them to root in the sediment, the *Elodea* and *Ceratophyllum* being held in a wire mesh 'sandwich' to avoid weighting the plants down individually. Planting was restricted to one half of the carousel in order to allow the plants to spread out in later years.

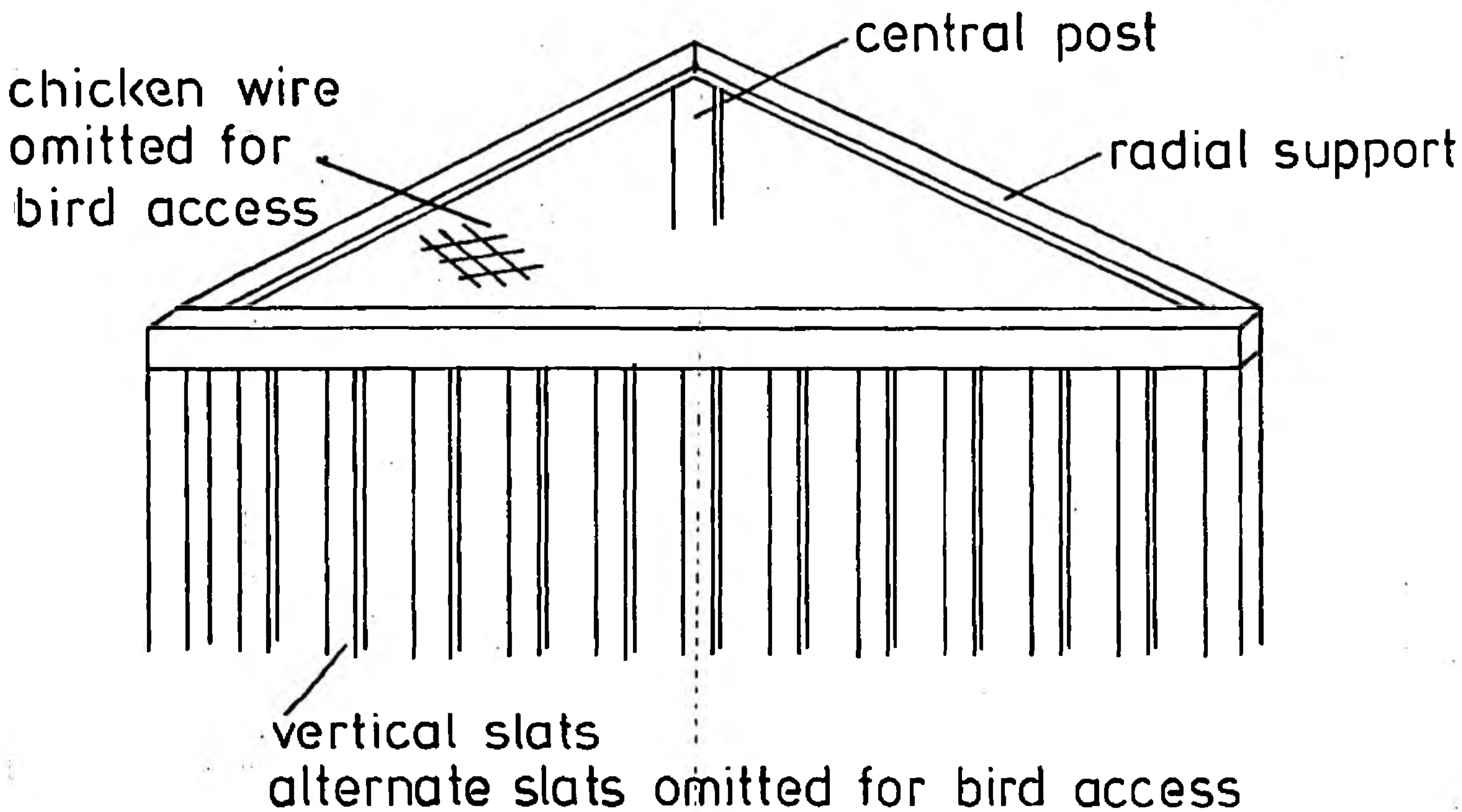
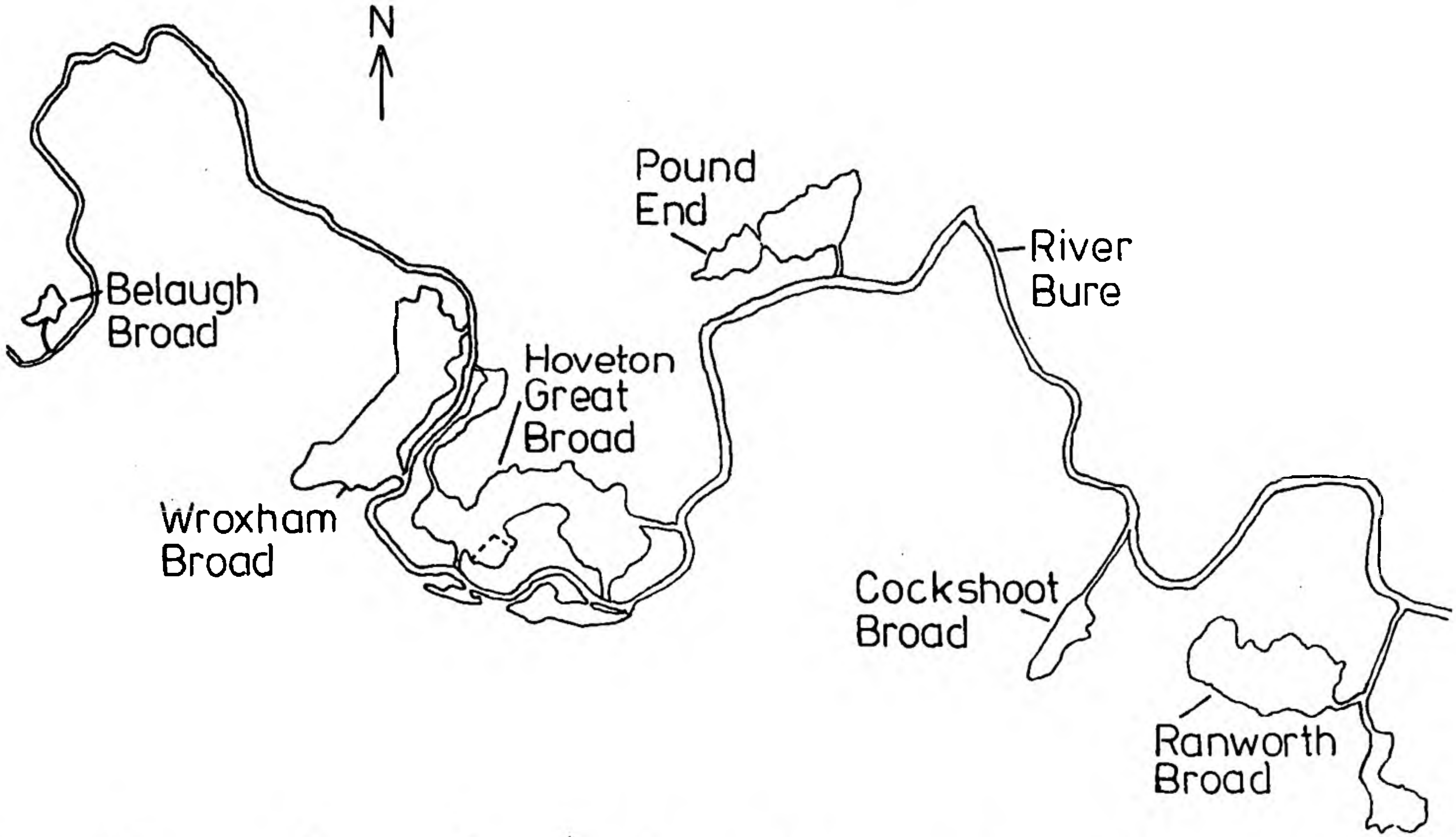


Fig 6.1 Single section of octagonal plant enclosure

Fig 6.2 BROADS IN THE BURE VALLEY



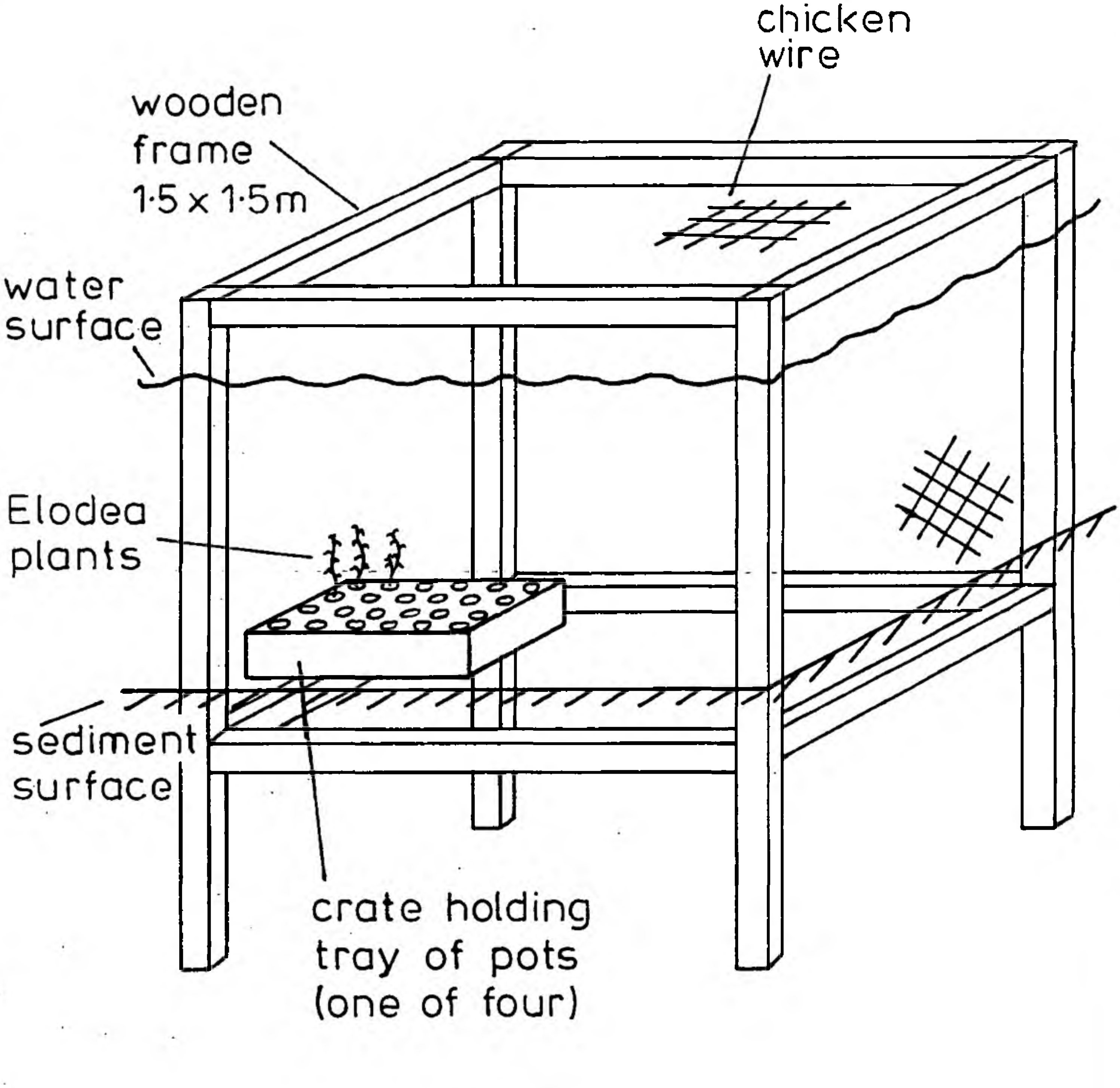


Fig 6.3 Simplified diagram of single species experimental setup

Monitoring

Plant growth was assessed by estimating percentage cover of each species separately at monthly intervals. Two people provided simultaneous independant estimates and a compromise agreed in the case of discrepancies.

6.2.2 Elodea experiment

Setup

In order to facilitate planting and retrieval for analysis, horticultural 'pot trays' were used. These are moulded from plastic with 24 3" pots in each semi-rigid tray. The trays were then placed in crates to provide rigidity. When testing the method in 1990, it was found the plants could not be harvested individually as they tangled together while growing and were too brittle to separate. The replicated units were therefore whole crates of twenty plants. For each treatment, twelve crates were planted to allow four per year to be harvested over three years.

Sediment from the broad was used as a planting medium. The four corner pots were filled with small stones so as to counteract the bouyancy of the plastic, leaving 20 pots free. Young *Elodea* plants were collected from a dyke the previous day. Healthy shoots with at least one root were chosen, the root end being pushed into the sediment with a finger. Crates were put in position in the broad soon after planting, before drying had occured. Rope handles were tied around the crates so they could be lowered and retrieved and plastic floats attached to the ropes to keep them at the water surface and mark their position. Four crates were placed in each $1.5m^2$ enclosure, or four placed close together in the open water.

Monitoring

The plants were harvested at the end of the growing season, in late August. The crates to be harvested were pulled up from the broad bed and all visible plant material removed. The sediment from the crate was then sieved and any roots or shoots missed added to the sample. In the laboratory ten 11cm apical sections of shoot were removed from each sample for determination of leaf number. The top 1cm was removed as the leaves were too closely spaced to be counted. The samples were divided into shoots and roots, dried at $90^{o}C$ and weighed. Total biomass, root to shoot ratio and leaf density were calculated.

6.3 <u>Results</u>

Both plant experiments have been affected by the breaching of the fish-proof barriers and subsequent loss of clear water environments. Belaugh Broad is now the only site providing substantially clear water. Mean summer chlorophyll values for the five sites are presented in Table 6.1.

Table 6.1 Mean summer (Apr-Sept) chlorophyll a concentrations in plant carousel experimental sites

Site	Chlorophyll a concentration (μgl-1) Mean	S.D.
Belaugh Broad	48.7	22.8
Hoveton Great Broad	109.9	71.7
Hoveton exclosure	102.4	68.9
Pound End	89.8	29.0
Ranworth	193.7	51.1

6.3.1 Carousel experiment

The results for the three most successful plant species are presented in Fig 6.4. White lilies generally did best, achieving 30-40% cover in all sites except Belaugh where the maximum was 18%. The maximum was generally achieved in September. Unprotected plants did noticably worse in the two Hoveton sites, especially the main broad where they were virtually eliminated, but there was very little difference due to bird grazing at the other sites. Yellow lilies survived poorly compared to white ones. There was virtually no growth in Belaugh unprotected, Hoveton Great Broad, Pound End or Ranworth Broad, very little in the exclosure (max 3%) and maximum 9% cover in Belaugh protected.

Growth of *Stratiotes* was apparently greatly affected by bird grazing. Plants quickly disappeared when not protected at all sites. Growth in the protected carousels varied between the sites. Belaugh Broad and the exclosure did best with maxima of 24% and 32% respectively. Hoveton Great Broad and Pound End achieved 10% and 11% but there was no survival in Ranworth Broad. Maxima were reached in June or July.

Submerged species achieved much lower results. *Ceratophyllum* reached a maximum of 5% in Hoveton Great Broad and Belaugh protected carousels. Other species achieved 2% or less. No submerged species were recorded in any of the non-protected carousels nor in Pound End or Ranworth protected carousels.

6.3.2 Elodea experiment

The results of the Elodea experiment for biomass, shoot to root ratio and leaf density are shown in Table 6.2. No plants were recovered from the unprotected crates inside the exclosure, very few from protected crates in the exclosure (mean 5.0 g dry wt m^{-2}, max 19.1 g dry wt m^{-2}) or unprotected crates in the main broad (mean 3.6 g dry wt m^{-2}, max 9.4 g dry wt m^{-2}). Only in the protected crates in the main broad had there been a good growth of plants (mean 239 g dry wt m^{-2}, max 348 g dry wt m^{-2}). Leaf density for this treatment was 3.84 +/- SD 0.33 leaves cm^{-1}. Shoot:root ratio for this treatment was 14 +/- SD 4.3, with a range of 10.3-18.2. The other treatments produced insufficient material for leaf density to be calculated.

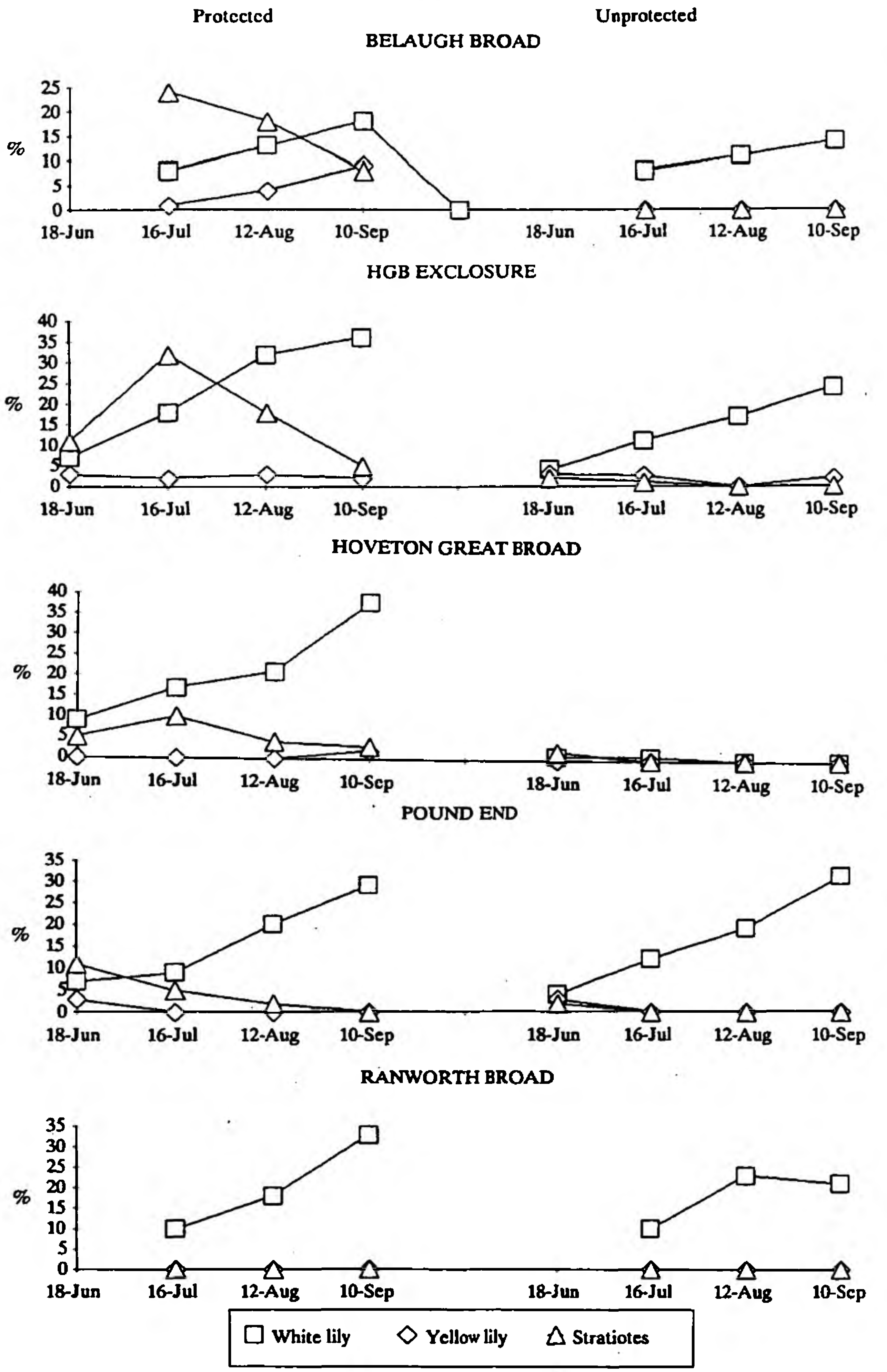


Fig 6.4 Percentage cover of white and yellow water lilies and *Sratiotes* in carousel enclosures 1991

INSIDE EXCLOSURE PROTECTED

No.	Biomass g dry wt m-2	Shoot:root ratio	Leaf nodes cm-2
1	0.67	11.30	-
2	0.00	-	-
3	0.22	9.80	-
4	19.10	57.00	4.00
MEAN	5.00	26.00	
S.D.	9.40	26.80	

INSIDE EXCLOSURE UNPROTECTED

No.	Biomass g dry wt m-2	Shoot:root ratio	Leaf nodes cm-2
1	0	-	-
2	0	-	-
3	0	-	-
4	0	-	-
MEAN			
S.D.			

MAIN BROAD PROTECTED

No.	Biomass g dry wt m-2	Shoot:root ratio	Leaf nodes cm-2
1	208.5	10.4	3.84
2	132.5	10.3	3.38
3	348.3	17.2	4.12
4	267.6	18.2	4.02
MEAN	239.2	14.0	3.84
S.D.	91.4	4.3	0.33

MAIN BROAD UNPROTECTED

No.	Biomass g dry wt m-2	Shoot:root ratio	Leaf nodes cm-2
1	3.65	10.50	-
2	1.21	14.80	-
3	0.00	-	-
4	9.44	8.20	2.87
MEAN	3.60	11.20	
S.D.	4.20	3.30	

Table 6.2 Results of biomass, shoot:root ratio and leaf density for the *Elodea* experiment in Hoveton Great Broad 1991

6.4 Conclusions

6.4.1 Carousel experiment

The ability to compare between sites of varying water quality has been curtailed this year because of the problems with the barriers. Nevertheless, information about the different species and different sites is available.

White water lilies were the most successful species, yellow lilies the least successful of the floating-leaved species. Both species are known to be among the most resistant of British aquatic plants to eutrophication. Yellow lilies still form two substantial stands in Hoveton Great Broad and a dense cover on adjoining Hudsons Bay and also Salhouse Little Broad. White lilies have been successfully reintroduced to Cockshoot and Belaugh Broads. In these situations they survive despite being unprotected from bird grazing.

The very different results in the experiment between these two species seem to indicate a different reaction to transplantion, which the white species survived very well and the yellow species very poorly. The fact that yellow lilies grow naturally in Hoveton Great Broad indicates that it is unlikely to be the physical conditions that are preventing growth, at least in this broad, although growth from transplanted rhizomes may be more severely affected by water or sediment quality. Because of the poor growth under protected conditions, nothing further can be said of this species resistance to bird grazing.

White lilies are evidently able to grow well under water quality conditions even as poor as Ranworth Broad. The relatively poor results in Belaugh Broad were probably due to a phytophagus beetle which had been able to build up a large population in carousels planted with water lily two years ago.

The amount of bird grazing seemed to vary from broad to broad, being low in Belaugh, Pound End and Ranworth, higher in the exclosure and very high in Hoveton Great Broad. This experiment is unreplicated so we cannot tell if these differences are representative, but it is interesting to speculate. The results could be due to the balance between the number of birds visiting a broad and the amount of plant material present to sustain them.

Pound End and Ranworth Broad have no plant growth and as such may not be visited by birds looking for food. Belaugh Broad is small and this may limit the numbers of visiting coot and geese, which are often territorial. The relatively large amount of plant material present may dilute the effects on any one stand. Hoveton is a large broad with long established but small plant stands and therefore can attract and accomodate plenty of birds while offering little sustenance. Grazing may be less in the exclosure because of the presence of the barrier and the proximity people on the nature trail. A breeding pair of greylags also established within the exclosure and may have driven other geese away. Such a combination of factors could produce the observed results. Certainly bird grazing should not be seen as a constant factor between broads, affected as it is by the complex behaviour patterns of these animals.

Growth of *Stratiotes* was variable when protected but very poor at all sites without protection. It therefore seeems to be more susceptible to grazing than water lilies, possibly due to greater palatability. The sites where it was most successful were Belaugh Broad and the exclosure. This does not fit in with the pattern of chlorophyll concentrations, but these two sites are the most sheltered. Ranworth Broad, where there was no survival, is the most exposed.

It is likely that the plants, which are unrooted and floating during the summer, were damaged by being knocked against the sides of the carousels by wave action, or that being pushed against the sides allowed them to be eaten by birds. Their normal habitat is in amongst dense plant beds where they are supported by the surounding vegetation. It is not proposed to conduct further trials with this plant.

Submerged species comprised *Elodea* and *Ceratophyllum* which were purposely planted and *Potamogeton* spp, and *Berula erecta* (water parsnip) which came in with the other species or occured naturally. Maximum cover achieved was 5% for *Ceratophyllum* and 6% altogether. The inocculum provided was certainly smaller than the floating-leaved plants, but the results are still disproportionately low. One problem is that in turbid water the plants cannot be seen until they reach near the water surface. If they never reach high enough they will be missed altogether. This is certainly a factor which could be affecting our results.

Belaugh Broad provided the only clear water environment. Submerged species did best here, but still only achieved 6% overall cover. Competition with the floating leaved species may have been a factor, these formed 33-35% cover, but as only half the carousels were planted up, this represents a considerable proportion of the planted area.

6.4.2 Elodea experiment

The results from this experiment are inconsistent with water quality in the different treatments. As chlorophyll concentrations were very similar between the exclosure and the main broad (see Fig 4.4 and Table 6.1), similar results would be expected between the two protected treatments and the unprotected crates. While this is the case for the unprotected plants, protected plants fared very differently in the two sites. The protective structures in the exclosure were still intact so this could not be due to bird grazing.

When harvesting the plants, growths of a bryozoan were observed on the ropes and crates recovered from inside the exclosure, but much less so from those in the main broad. It is thought that the exclosure plants had been smothered by the bryozooan growing epiphytically. Some years ago the area that the exclosure now occupies was used for an experiment involving large pens and bryozooan growths were observed on these structures. It seems that the area is heavily infested with their resistant spores. The results from the exclosure treatments must therefore be discounted. However, the results from the main broad show two interesting points.

Firstly, *Elodea* has been able to grow in the poor water quality of the broad. The fact that the plants were never visible from the surface atests to poor light penetration through the water. Although the experiment is being restarted in 1992, these crates are being left in position in order to test overwintering ability under these conditions. Secondly, bird grazing has apparently been able to virtually eliminate the unprotected plants. It is possible that the bryozooan could have affected these plants, but the evidence from the small amounts found growing on the crates argues against this.

In Cockshoot Broad in 1990, one enclosure containing four crates of Elodea was set up to test the method and equipment. One crate was harvested in each of the next two years and these results are available as an example of a clear water environment. In the first year only wet weight of plants was obtained. This corresponded to roughly 55 g dry wt m^{-2}, an order of magnitude less than the first years growth in Hoveton Great Broad. This indicates that the reduction in light penetration in Hoveton Great Broad is not affecting growth. Nutrient limitation may be the cause of reduced growth in Cockshoot.

In 1991 dry weight was measured at 0.47 g dry wt m^{-2} and shoot:root ratio at 9.4. Conditions in Cockshoot Broad have been similar over the two years so this result indicates that overwintering may be a problem for these newly established plants.

The first year of the *Elodea* experiment has produced an interesting set of results. The presence of the Bryozooans in the exclosure has prevented a comparison between areas of different water quality, which could not be counteracted by including the results from Cockshoot Broad because of the lack of replication. The presence of the Bryozooans is unfortunate, but very little can be done now that the exclosure has been built on this site.

Two measures are proposed to avoid the problem. Firstly, half the plants will be sampled in July instead of all in August as previously. Hopefully this will catch the plants before they have been completely smothered and so give a better idea of comparative growth and provide proof that this is indeed the problem. Secondly, replicated crates will be set up in Cockshhoot Broad as a standby clear-water environment. The relevance of this comparison is compromised by known and unknown differences between the two broads, in particular Cockshhoot has been mud-pumped and isolated whereas Hoveton Great Broad has not.

On the basis of this first years results it seems that bird grazing is more important than water quality for *Elodea*, under the conditions of light penetration and bird numbers present in Hoveton Great Broad.

7. STUDY OF THE INDIRECT EFFECTS OF BIOMANIPULATION IN HOVETON GREAT BROAD 1991

7.1 Introduction

The study of the benthic invertebrates in Hoveton Great Broad has continued this year. The main aim of this study is to investigate the indirect effects of biomanipulation, particularly upon the benthic invertebrates. This will look at their response to reduced predation and changes in the water quality. This may involve shifts in species composition and population dynamics, which in turn may influence and alter phosphorus recycling in the sediment. This was not achieved this year due to problems with excluding fish. Monitoring of the exclosure was suspended in September and hence the data in this report is mainly concerned with the results from the main Broad.

The benthic invertebrates found in Hoveton Great Broad are typical of those found in other Broads where there are no plants but fish are present. The main invertebrates found are chironomid larvae and oligochaete worms. The few species found are typical of the uniform habitat offered by the Broads.

7.2 Methods

7.2.1 Routine Monitoring

Ten samples from the exclosure and the main broad were taken fortnightly using a corer tube. This sampled an area of 63.6 cm2 and to a depth of 40 cm. This method was adopted in April 1991 after observing that chironomids were living deeper than we had originally realised. Previously we used an Ekman grab that only sampled to a depth of 15 cm.

Samples were sieved in the field using a 500um mesh net. Samples from the exclosure were sieved in the main broad. Samples were sorted live, as soon as possible after collection, using the subsampling method described in Moss et al (1991). Sorted animals were placed in 70% alcohol in a tray, the chironomids and oligochaetes being counted and kept separately. A further subsampling method of the oligochaetes was employed to speed up the process of identification.

Oligochaete subsampling method.

It was decided that a minimum total number of 100 oligochaetes should be identified from each of the areas. This meant that when ten samples had been taken then approximately 10 oligochaetes from each sample needed to be removed for identification. This was done by taking three or four subsamples and placing them in a white tray. All the oligochaetes in this tray were removed and set aside for identification. If the number of oligochaetes was greater than 10 then the remainder of the oligochaetes in the rest of the subsamples were counted and then discarded. However, if the number of oligochaetes was less than 10 then a further suitable number of subsamples were taken to increase the number of oligochaetes identified to 10 or more.

No subsampling method was employed for the chironomids as fewer individuals are recovered and it is important to look at each animal.

Oligochaetes were identified immediately using Brinkhurst (1971). The worms were removed in to water and then placed on a slide and observed at a high magnification. After identification the worms were discarded.

Chironomids were removed from the alcohol into water prior to being placed in concentrated lactic acid where they were left for the minimum of a week. This is to clear the tissues making identification easier. After a week the chironomids were removed from the lactic acid back into water.

Chironomids were identified to genus by looking at features on their heads using Wiederholm (1983). Species can only be determined by looking at the male adults. Attempts were made to hatch out flies with reasonable success although mostly females were collected.

The heads were removed in a drop of water on a slide. Before applying the coverslip the head capsule width of each individual was measured using a calibrated graticule in the eyepiece of the stereo microscope. This allows the instar number to be determined as the head capsule size is discrete and unique for each instar. Both the head and body of each individual were mounted under a round coverslip, gentle pressure being applied to squash the head. After examination at high power the slides were discarded.

7.2.2 Organic Content Survey.

In March a survey of the water, organic and calcium carbonate content of the sediment was done in the exclosure and the main broad. This was to make sure that there was no difference in the habitats offered by the two areas.

Ten samples were taken from each area. Samples were taken using a corer from which the sediment could be extruded. The top 15 cm of each core was taken, the rest was discarded. In the laboratory each sample was mixed thoroughly and a subsample of approximately 1 g removed into a crucible and weighed. These were then left to dry overnight at 100^{o}C thus allowing water content to be determined. To obtain the organic content the sediment was reweighed and placed in a muffle furnace at 550^{o}C for 1 hour. Once cooled the sediment was then reweighed and placed in the muffle furnace at 950^{o}C for 1 hour and the calcium content determined.

7.2.3 Depth Profiles of Animals through the Sediment

Three profiles were done in 1991, one in March, one in June and one in October. These were done to obtain more information on the animal's distribution through the sediment after observing animals living below 15 cm. After the first survey it was decided to continue them at intervals through the year to see if any differences developed between the exclosure and the broad and to find out if the distribution through the sediment altered during the year.

Eight samples from each area were taken. Samples were taken using a corer from which the sediment could be extruded at fixed intervals. The sediment of each sample was cut into 5 cm sections down to a depth of 40 cm. In each area the corresponding sections from each of the eight samples were bulked in the same pot. Samples were sorted live using the subsampling method.

7.3 Results

7.3.1 Routine Monitoring

Animal Numbers

The graphs include the result from the exclosure until sampling was discontinued at the end of September. Oligochaete numbers, figure 7.1a, after rising substantially during February were generally higher by a factor of eight in the exclosure. They remained at this elevated level until the beginning of August when numbers, having risen to a peak in mid July, fell considerably to reach similar numbers to those seen in the broad. From this time until the end of September the numbers in both areas remained very similar. In the main Broad the oligochaete numbers increased from the middle of May until the beginning of August when there was a decrease in numbers. Numbers fell reaching similar values as those seen at the beginning of the year.

On the other hand chironomid numbers, figure 7.1b, in both areas have remained very close and have followed similar fluctuations over the year. At the beginning of the year there was a difference between the two areas but this was lost by the end of March. After that time the numbers remained very close and followed similar fluctuations although the numbers in the main broad remained slightly higher from mid April until the end of August. On the two final sampling occasions of the exclosure, numbers were higher in the exclosure than the broad. The numbers increased after a large decrease in the number of oligochaetes in the exclosure.

Seasonal fluctuations in the numbers of both the exclosure and the broad were very similar apart from a difference at the beginning of the year. Chironomid numbers began to increase at the beginning of April and a peak was reached at the beginning of May. Numbers then fell until the beginning of July in the Broad and the middle of July in the exclosure after which a small increase was seen. Numbers then fell again until the increase seen in the middle of September and the end of August respectively. Numbers in the broad were considerably higher at the end of the year than those seen at the beginning of the year.

Fluctuations in Chironomidae Numbers.

Seven genera of chironomids have been found to occur regularly. Several other genera have been recorded occasionally. All are listed in table 7.1.

Figure 7.2 shows the variation in numbers of the seven commonly found genera. It can be seen that during the summer the population was mostly composed of either *Einfeldia*, *Microchironomus* or *Chironomus*. These feed on algae from the water and the sediment. *Tanytarsus* and *Polypedilum* are also of this type but they were not found in large numbers during the year.

The other two chironomids, *Tanypus* and *Procladius* are mixed feeders, feeding upon other chironomids and oligochaetes as well as detritus and algae. These do not reach very high numbers during the year.

Due to the mesh size of the sieving net the smaller instars were not retained accurately and this is reflected when looking at the instar data. Difficulty is also encountered in determining whether decreased animal numbers are due to hatching or predation by

Figure 7.1a

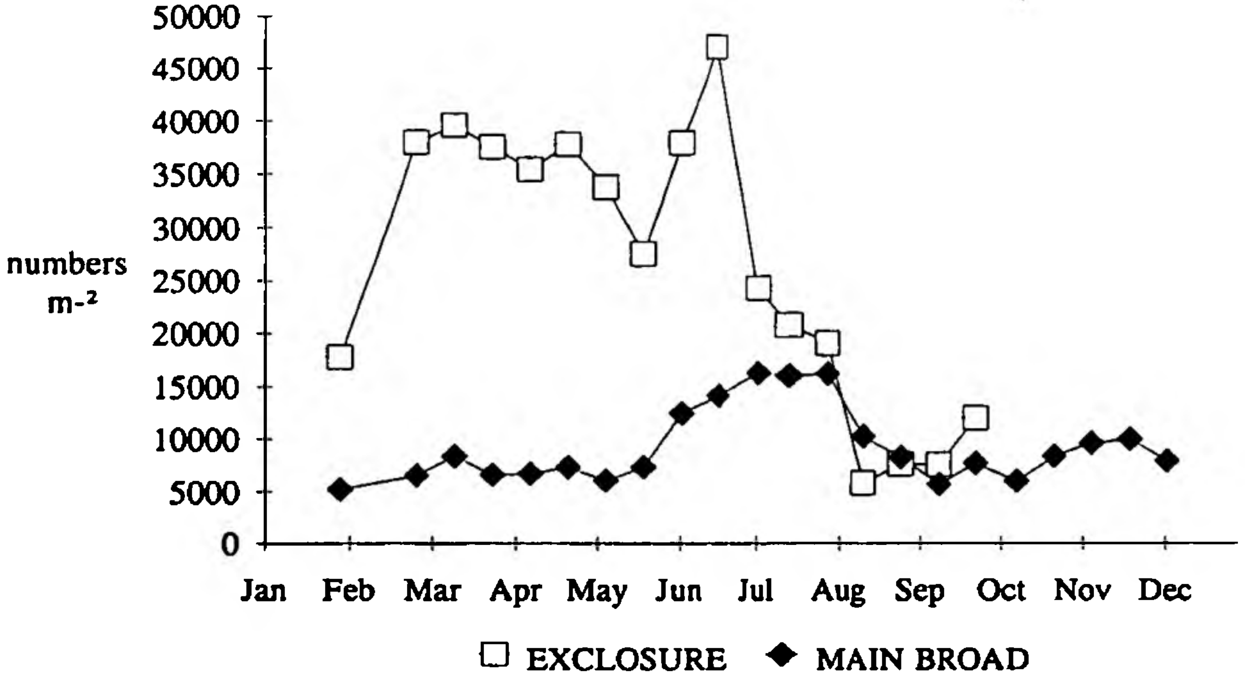


Figure 7.1 b

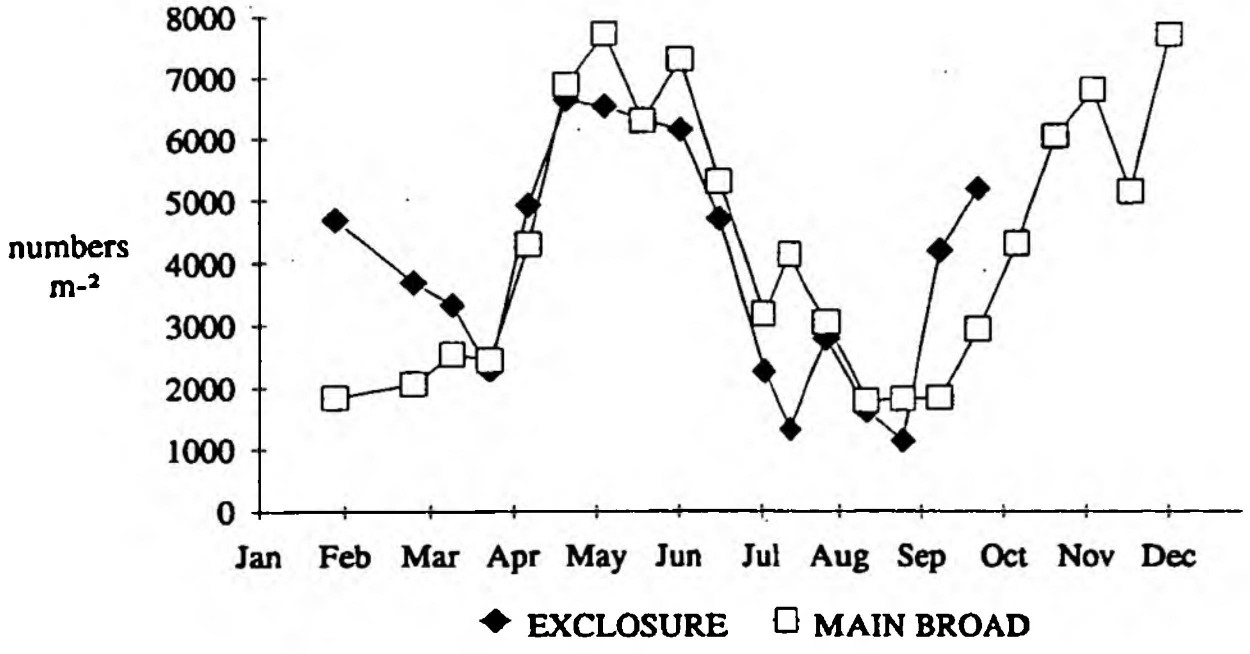


Figure 7.1 **Animal numbers m^{-2} in Hoveton Great Broad 1991 a) oligochaetes and b) total chironomids.**

Table 7.1 Chironomid genera found in Hoveton Great Broad

	Routinely	Occasional	umber of Occasions
Family Chironomidae	*Chironomus*	*Fleuria*	6
	Einfeldia	*Glyptotendipes*	5
	Microchironomus	*Corynocera*	1
	Polypedilum		
	Tanytarsus		
Family Tanypodinae	*Procladius*	*Psectrotanypus*	1
	Tanypus		
Family Orthocladinae		unidentified	1

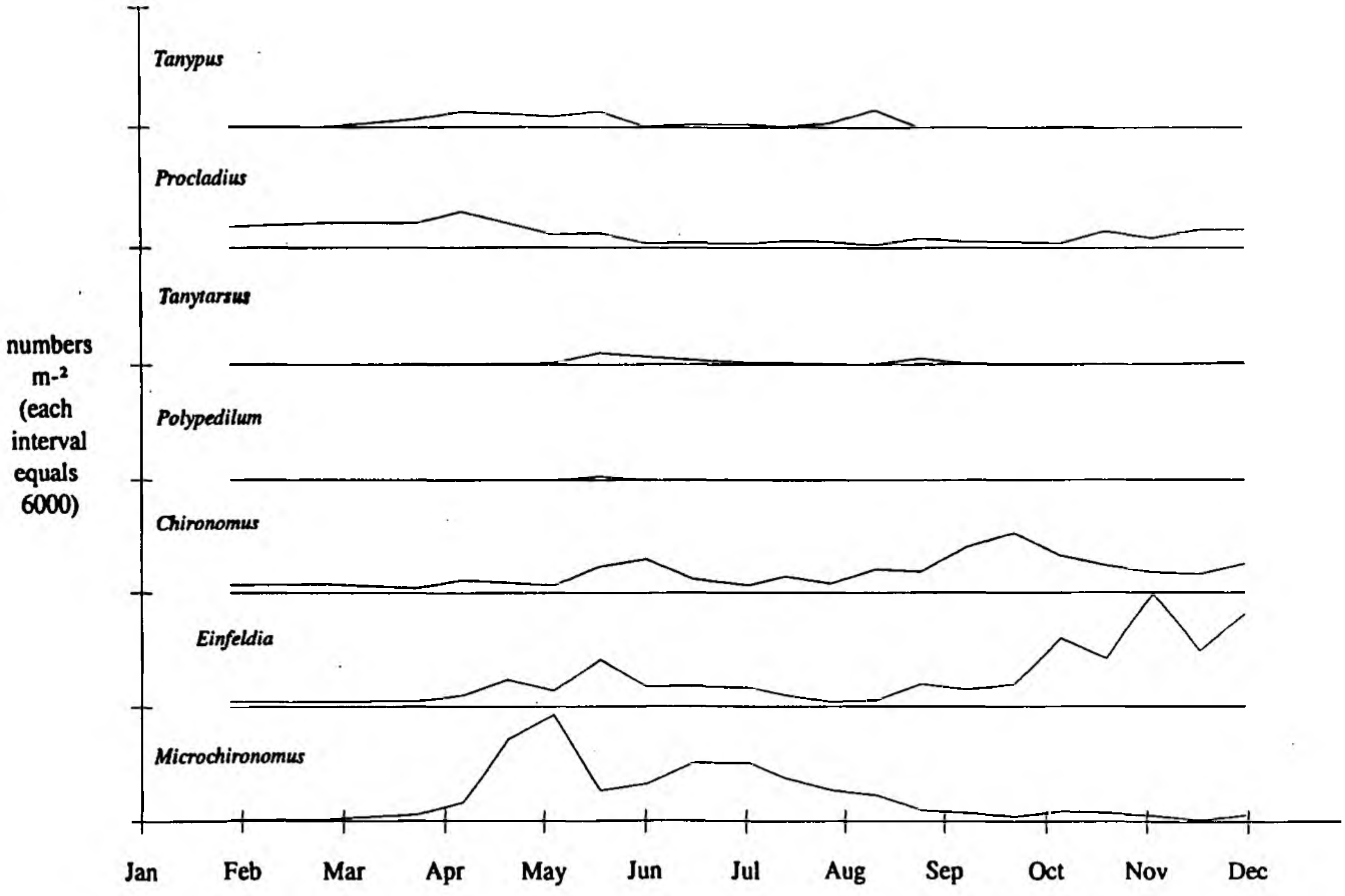


Figure 7.2 **Variation in Chironomidae numbers in Hoveton Great Broad 1991**

fish. Substantial drops in fourth instar larvae numbers have generally been attributed to hatching rather than predation.

Figure 7.3 shows in detail the variation in numbers of the different instars of *Chironomus*, which were obtained by examination of the head capsule size. *Chironomus* is the largest chironomid found in the broads.

Over the winter period the population was mainly composed of fourth instar larvae with some third instar larvae. In early spring numbers of both instars increased, this was probably due to the development of first and second instar larvae. At the end of May fourth instar larvae numbers decreased, this continued into early June. This substantial decrease in numbers was probably due to emergence of the adult flies, this is expected in late May and early June as was seen by Hilsenhoff (1966). Most of the animals that had overwintered probably hatched so animals found subsequently are probably derived from eggs laid at the beginning of the hatch. There appeared to be no further hatch and the animals overwintering consisted of third and fourth instar larvae. During September there was a large increase in the numbers of third instar larvae followed by a dramatic decrease at the end of September. As there was no increase in the fourth instar larvae this large decrease may have been due to predation.

Figure 7.4 shows the variation in numbers of *Einfeldia. Einfeldia* is a medium sized chironomid. It would appear that three hatches occurred during the year, in June, July and October. After the July hatch the numbers of all instars were very low, the majority of the animals that overwintered having hatched. In the middle of August an increase in second instar larvae was seen. These animals probably originated from eggs laid during the first and second hatch. After a slight increase in fourth instar numbers these decreased to almost zero in October, indicating another hatch. The overwintering population consisted of second and third instar larvae at greatly increased numbers compared to the previous year.

Figure 7.5 displays the data showing the variation in numbers of *Microchironomus*. This is a small larva reaching body lengths of only 8mm. Due to their small size third and fourth larvae only were found. There appear to have been two hatches over the summer one occurring for a short period in early summer and the other from June to the end of August. Predation may also have contributed to the sustained fall in fourth instar numbers seen over the summer.

The overwintering population consisted of low numbers of third instar larvae only. Numbers remained low until the end of March when there was a large increase in the numbers of fourth instar larvae and a similar, but not so large, increase in third instar larvae. By the middle of May third instar numbers had fallen to almost nothing, possibly due to predation, whilst there had been a corresponding fall in the number of fourth instar larvae, probably as a result of hatching.

Numbers of fourth instar larvae rose again by mid June and there was also a small increase in the number of third instar larvae. After this time, from June to the end of August, the fourth instar larvae gradually decreased in numbers until there were none left suggesting almost continuous hatching over this period although predation cannot be ruled out. Third instar larvae again constitute the majority of the population at the start of the winter.

Figure 7.6 shows the variation in the instar numbers of *Procladius*. This comes from the family Tanypodinae. It is a mixed feeder and consumes other smaller chironomids and oligochaetes as well as algae and detritus. It is a medium sized larvae.

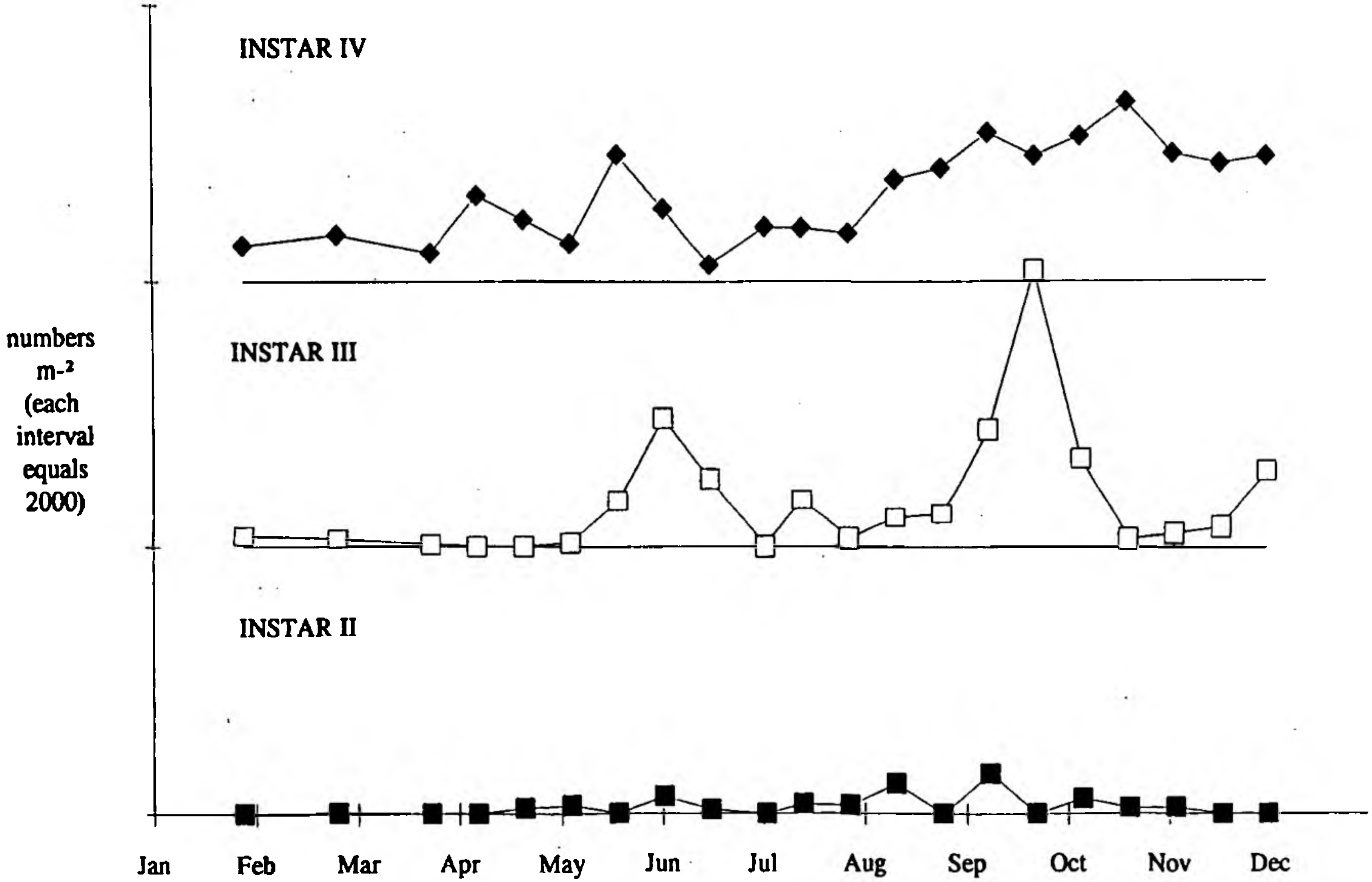


Figure 7.3 **Variation in numbers of *Chironomus* instars. Hoveton Great Broad 1991**

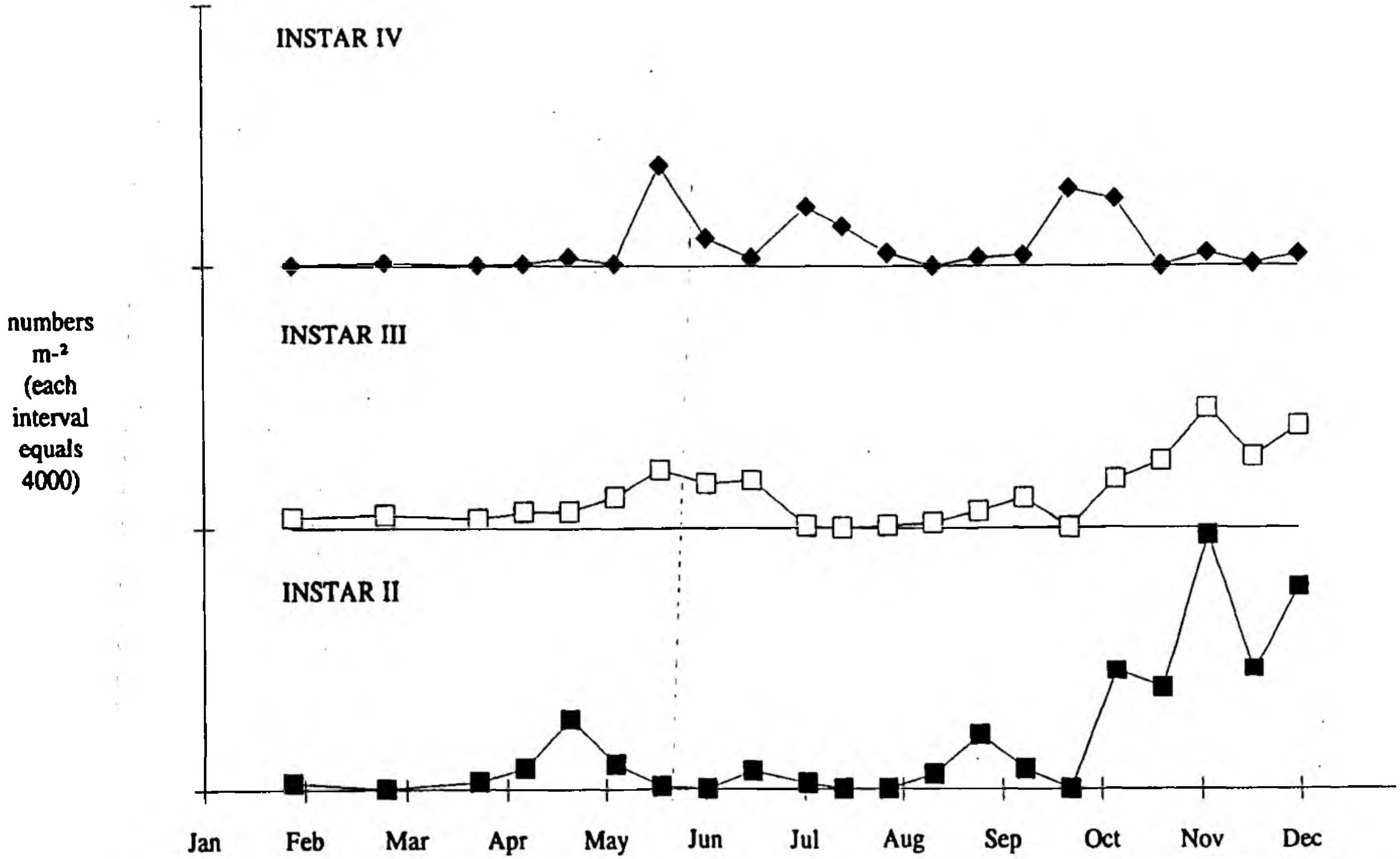


Figure 7.4 Variation in numbers of *Einfeldia* instars. Hoveton Great Broad 1991

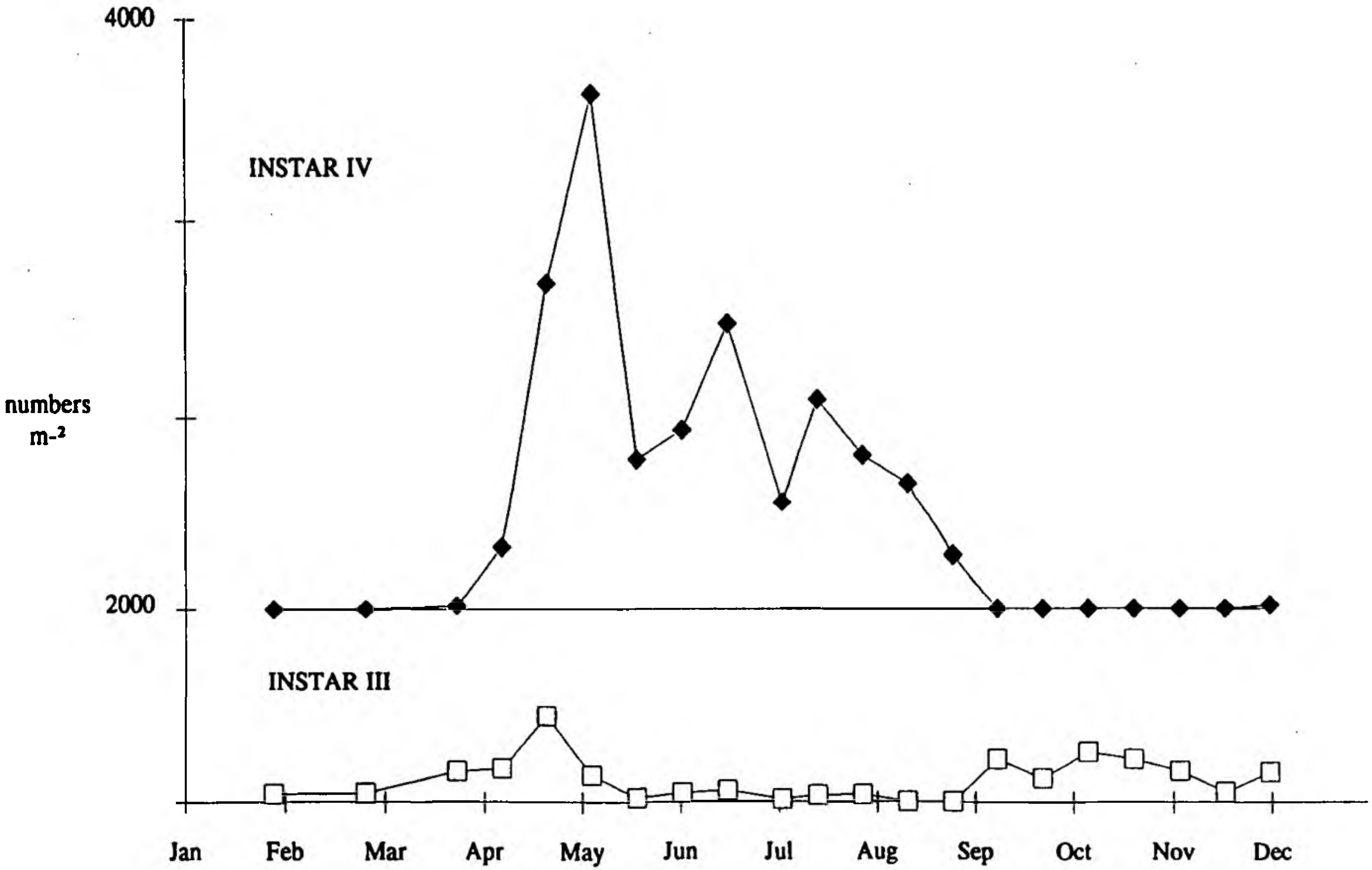


Figure 7.5 **Variation in numbers of *Microchironomus* instars. Hoveton Great Broad 1991**

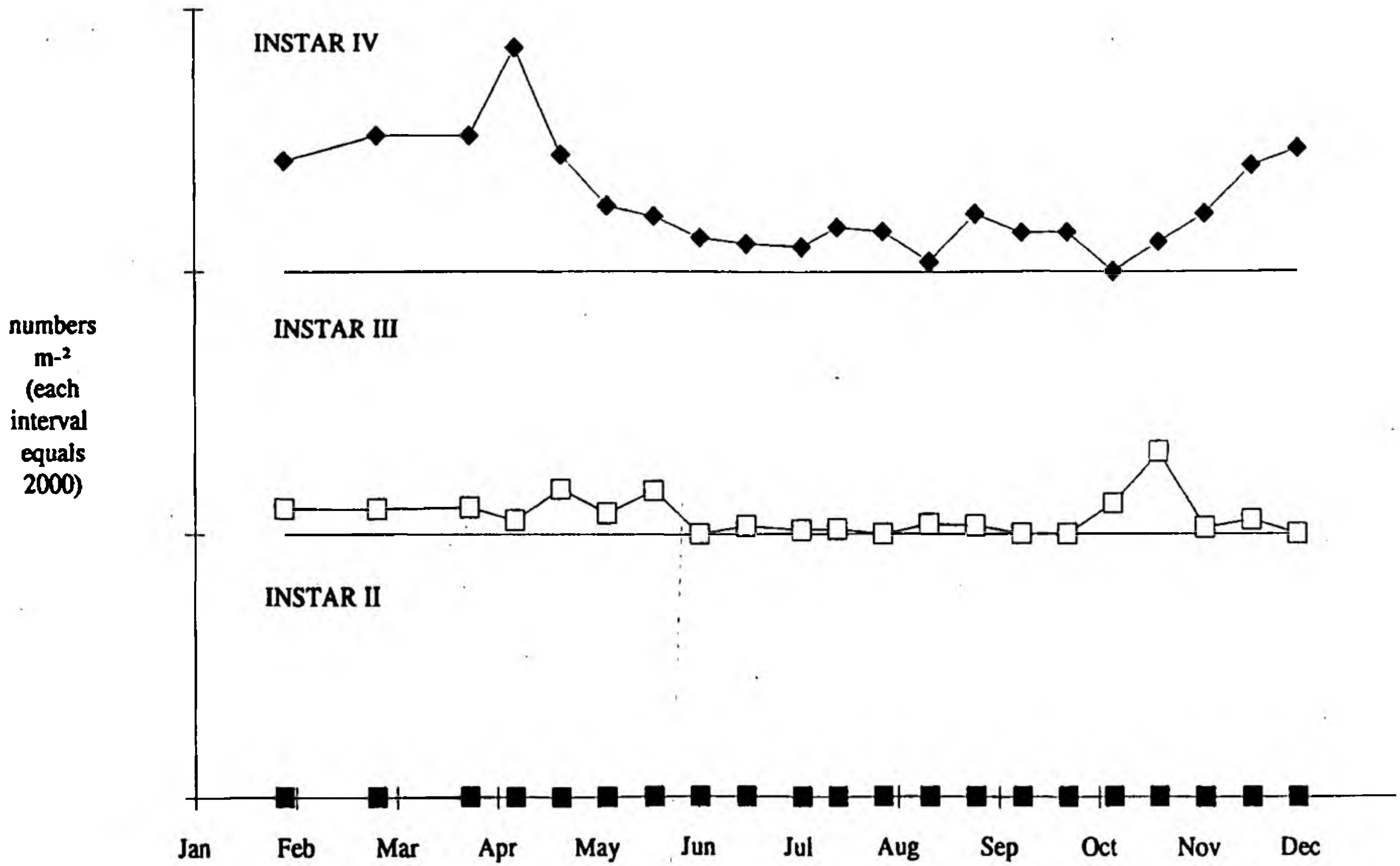


Figure 7.6 **Variation in numbers of *Procladius* instars. Hoveton Great Broad 1991**

Procladius produces an early generation of flies from animals that have overwintered. There may also be another hatch in the early Autumn with the remainder of the overwintered larvae hatching plus any animals that may have developed from eggs laid after the first hatch. At the end of the year it appeared that fourth instar larvae, produced from eggs during the summer, overwintered.

The other three genera found regularly, *Tanytarsus*, *Polypedilum* and *Tanypus* generally did not occur in sufficient numbers to make it possible to look at their lifecycles in a detailed manner.

7.3.2 Depth Profile of animals through the sediment.

The results of the animal profiles are shown in figures 7.7a,b,c and 7.8a,b,c. Figures 7.7a,b,c show the profiles of the oligochaetes on the three different dates that the survey was done.

The oligochaetes showed a similar distribution on all occasions. The majority of the oligochaetes were found in the depth range of 5 - 15 cm below the surface. As the year progressed the animals moved up the sediment column. In March animals could be found as deep as 30 cm but by June the animals were only found as deep as 25 cm. The first two surveys also show the data from the exclosure. Both areas follow similar distributions although there were fewer animals in the broad as would be expected.

Figures 7.8a,b and c show the results of the chironomid profiles. On all three dates the distribution of the animals through the sediment is again very similar. The majority of the chironomids occurred in the top five centimetres of sediment. However, these were mainly small chironomids, it is unusual to find a fourth instar *Chironomus* close to the surface. It can also be seen that as the year progresses so the animals were not found so deep in the sediment. In March animals were found down to a depth of 30 - 35 cm and by June only a few were found as deep as 20 - 25 cm.

Figures 7.9a,b and c show the depth profiles of the fourth instar *Chironomus* only. Again the distributions are all very similar and it can be seen that the animals move up the sediment column as the year progresses. These results also confirm that the smaller chironomids live in the surface layers whilst the larger fourth instar larvae mainly live at a depth of 15 - 20 cm early in the year and mainly at 15 cm later in the year.

In summary, all the animals show similar distributions throughout the year. However, the animals do tend to move up the sediment column during the summer months. Small chironomids live in the surface layers whilst the larger ones mainly live at a depth of 15 cm.

7.3.3 Sediment Content Survey.

The results of the sediment content survey are shown in figure 7.10. These results show that all the parameters measured in the two areas are the same and that there is no difference in the habitat offered by the two areas.

7.4 Discussion

The number of different chironomid genera found in the broad was similar to that found in studies done in Broadland and other temperate shallow lakes,see table 7.2. All

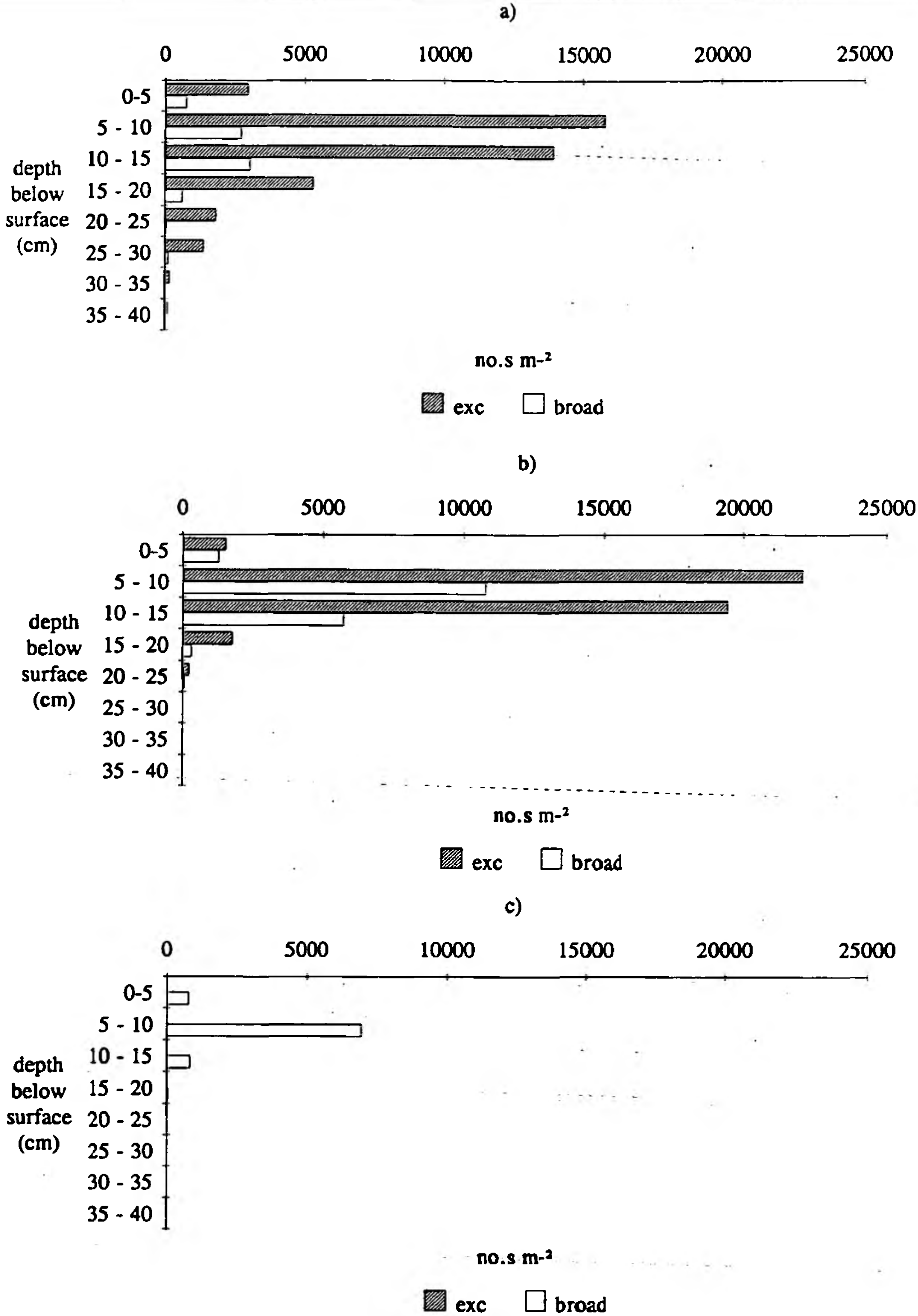


Figure 7.7 **Depth profiles of oligochaetes in Hoveton Great Broad 1991 a) 20/3/91 b) 12/6/91 and c) 9/10/91**

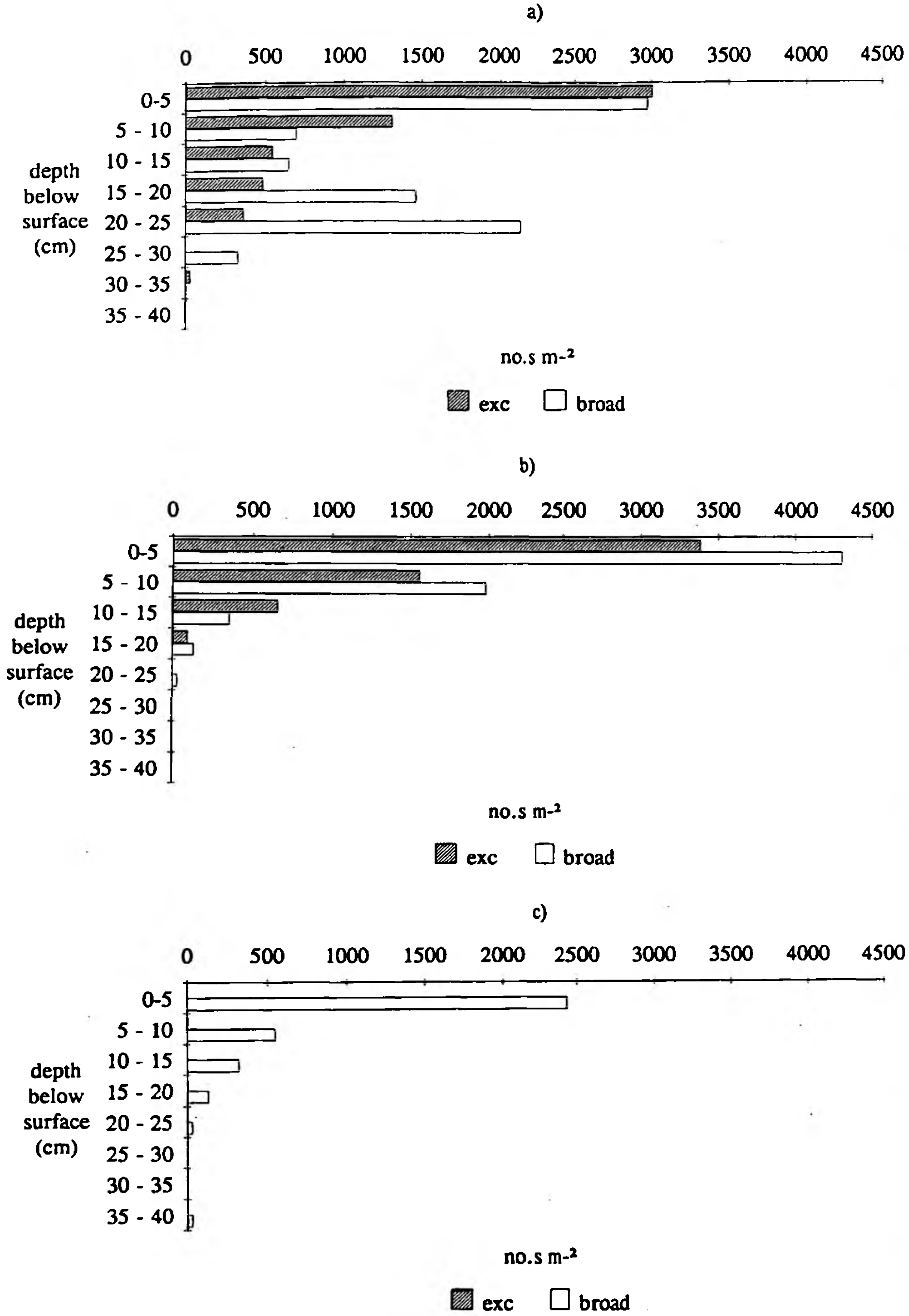


Figure 7.8 **Depth profiles of chironomids in Hoveton Great Broad 1991 a)20/3/91 b) 12/6/91 and c) 9/10/91**

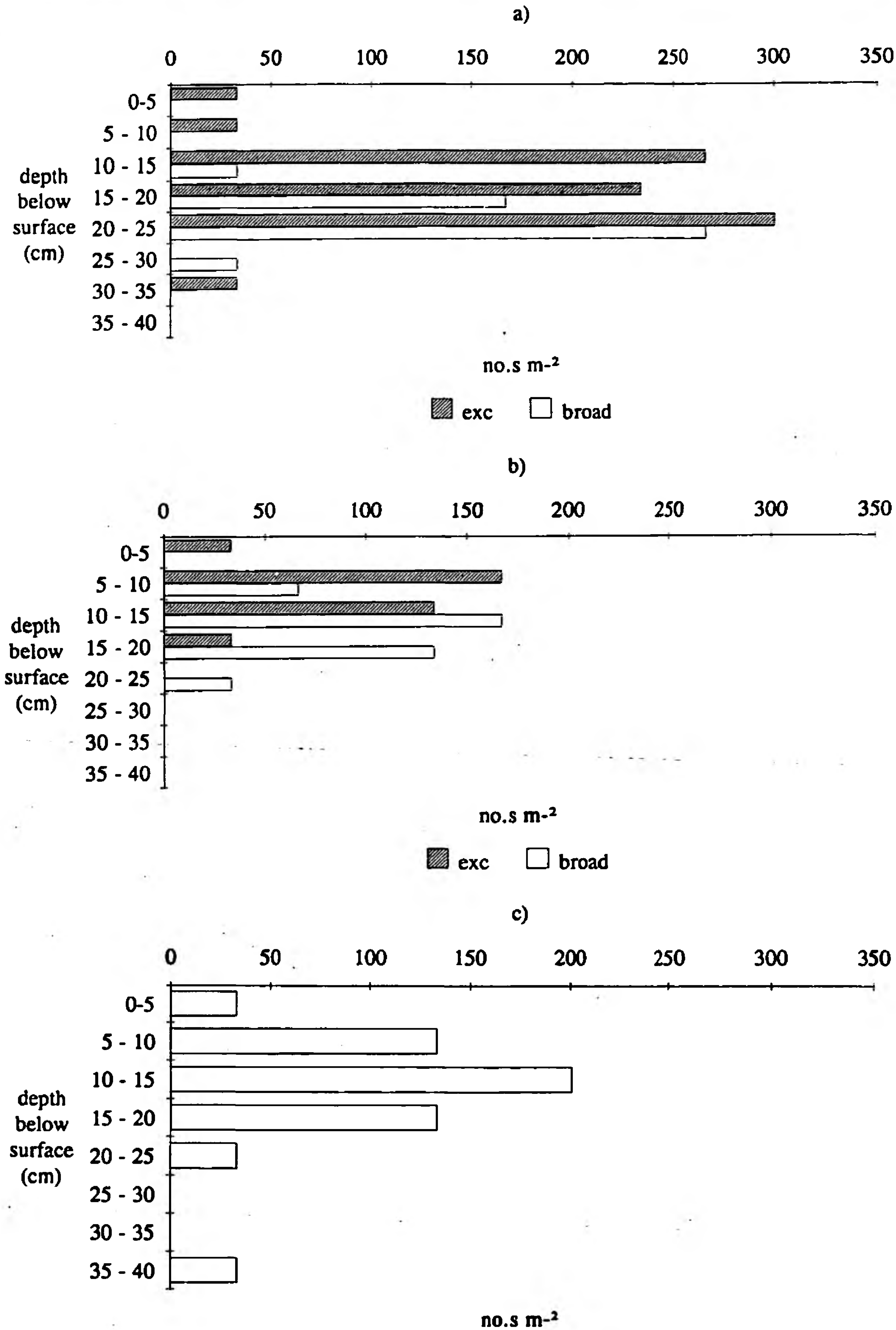


Figure 7.9 **Depth profiles of fourth instar *Chironomus* in Hoveton Great Broad 1991 a)20/3/91 b) 12/6/91 and c) 9/10/91**

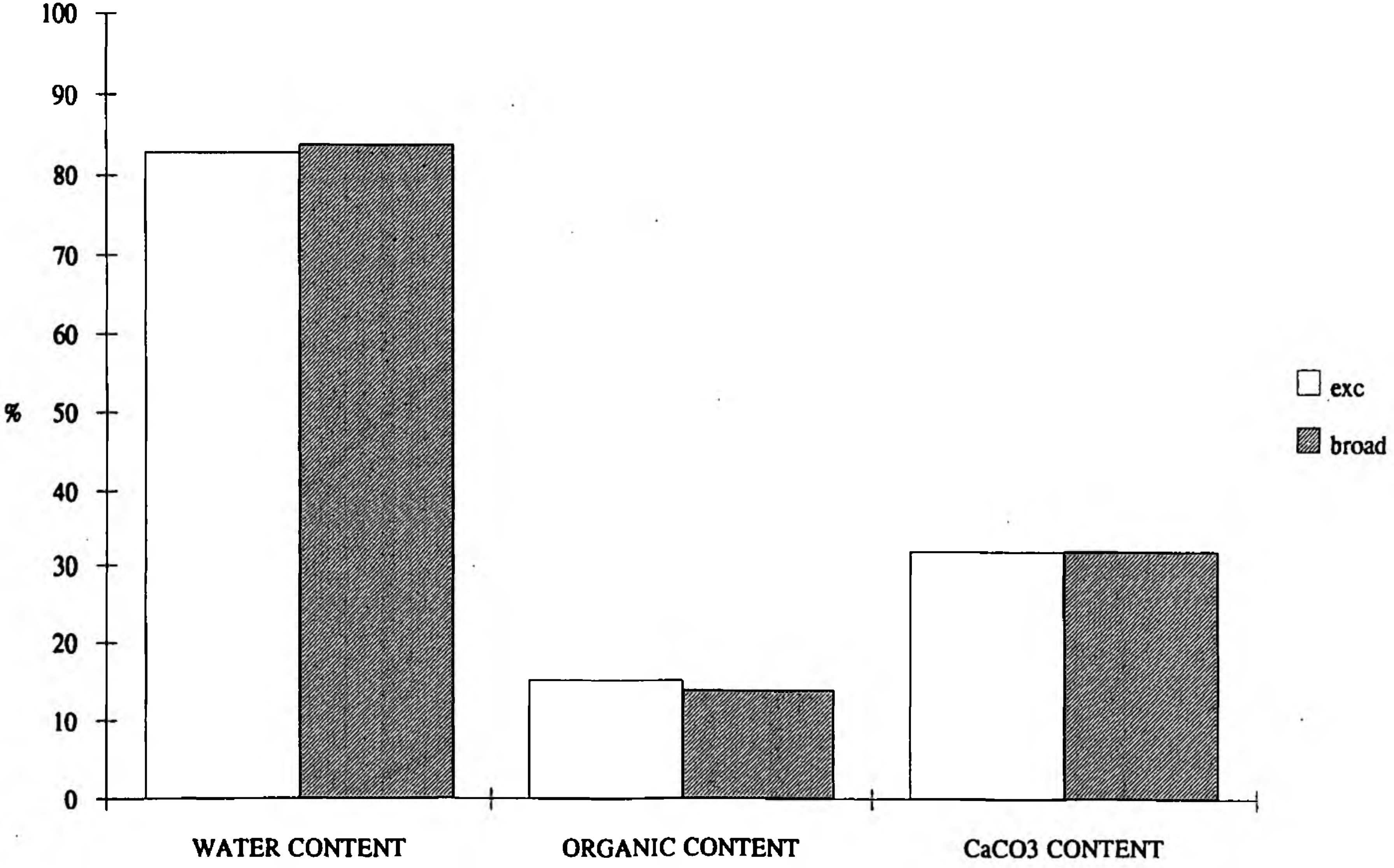


Figure 7.10 **Mean water, organic and calcium carbonate content of Hoveton Great Broad sediment, March 1991**

Table 7. 2 A Comparison of Chironomid numbers found in shallow lakes.

Year	Lake	Species				Author
		numbers m-2				
		Chironomus plumosus				
		max	min			
1973	Federsee	1750	410			
1974		1320	390			
1975		1285	70			Frank (1982)
1976			500			
		4th instar *Chironomus plumosus*		4th instar *Glyptotendipes sp.*		
		max	min	max	min	
1970	Tjeukemeer	150	50	1050	100	Beattie (1982)
1971		450	50	1100	25	
1972		450	100	1050	0	
		Chironomus plumosus				
		max	min			
1966	Winnebago	3120	1940			Hilsenhoff (1966
Norfolk Broads						
		4th instar *Chironomus*		Total chironomids		
		max	min	max	min	
1989	HGB	/	/	558	/	Moss (1989)
1991	HGB main	1315	127	7726	1787	This study
		Chironomus		*Tanytarsus*		
		max	min	max	min	
1973	Upton	590	/	130 000	< 100	Mason (1977)
1974	Alderfen	704	129	/	/	Mason (1977)

the different chironomid populations in Hoveton undergo one or more hatches during the summer months, one in the early summer where animals that have overwintered emerge and one in the late summer consisting of animals that have developed from eggs laid after the first hatch. Not all the chironomids developing from these eggs hatch, some remain as larvae which then overwinter. Eggs do not remain viable over the winter period, eggs of *Chironomus plumosus* were reported by Hilsenhoff (1966) to deteriorate below 8°C. Two to three hatches per year are typically expected in temperate lakes such as the Broads as the lifecycle can be completed quickly in the summer months. Completion and duration of the lifecycle is determined by several factors. These factors are temperature, food availability and oxygen concentrations. Temperature is probably the most important, the higher the temperature the faster development. In the Arctic completion of the lifecycle, from egg to emergence has been reported as taking from two to seven years to complete (Pinder 1986).

When it is cold larvae stop feeding. Before hatching chironomids have to reach a certain weight in the form of fat deposits (Jonasson 1972) so when it is cold or there is a limited food supply development time will increase. In the Broads food supply should not be a limiting factor. Also oxygen concentrations should not be a limiting factor in Broadland as the water column remains well aerated during the summer months. Under low oxygen conditions, as is found in deeper lakes, Jonasson (1972) found that larvae stopped feeding and hence development time was extended. The typical development time of *Chironomus* is quoted by Hilsenhoff (1966) and Frank (1982) as being, after a spring hatch, ten to eleven weeks. This is the time from the eggs hatching to the adult fly emerging.

The chironomid population in Hoveton Great Broad is dominated by *Microchironomus*, *Chironomus* and *Einfeldia*. *Microchironomus* occurs from late spring until September. The peak in population numbers occurs early in the year and may be related to the algal content of the water column. This is followed by a peak in *Chironomus* numbers which is then closely followed by a peak in *Einfeldia* numbers. This succession may be associated with food supply and algal species present. Once all the fish have been removed from the exclosure and the water remains clear this may affect the species present and the succession order due to an altered food supply. In oligotrophic lakes *Tanytarsus* rather than *Chironomus* are reported by Frank (1982) as being the dominant species.

Lack of fish may also play a role in determining chironomid species present as a result of altered sediment structure due to no fish disturbance. When fish are present at a certain optimum number then the predation effect on invertebrate numbers is low and fish feeding activity oxidises the sediment. This imparts a better structure promoting chironomid production. When fish numbers are greater than this optimum level then chironomid numbers may begin to decrease due to predation effects and when fish numbers are very low then chironomid numbers may fall due to poor sediment aeration and structure. In this final case the system may become oligochaete dominated.

Procladius and *Tanypus* numbers do not reach such high levels as the three dominant chironomids. These animals may be limited by food. *Procladius* maintains a fairly substantial overwintering population whilst *Tanypus* numbers are very low. *Procladius* numbers reach a peak in the spring and then remain low for the remainder of the summer. On the other hand *Tanypus* has two peaks, one in the late spring and one in August. The numbers reached in these peaks are lower than the *Procladius* peak.

It has been impossible to say whether falls in fourth instar larvae numbers were due to hatching or predation by fish. However, from the results of the depth survey we found that the majority of the fourth instar *Chironomus* larvae live at depths of 15 cm and below, figures 7.9a,b & c, in the sediment suggesting that fourth instar *Chironomus* may be less vulnerable to predation than the smaller *Chironomus* instars and other types of fourth instar larvae. In the case of *Chironomus* a decrease in fourth instar larvae numbers is more likely to be attributable to hatching rather than predation. It is still not possible to separate the effects of predation and hatching upon all the other species that live nearer the surface and may be more vulnerable to predation. These animals live nearer the surface as they are probably not physically able to burrow into the sediment as well as not being able to survive the conditions found deeper in the sediment. To determine when hatching is taking place in future emergence traps will be placed in the broad and exclosure.

We were unable to study the primary objective of this project, which was the response of the benthic invertebrates to a fish free environment, as a result of complications with the barrier wall. It now seems likely that there was a substantial fish population in the exclosure for the duration of the summer. However, the study of the benthic invertebrates of the main broad has provided valuable background data that can be used as a comparison in the coming year once a fish free environment is achieved in the exclosure.

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8. PHOSPHORUS RELEASE IN HOVETON GREAT BROAD AND THE EXCLOSURE

8.1 Introduction

Phosphorus release from sediments in Broadland is a regular and well noted occurence. The eutrophic status of the Broads means that phosphorus release is more likely to occur due to the continual input and sedimentation of phosphorus.

Remobilisation is generally dependant on a number of factors and in Broadland the main ones appear to be as follows. Soluble phosphorus and iron levels in the pore water; sulphide concentrations in the pore water; loss of the oxidised surface layer and chironomid activity (bioturbation).

Most release occurs in the summer months once temperatures and biological activity in the sediment begin to rise. During the summer the sediments become very anoxic whilst the shallow overlying water remains well aerated. Usually a thin surface zone of oxidised sediment is maintained. In the anoxic sediment any iron present exists in the reduced ferrous state and this does not bind with phosphorus. Under oxidised condtions, such as those existing at the surface, the iron is oxidised to ferric iron which will then precipitate out the phosphorus as a ferric phosphate complex. However, the oxidised surface layer may be lost periodically due to excessive microbial activity. At this time, depending on the iron and phosphorus concentration of the pore water, release may take place.

There is also evidence that phosphorus release may be related to the numbers of chironomids present in certain instances. Species that construct tubes in the sediment are the ones most likely to exert an effect. The animals pump water through these tubes, probably for aeration. When these tubes are near the surface then phosphorus rich water may be passed out directly into the overlying water column. Deeper in the sediment they may facilitate the movement of phosphorus up through the sediment profile.

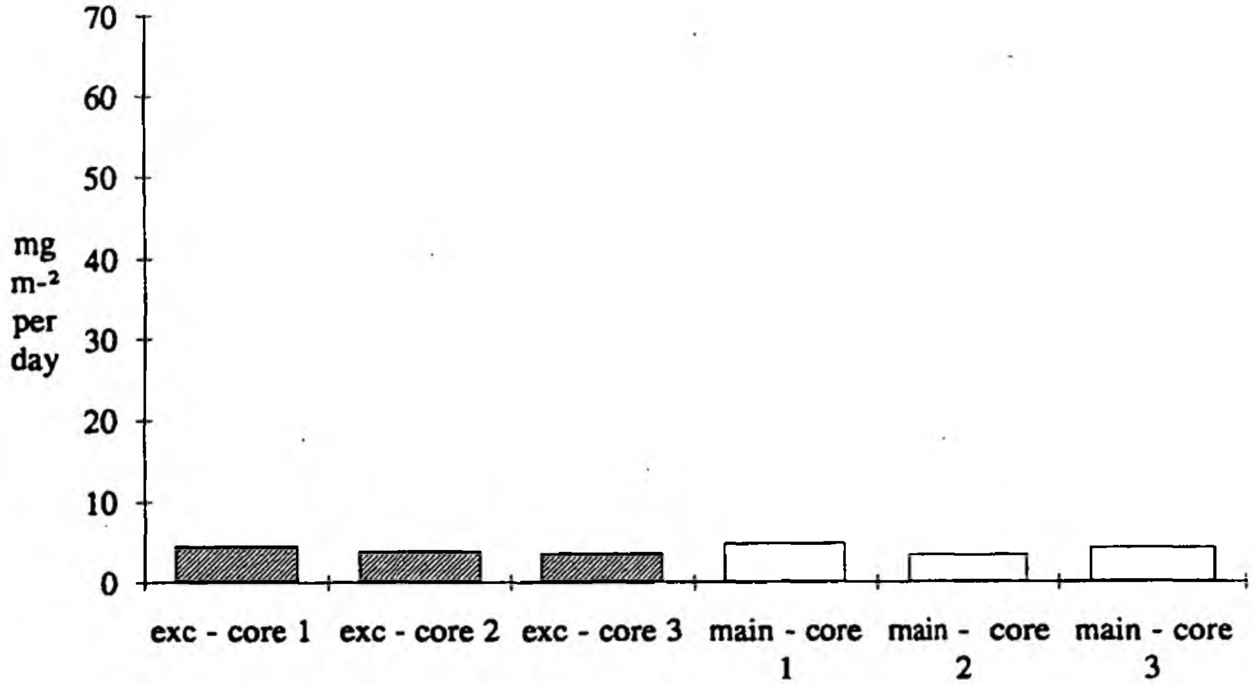
This situation is further complicated by the presence of sulphide which forms a ferrous sulphide complex with interstitial iron which is then no longer available to react with phosphorus.

In Hoveton sets of cores were taken from both the exclosure and the broad at intervals over the summer to look at phosphorus release rates and pore water concentrations. These were done to look at any differences in phosphorus recycling that may occur once all the fish had been removed. However due to problems with fish exclusion this year we were not able to consider this particular aspect.

8.2 Work Done in 1991

Sediment cores were taken from the exclosure and the broad every four weeks from May to August using a perspex tube and corer as described by Jackson and Phillips(1990). On each sampling occasion a total of five cores from each area were taken. Of the five cores, three were used in a release experiment, one was used to measure the redox and pH and the final one was used to measure the interstitial levels of soluble reactive phosphorus, ferrous iron and soluble sulphide.

a)



b)

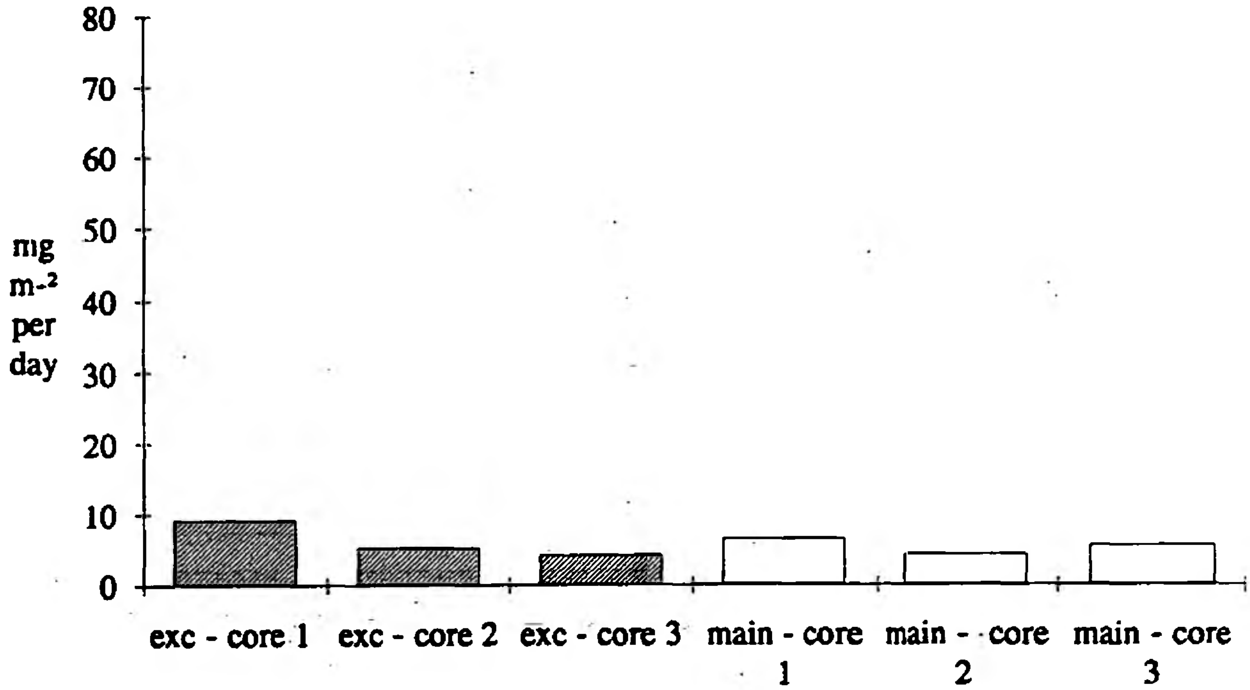


Figure 8.1 a)Average and b) maximum experimentally determined phosphorus release rates (mg/m^2/day) from cores collected at Hoveton Great Broad on 7 May 1991.

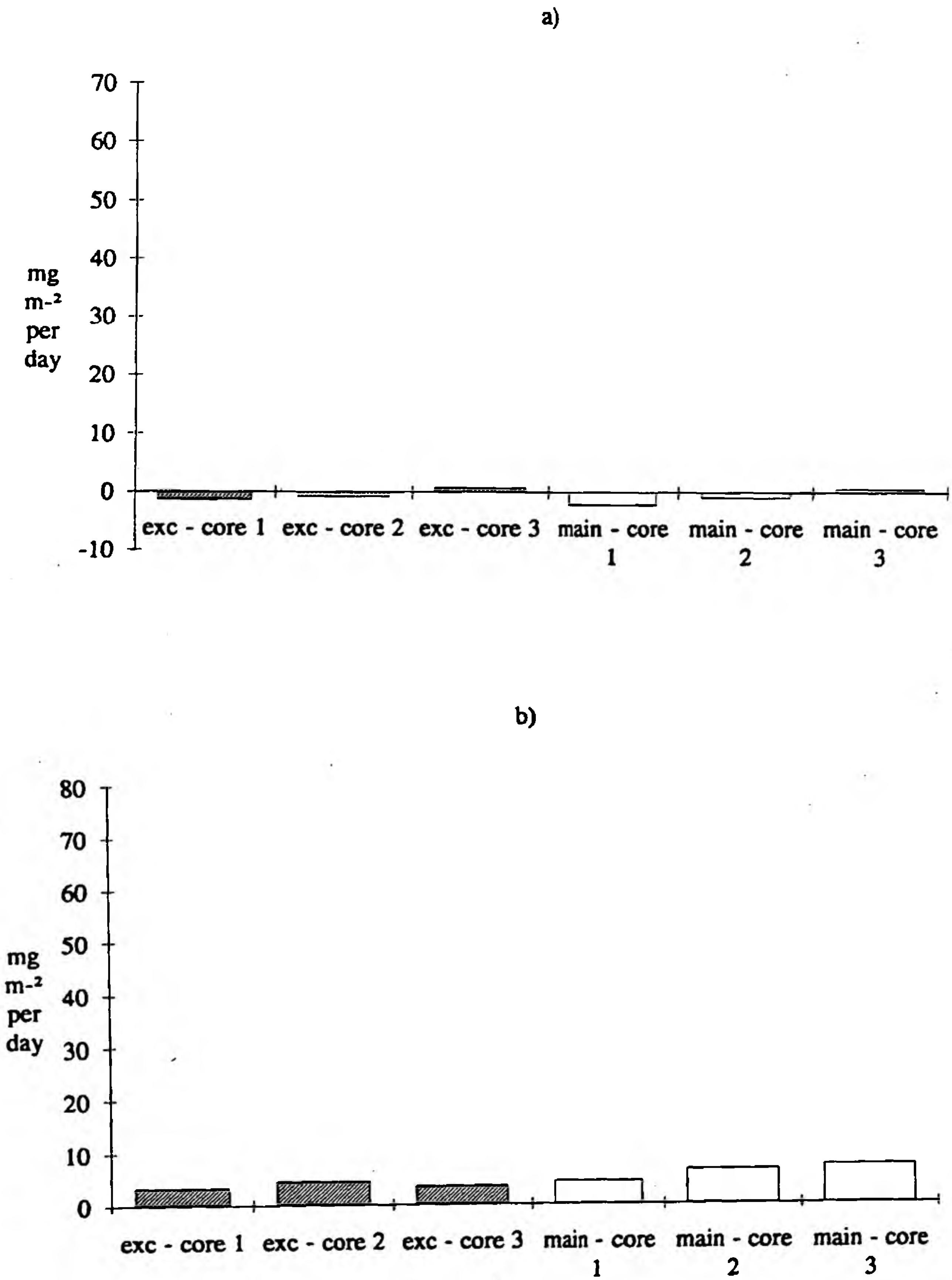


Figure 8.2 a)Average and b) maximum experimentally determined phosphorus release rates (mg/m^2/day) from cores collected at Hoveton Great Broad on 5 June 1991.

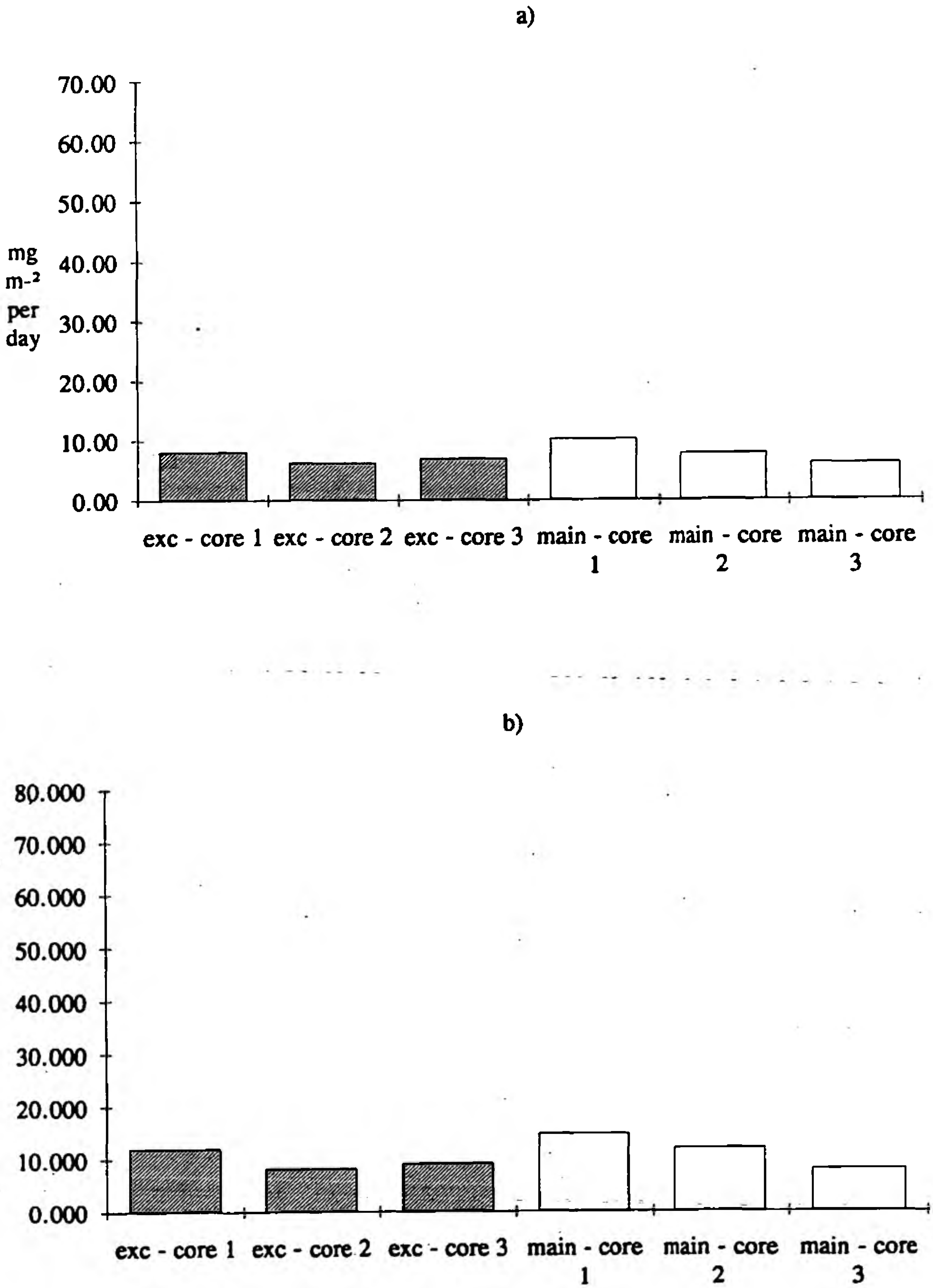


Figure 8.3 a)Average and b) maximum experimentally determined phosphorus release rates (mg/m^2/day) from cores collected at Hoveton Great Broad on 1 July 1991.

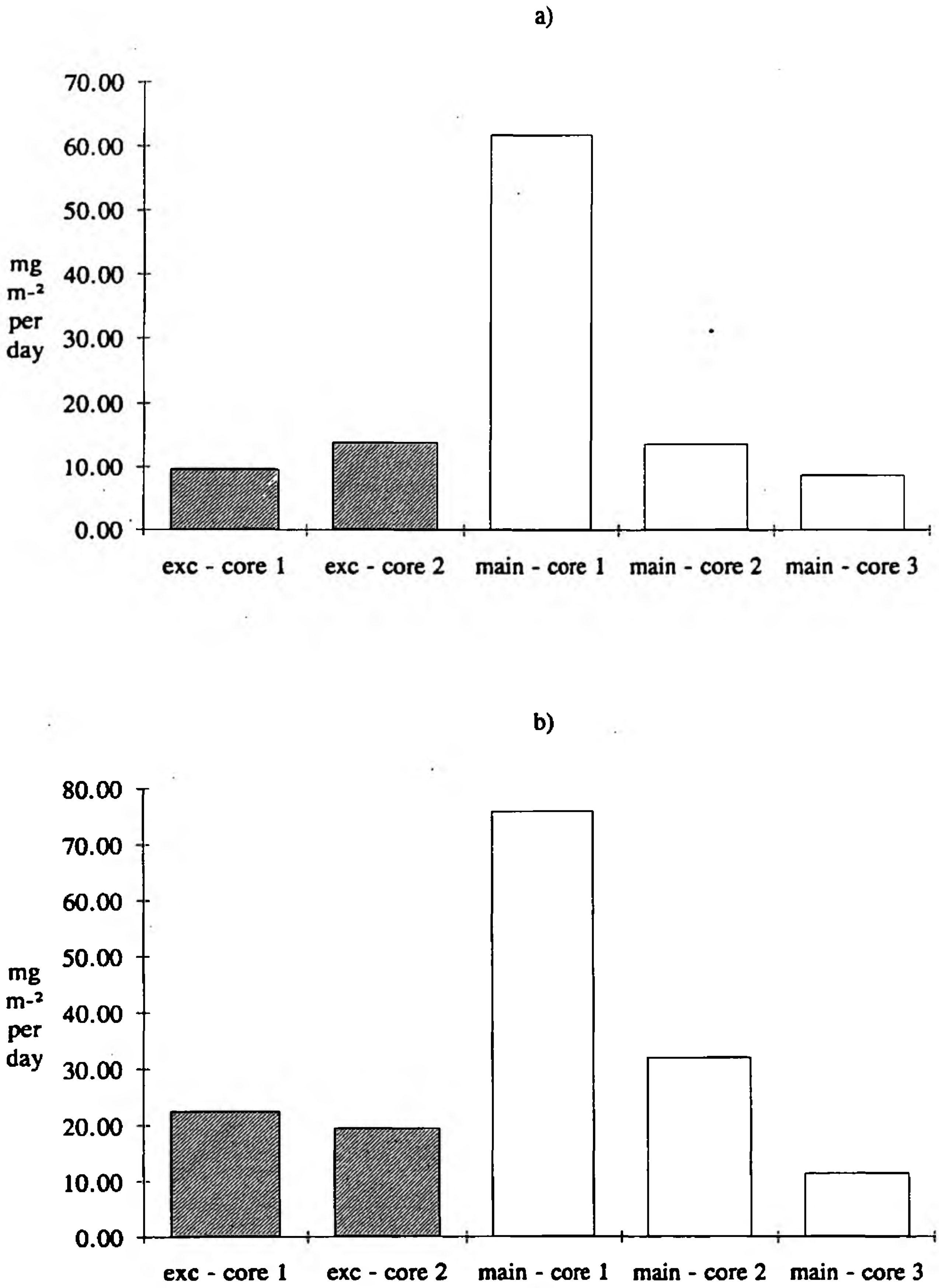
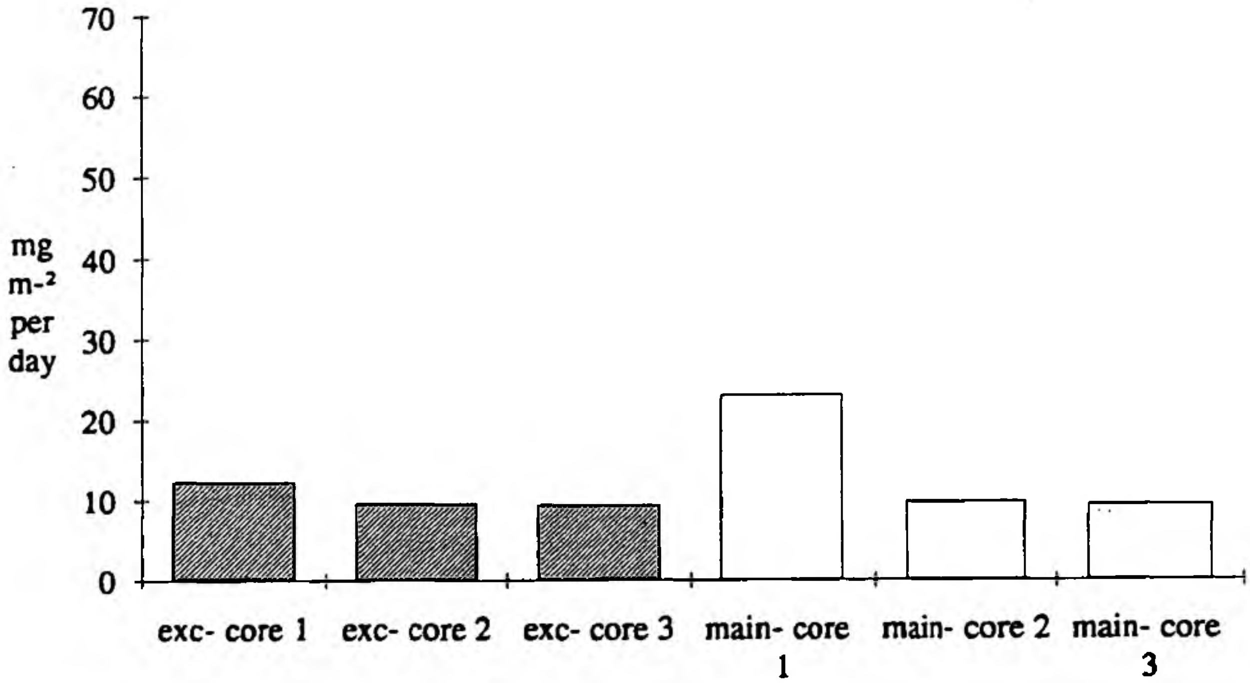


Figure 8.4 a)Average and b) maximum experimentally determined phosphorus release rates ($mg/m^2/day$) from cores collected at Hoveton Great Broad on 30 July 1991.

a)



b)

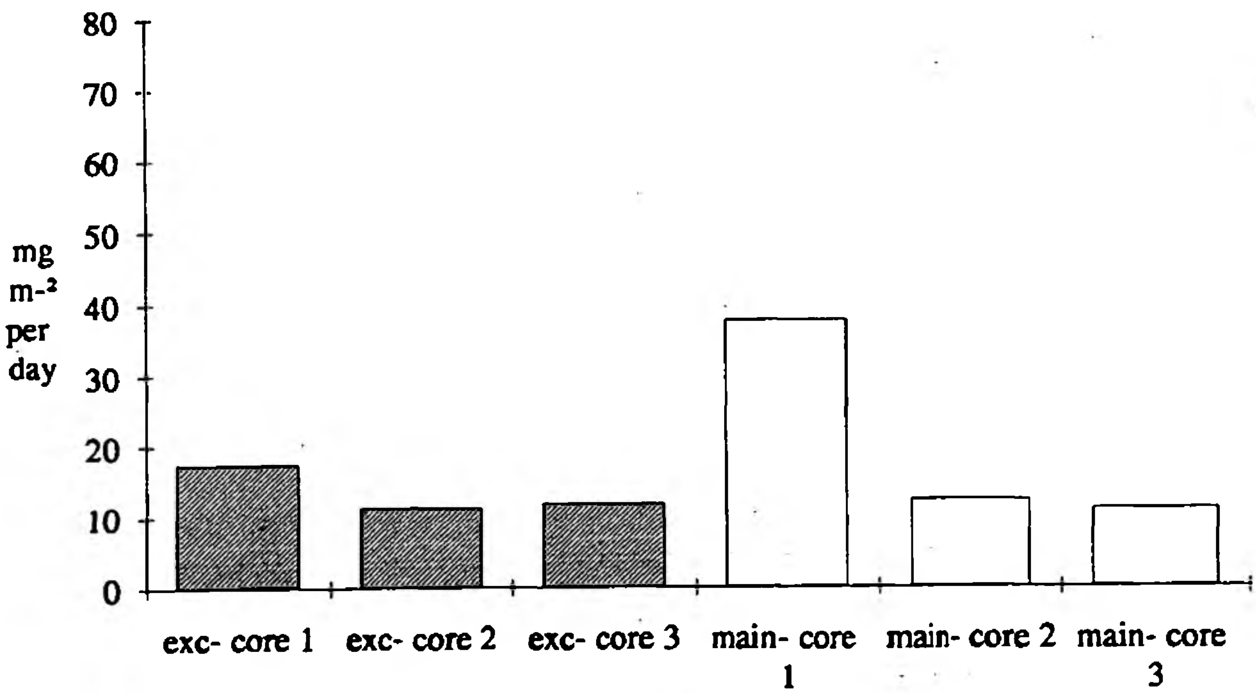


Figure 8.5 a)Average and b) maximum experimentally determined phosphorus release rates (mg/m^2/day) from cores collected at Hoveton Great Broad on 27 August 1991.

Release experiments

These were carried out as described by Jackson and Phillips (1990). Filtered water from Ormesby Broad was used in the release experiment as this had a low soluble phosphorus concentration over the summer period.

Sediment Analysis

The redox, pH, soluble reactive phosphorus and ferrous iron were all carried out using the method described by Jackson and Phillips (1990). The soluble sulphide was also measured using a method developed by Jackson (unpublished). Chemical analysis of the cores was performed the day after collection. The cores were placed in an incubator overnight, with the top bung removed, at ambient temperature. Redox and pH measurements were taken on the day of collection whenever this was possible otherwise the cores were stored overnight.

8.3 Results

Phosphorus Release Experiments.

The results of these experiments are shown in figures 8.1 to 8.5. The results for each month are presented on a separate graph. Very little release was seen from both areas over the summer except on one occasion when release occurred in the main broad. This was on the 30th July from one core(figure 8.4) when phosphorus was released for the duration of the experiment at an average rate of 61.7mg/m^2/day. The other cores did release small amounts at the beginning of the experiment but this soon tailed off.

In August(figure 8.5) some release was again seen from one core from the main broad where the average release rate was 23.08mg/m^2/day.

Sediment Profiles

The sediment profiles for each month for the determinands soluble reactive phosphorus, ferrous iron, soluble sulphide and redox are shown in figures 8.6 to 8.10.

At the beginning of the summer, May (figure 8.6) we can see that in both areas the sediment is very oxidised even at depth. Although there is no iron present to prevent release there is also very little phosphorus present that could be released. Sulphide levels are also low although at 8 cm there is a substantial amount present in the main broad.

By June (figure 8.7) the sediment has become more anoxic although the surface layers in both areas are still oxidised. In the surface layers more phosphorus is present and ferrous iron has appeared. Thus conditions are not suitable for phosphorus release to take place. Sulphide levels have also increased at depth, there being more in the exclosure than the broad. Where the sulphide is present there is no ferrous iron and the phosphorus levels have increased accordingly. In the exclosure the phosphorus levels are higher than the broad as would be expected with the high sulphide levels.

At the beginning of July (figure 8.8) phosphorus levels in the sediment have fallen in both the broad and the exclosure, although levels are still higher in the exclosure than the broad. Ferrous iron levels in the surface layers are higher than June. Levels in the broad are higher than the exclosure but the iron is only present in the top three centimetres compared to the top six centimetres in the exclosure. Sulphide levels have fallen substantially in both areas although the exclosure again has more. Again release would not be expected because of the oxidised state of the sediment, the low phosphorus levels and the prescence of ferrous iron in the surface sediment.

At the end of July (figure 8.9) when some release was seen, the sediment from both areas, had a very negative redox even in the surface layers. There is very little ferrous iron present and where found in the broad it is very patchy. Phosphorus is present throughout the sediment profile, although more is present in the exclosure than the broad. Sulphide levels have increased since the beginning of July, again there is more present in the exclosure than the broad. Phosphorus levels follow closely the fluctuations in sulphide concentration down the profile. In this case it would be possible for phosphorus to be released and we can see that in figure 8.4, there was some release from all the cores and a substantial amount from one core. Surprisingly this core was from the main broad.

At the end of August (figure 8.5) there was a small amount of release from the cores collected. The quantity of phosphorus in the sediment profile (figure 8.10) is greater than was present in July but there is more ferrous iron in the surface layers which will restrict the amount of phosphorus that can be released. Also the surface layers of the sediment are oxidised preventing much phosphorus from being released. No sulphide was detected.

8.4 Discussion

In summary, sediment release can only be expected when certain conditions are present in the sediment. These conditions were only found on one occasion this year. This was at the end of July when some release was seen. Otherwise the sediment either did not contain enough soluble reactive phosphorus or when there was enough present its release was countered by the prescence of iron and oxidised conditions.The exclosure tended to have more soluble reactive phosphorus and sulphide during this summer although there was no phosphorus release. Due to the fish presence in the exclosure the effect of chironomids on sediment phosphorus release could not be determined. In the coming year monitoring will continue. We should have an opportunity to study the effect of chironomids on phosphorus release.

8.5 References

Jackson, R.H.J., and Phillips G.L. (1990) The importance of sediment release in the restoration of the Norfolk Broads. National Rivers Authority publication.