



NRA

**THE IMPORTANCE OF SEDIMENT RELEASE  
OF PHOSPHORUS IN THE RESTORATION  
OF THE NORFOLK BROADS**

G.L. Phillips and R.H. Jackson

*National Rivers Authority  
Anglian Region*

THE IMPORTANCE OF SEDIMENT RELEASE OF PHOSPHORUS  
IN THE RESTORATION OF THE NORFOLK BROADS.

Summary report for the Nature Conservancy Council Contract HF3-03-350  
Monitoring the extension of the phosphorus removal programme to the  
River Bure - Investigation of phosphorus dynamics at the  
sediment/water interface.

G.L. Phillips<sup>1</sup> and R.H. Jackson<sup>2</sup>

<sup>1</sup> National Rivers Authority, Anglian Region, Norwich.

<sup>2</sup> Nature Conservancy Council, working at National Rivers Authority,  
Anglian Region, Norwich.

**ABSTRACT**

An overview is presented of the changes in the phosphorus loading of the River Ant and Barton Broad since the introduction of phosphorus removal from the effluent discharged to the system. The phosphorus budget for Barton Broad indicates additional phosphorus input from the sediment.

Phosphorus release experiments were carried out on intact sediment cores from a number of sites in the Norfolk Broads and pore water profiles of ferrous iron and soluble phosphorus concentration, redox and pH were measured.

Sediment phosphorus release was shown to occur from a range of sediments. Phosphorus release appears to be seasonal (occurring in spring and summer), increased by the presence of *Chironomus plumosus* larvae, and related to low pore water concentrations of ferrous iron relative to the interstitial phosphorus concentration.



## INTRODUCTION

### Restoration of the Norfolk Broads

Norfolk Broadland consists of a network of lowland rivers and shallow lakes (Figure 1), called Broads, which formed in the 12th century when mediaeval peat diggings became flooded (Lambert *et al.*, 1960). Until the 1960's the open water of these lakes contained very clear water with abundant growth of submerged aquatic plants and associated animals (Ellis, 1965). In the last thirty to forty years most of the lakes have changed dramatically as a result of eutrophication; the water has become turbid with dense populations of phytoplankton and almost all these plants have disappeared (Moss, 1983). The mechanism of the disappearance of these plants is complex, but can be linked to the effect of eutrophication (Phillips *et al.*, 1978). These submerged plants provide cover, egg-laying sites and a food source for a wide variety of animal life and their disappearance has resulted in a general reduction in the diversity of wildlife in Broadland.

It is now widely accepted that phosphorus is the key element to control if eutrophication of lakes is to be reversed. The models of Vollenweider (1968), Dillon & Rigler (1975) and the OECD (1982) all relate phosphorus load to indicators of trophic status such as algal biomass. There are now many examples of lakes where phosphorus input has been reduced. Both the extent and speed of recovery are very variable and it is clear that these models are unable to reliably predict ecosystem change following phosphorus reduction. Sas (1989) has reviewed the data from a number of these schemes, including the Broads, and concludes that shallow well mixed lakes behave in a different way to deep, often stratified lakes. He states that this is probably due to a continued supply of phosphorus from the sediment of shallow lakes.

Evidence for the release of phosphorus from the sediment of the Norfolk Broads was first provided by Osborne & Phillips (1978) and Phillips (1984). The work of Wilkinson (1985) on Barton Broad indicated that phosphorus release was controlled by redox processes, releasing iron and manganese bound phosphorus, but the extent to which this internal phosphorus source could override the control of phosphorus inputs was not known. The objectives of this project were to obtain estimates of phosphorus release rates in the Broads and to determine the factors influencing them.

### Sediment phosphorus release

Lake sediment plays an important role in the phosphorus cycle. The relative importance of this role depends on the type of sediment and the trophic state of the lake. In oligotrophic lakes a major proportion of the phosphorus input is retained by the sediment: 81% phosphorus retention was measured in a Canadian lake (Schindler, Hesslein & Kipphut, 1977). In most eutrophic lakes there is a net annual deposition of phosphorus, but the ability of sediment to retain phosphorus decreases as lakes become more eutrophic. Sediment release of phosphorus is generally seasonal, occurring mainly during the summer months (Osborne, 1981, Jacoby *et al.*, 1982; Boers & Van Hese, 1988) when it can greatly enhance primary productivity.

Sediment phosphorus release is dependent on a number of interrelated factors (see review of Marsden, 1989): including, the oxic state of both the overlying water and the sediment; the pH of the overlying water and the sediment pore water; the soluble phosphorus concentration in the sediment pore water relative to that in the overlying water; the quantity of available phosphorus in the sediment; and physical effects such as temperature, bioturbation and sediment resuspension.

The release of phosphorus from lake sediment is well known. One of the first studies on sediment phosphorus release was that of Mortimer (1941, 1942) who demonstrated that under oxidised conditions lake sediments bind phosphorus, whereas under reducing conditions phosphorus is released. Phosphorus release from reduced sediment was explained by the reduction of iron (III) and manganese (IV) to the soluble forms, iron (II) and manganese (II), releasing associated phosphorus into solution in the sediment pore water. If this reduced sediment is overlain by anoxic water, then phosphorus is released into the overlying water by diffusion. Sediment becomes reduced when the oxygen supply from the overlying water is insufficient for the mineralisation processes in the sediment; when the oxygen supply is depleted, anaerobic respiration takes over and, as this requires electron acceptors, this produces a reducing environment. These events generally occur in stratified lakes where the lower isolated water (hypolimnion) frequently becomes depleted of oxygen. For many years phosphorus release was only thought to occur in deep eutrophic lakes with anoxic hypolimnia.

However, more recently, a number of investigations have shown that phosphorus release can also occur under aerobic conditions (Boström, Jansson & Forsberg, 1982; Marsden, 1989), and can be of the same order of magnitude as anaerobic release. Aerobic phosphorus release is particularly important in well mixed, shallow lakes like the Norfolk Broads where the ratio of sediment surface area to overlying water is high and the released phosphorus is immediately available for primary production. Aerobic phosphorus release includes both release of phosphorus from anaerobic sediment and aerobic sediment into oxygenated water.

In lakes with anaerobic sediment and an aerobic water column, the anaerobic sediment is normally separated from the overlying water by an oxidised layer of sediment, which may be only a few millimetres thick (Fillos, 1977). This situation was observed in Barton Broad sediment by Wilkinson (1985). The thickness of the aerobic layer is determined by the oxygen supply from the overlying water and the microbial oxygen demand in the sediment. The presence of a layer of photosynthetic benthic algae may help to maintain the aerobic state of the surface sediment (Wilkinson, 1985). At high temperatures; however, the microbial oxygen demand may destroy this oxidised surface layer and allow the anaerobic pore water direct access to the overlying water. Where the oxidised surface layer remains intact, the major factor determining phosphorus release is the ratio of iron to phosphorus. In the aerobic layer, the oxidation of ferrous iron to ferric should precipitate phosphorus; however, if the atomic ratio of iron to phosphorus is  $<1.8$  then phosphorus will be released (Holdren & Armstrong, 1986). Sediments with high interstitial Fe:P ratios will be unlikely to release phosphorus. However, if hydrogen sulphide is

present in the sediment, this can precipitate interstitial iron as ferrous sulphide and will therefore allow phosphorus release (Holdren & Armstrong, 1986). Phosphorus release in anaerobic sediment can also be assisted by upward movement of gas bubbles of hydrogen sulphide or methane.

In aerobic sediment, phosphorus is released from sediment particles into the pore water mainly by desorption from inorganic complexes, such as iron, manganese and aluminium hydroxides, and the mineralisation of organic phosphates. Desorption of phosphorus from iron-hydroxyphosphate complexes is a result of ligand exchange with hydroxyl ions and is pH dependent (Lijklema, 1977). At pH values between 8 and 9 the rate of phosphorus release is correlated with the pH; above pH 9, phosphorus is precipitated by adsorption onto calcite forming hydroxyapatite (Andersen, 1975).

pH-dependent phosphorus release can also occur from resuspended sediment when the pH of the overlying water is high (Rippey, 1977); at lower pH, the resuspended sediments tend to adsorb phosphorus (Holdren & Armstrong, 1980).

In aerobic sediment from the Loosdrecht Lakes in the Netherlands, mineralisation of organic matter was shown to be the main source of released phosphorus (Sinke & Cappenberg, 1988; Sinke *et al.*, 1990). Mineralisation of organic matter is more rapid under aerobic rather than anaerobic conditions and can result in phosphorus release directly into the overlying water.

Transfer of soluble phosphorus from the sediment pore water into the overlying water can be assisted by any factors which cause turbulence above or in the surface sediment layer including: water currents, resuspension of sediment and gas bubbling, as already mentioned; and also bioturbation. The presence of tubificid worms and chironomid larvae in sediment cores from Lake Mendota were found to have a greater effect on phosphorus release rates than other physical or chemical mechanisms (Holdren & Armstrong, 1980). Granéli (1979) suggested that the greatly enhanced phosphorus release after the addition of *Chironomus plumosus* larvae to sediment cores from Swedish lakes was probably a purely physical mechanism, due to their respiratory movements; however, Andersen & Jensen (In press) concluded that increased sediment phosphorus release occurred by stimulating microbial decomposition of organic matter.

## **BACKGROUND**

### **Phosphorus reduction in the River Ant and Barton Broad.**

In 1977 Anglian Water began an experimental programme of phosphorus removal from the effluent of sewage treatment works discharging to the River Ant in an attempt to reverse the effects of eutrophication. In 1986 this work was extended to include selected sewage works discharging to the River Bure.

Phosphorus is removed from sewage effluents by chemical precipitation with ferric sulphate. Phosphorus combines with ferric sulphate to form an insoluble chemical floc which can be sedimented out in the final sediment tanks of the sewage treatment process.

The reduction of phosphorus discharged to the River Ant began in 1977 with the introduction of ferric dosing at Stalham STW's. In 1980, effluent from North Walsham STW's was diverted to a marine discharge at Mundesley and between 1980 and 1983 phosphorus removal was installed at an industrial discharge and three remaining sewage discharges in the River Ant catchment.

Prior to the introduction of phosphorus removal, the River Ant received approximately  $47\text{kg day}^{-1}$  of phosphorus, the bulk of this ( $39\text{kg day}^{-1}$ ) from sewage treatment works discharging directly to the River Ant upstream of Barton Broad. In 1978, when Anglian Water began its research programme, the sewage effluent was reduced to  $32\text{kg P day}^{-1}$  and since 1981 has averaged  $2\text{kg P day}^{-1}$ . Other point sources discharging to a tributary of the River Ant (Smallburgh) have also been reduced and the amount of phosphorus discharged to the River Ant upstream of Barton Broad is now approximately  $4\text{kg day}^{-1}$ , a 90% reduction (Figure 2).

These changes have resulted in a reduction in the phosphorus load in the River Ant (Figure 3). However, this reduction in load decreases in a downstream direction: at Honing lock (the tidal limit of the River Ant, 1.5km upstream of Barton Broad) the phosphorus load has been reduced by 77% but at the inflow to Barton Broad the reduction in load is only 58% suggesting additional phosphorus input. A comparison of the phosphorus loading of the tributaries and major discharges along the River Ant with the increase in loading along the river cannot account for this increased phosphorus loading, indicating phosphorus input from the river sediment.

These budgets only represent net movement of total phosphorus and do not take into account partitioning between the soluble and particulate components. After 1980 the greatest reduction in phosphorus was in the soluble component and hence movement in the particulate component became relatively more important. Upstream of Barton Broad intense boat activity can keep sediment in suspension (Hilton & Phillips, 1982) and this resuspended sediment could account for the increase in the total phosphorus load in the inflow to the broad.

#### **Nutrient budget for Barton Broad**

In general, since the start of the phosphorus reduction experiment, slightly more total phosphorus leaves Barton Broad than enters via the inflow (Figure 4), giving an average annual net loss of  $1.5\text{ kg day}^{-1}$  ( $2\text{ mg P m}^{-2}\text{ day}^{-1}$ ). The bulk of the phosphorus entering the broad is in the form of resuspended sediment, whilst the major component of phosphorus leaving the broad is in phytoplankton. As phytoplankton are only able to take up phosphorus in soluble form; transfer of phosphorus between the particulate and soluble phase must be taking place within the broad.

The average yearly net sedimentation/release of phosphorus was calculated for Barton Broad from the difference between the inflow and the outflow. If this is expressed as a function of the total input of phosphorus to the broad (Figure 5), the oscillation in the amount of release relative to the loading, subsequent to the introduction of phosphorus stripping in 1980, suggests that a new steady state between

sediment phosphorus release and total input is being approached (Sas, 1989). However, if the net sedimentation/release of phosphorus in the broad is expressed as a function of river flow (Figure 6), a linear relationship is found between the net release of phosphorus in the broad and the flushing rate, during the period after the introduction of phosphorus stripping (post 1980). The lower the flushing rate, hence the longer the residence time, the greater the release of phosphorus from the sediment; contrary to expectation. Net phosphorus sedimentation is equal to total phosphorus deposition minus the phosphorus released from the sediment. Extrapolating this regression back to zero river input, ie zero deposition, would give an estimate of the total phosphorus released from the sediment in the broad; which from these data is equal to  $8.8 \text{ g m}^{-2} \text{ yr}^{-1}$ .

## **METHODS**

### **Sediment sampling**

Sediment cores were taken from Barton Broad at regular intervals during the latter half of 1987 and throughout 1988. Cores were also taken from a number of other broads during August and September 1988 to obtain comparative data. Undisturbed sediment cores were collected using perspex tubes and a relatively simple corer (Figure 7); described in Jackson & Phillips, (In preparation). The soluble phosphorus and ferrous iron concentration of extracted pore water was determined and direct measurements were made of the pH and redox potential of the sediment cores. The gradient of soluble phosphorus across the sediment/water interface was used to calculate the potential diffusive flux (Berner, 1980). Incubations were carried out with intact cores to determine the rate of release or uptake of phosphorus from the sediment. The chironomid larvae and tubificid worms present in these cores were counted.

Water samples were collected from the surface of the broad and from just above the sediment surface and the soluble reactive phosphorus content analysed; the total phosphorus concentration of the surface water was also determined using the sulphuric acid/nitric acid digestion method (Standing Committee of Analysts, 1981).

### **Sediment analysis**

The redox profile of the top 10 cm of sediment was measured in intact cores using an array of platinum electrodes (Mortimer, 1941) and a calomel reference electrode. Sediment pH was also measured.

Sediment samples were removed from the top 12cm of the cores at 1cm intervals through sample holes in the tube wall using 60ml syringes and 12 gauge stainless steel needles. Pore water was extracted by filtration through GF/D pre-filters and  $0.45\mu\text{m}$  membrane filters held in Swinnex filter units and analysed for soluble reactive phosphorus, using the molybdenum blue method (Murphy & Riley, 1962), and ferrous iron, using the bathophenanthroline method (Nürnberg, 1984). This extraction method produced only a small volume of pore water (2-3ml) from each sediment sample and therefore the analytical methods had to be adapted for small volumes. As mentioned in the introduction, Barton Broad was known to have an oxidised surface sediment layer underlain by anaerobic sediment (Wilkinson, 1985). The pore water in

the anaerobic sediment contains ferrous iron which, under aerobic conditions, is oxidised to ferric and precipitates phosphorus as ferric phosphate. It was therefore necessary to avoid oxidation of the sediment during extraction and analysis; hence, the sampling syringes, syringe needles, membrane filter holders and reagents were purged with nitrogen prior to use. Anaerobic sediment pore water which contained little or no ferrous iron was assumed to contain hydrogen sulphide; the ferrous iron having been precipitated as ferrous sulphide. Hydrogen sulphide interferes with the molybdenum blue method for determining phosphorus (Nürnberg, 1984) and therefore pore water samples containing hydrogen sulphide were bubbled with nitrogen prior to the addition of reagent.

### **Sediment incubation experiments**

Sediment cores were incubated in a culture cabinet at the ambient temperature of the Broad surface sediment with continuous light. The cores were placed in a flow-through system (Figure 8) using filtered water collected from Barton Broad. Water from Barton Broad was used because, owing to algal uptake, its soluble phosphorus concentration is generally low ( $<0.01 \text{ mg l}^{-1}$ ). Water was pumped into the cores 1 cm above the sediment surface at  $120 \text{ ml hr}^{-1}$ ; the exchange time of the overlying water was 2.5 hours. Water was not recirculated around the system to avoid a gradual build up of phosphorus thus maintaining the greatest concentration gradient between the sediment and the overlying water. This continual removal of phosphorus-rich water from above the sediment was intended to mimic the effect of algal uptake of phosphorus in the broad and allow an estimation of the maximum rate of release. The length of the incubations varied between 75 and 200 hours. Samples were taken at frequent intervals from the inflow and outflow of the cores and the soluble reactive phosphorus concentration determined using the molybdenum blue method of Murphy & Riley (1962).

Three replicate cores were taken from Barton Broad on a number of dates to compare the phosphorus release or uptake on different occasions with the sediment profiles of interstitial phosphorus and ferrous iron concentrations. An experiment was carried out to see what effect addition of chironomid larvae and tubificid worms had on phosphorus release. Sets of replicate cores were also taken from a number of sites throughout Broadland to compare phosphorus release from a variety of sediments. The sampling sites are shown on the map (Figure 1).

## **RESULTS**

### **Phosphorus concentration in Barton Broad**

In 1979-80, prior to the introduction of phosphate stripping, high concentrations of total phosphorus entered Barton Broad (Figure 9a) from the River Ant (Figure 9b); most of this in soluble form. On entering the broad the soluble phosphorus concentration was reduced (Figure 9a), presumably because of algal uptake. However, comparing the soluble phosphorus concentration in the broad with that in the inflow, on two occasions (June-July 1979 and 1980), the concentration in the broad was greater than that in the inflow, indicating phosphorus release from the broad sediment.

Phosphate stripping has halved the total phosphorus and considerably reduced the soluble phosphorus concentration in the inflow to the broad, as shown by the values for 1978-88 (Figure 10b) in comparison with those from 1979-80 (Figure 9b); however, high total phosphorus concentrations are still detected in the broad during the summer months (Figure 10a), much of this being in the form of phytoplankton. For most of the year, the soluble phosphorus concentration in the surface water of Barton Broad is now less than  $0.01 \text{ mg l}^{-1}$ . In 1987, two peaks in soluble phosphorus were measured in the broad surface water in June and July (Figure 10a). As the phosphorus concentration in the broad on these occasions was higher than that in the inflow, this was indicative of sediment release in the broad. Measurement of soluble phosphorus in the surface water alone is not suitable for detecting sediment phosphorus release owing to rapid uptake of phosphorus by phytoplankton. Hence, soluble phosphorus was also measured in the water just above the sediment surface. The soluble phosphorus concentration above the sediment surface in 1987-88 (Figure 10a) suggests that sediment phosphorus release occurs throughout most of the summer.

#### **Interstitial concentration profiles from Barton Broad**

Comparison of the soluble phosphorus concentration in the pore water during 1987-88 (Figure 11a) with that in 1979-80 (Figure 11c), prior to the introduction of phosphate stripping, indicates that there has been a reduction in the amount of available phosphorus in the sediment since the onset of phosphorus removal. This would concur with the nutrient budget for Barton Broad which gives an average annual net loss of  $2 \text{ mg P m}^{-2} \text{ day}^{-1}$  since 1981.

There is considerable variation in the distribution of soluble phosphorus and ferrous iron in the sediment throughout the year. High soluble phosphorus concentrations in the top 5-6 cm of sediment were generally associated with high ferrous iron concentrations; in most cases the ferrous iron concentration was at least twice the soluble phosphorus concentration. In most cores, ferrous iron was not found below 5-6 cm because of the presence of hydrogen sulphide. High interstitial phosphorus concentration gradients near the sediment surface were not necessarily associated with periods of apparent sediment phosphorus release (Figures 10 and 11). In late June 1979 an increase in soluble phosphorus was detected in the broad but this was associated with a minimum in soluble phosphorus in the sediment. In the period from July to October 1979, high soluble phosphorus concentrations were measured in the sediment but there was no indication of phosphorus release during this period. In 1980, phosphorus release was detected in June when the interstitial phosphorus concentration was not particularly high but not in August when there was a very high phosphorus concentration gradient in the surface sediment layer. In August 1987 however, a high sub-surface sediment concentration gradient of interstitial phosphorus was associated with a period of apparent phosphorus release. In 1988, the soluble phosphorus concentration in the water overlying the sediment indicates phosphorus release occurred throughout the period from April to September but the interstitial phosphorus concentration in the surface sediment layer was low throughout this period.

The interstitial phosphorus concentration during the period from December 1979 to May 1980 seems inexplicably low. However, it must be noted that a different technique was used to sample the pore water during 1979-80 and this method did not ensure that the samples remained anaerobic. High ferrous iron concentrations were measured in the sediment during the winter of 1988 and had similarly high concentrations been present in the sediment during the winter of 1980, phosphorus may have been lost by precipitation as ferric phosphate if the pore water became oxidised.

### **Redox and pH**

From February to September 1988 the redox ( $E_h$ ) of the surface sediment of Barton Broad became progressively more negative (Figure 12), indicating an increase in the anoxic state of the sediment. A similar temporal variation in redox profiles from Barton Broad sediment was observed by Wilkinson (1985); this was shown to be significantly correlated with temperature and total sulphide concentration.

The pH of the surface sediment was lowest in February and highest in September (Figure 13). In May and September, the pH in the surface sediment exceeded 8 and therefore pH-mediated phosphorus release could have occurred from this layer (Andersen, 1975). The pH of the underlying sediment did not exceed 8; therefore, it is unlikely that the production of phosphorus in the pore water was a result of dissolution from ferric and aluminium hydroxyphosphates (Lijklema, 1977). The pH of the overlying water in Barton Broad exceeded 8 throughout most of the spring and summer of 1987 and 1988 (Figure 14) due to high algal production, as shown by the chlorophyll a concentration. Hence, pH-mediated phosphorus release could have occurred in the water column from resuspended sediments (Rippey, 1977).

### **Phosphorus release rates from Barton Broad sediment**

A distinct seasonality in experimentally determined phosphorus release rates was observed (Table 1). Sediment phosphorus release appears to occur only in the spring and summer months. During the winter months, as indicated by the cores sampled in February, the sediment adsorbs phosphorus.

Concentrations of interstitial phosphorus and ferrous iron were measured at the start of each experiment. The potential diffusive flux was calculated from the concentration gradient of phosphorus across the sediment-water interface. The lack of relationship between the experimentally determined release rates and the calculated diffusive fluxes, indicates that sediment release was not a solely diffusive process.

Table 1. Average experimentally determined phosphorus release rates from Barton Broad sediment cores, compared with the calculated potential diffusive release rates determined from the concentration gradients of phosphorus across the sediment/water interface; the ratio of ferrous iron to interstitial phosphorus; and the density of 4th instar *Chironomus plumosus* larvae. Release rates are the mean daily release rate of 3 cores. Values in parentheses = 1 standard deviation of the mean.

Date	Ambient temp. (°C)	Incubation temp. (°C)	Release rate (mg m <sup>-2</sup> d <sup>-1</sup> )	Diffusive flux (mg m <sup>-2</sup> d <sup>-1</sup> )	Fe:P	<i>C. plumosus</i> larvae (No. m <sup>-2</sup> )
11/2/88 <sup>a</sup>	3.5	20.0	-13.6 (0.1)	0.4	5.6	ND
11/2/88 <sup>b</sup>		20.0	-65.3 (1.1)			
3/5/88	13.5	13.5	62.0 (13.5)	0.6	7.6	ND
20/6/88	20.5	20.5	130.8 (31.9)	1.3	2.1	1870 <sup>+</sup>
27/6/88	17.5	17.5	80.2 (30.7)	ND		ND
4/7/88	17.8	18.0	30.4 (5.9)	ND		ND
18/7/88	18.5	18.5	33.6 (12.8)	1.6	4.2	800 (710)
13/9/88	14.5	20.0 <sup>*</sup>	2.9 (1.9)	3.1	2.1	90 (150)

<sup>a</sup> Phosphorus concentration in inflow water = concentration in surface water from Barton Broad (0.02 mg P l<sup>-1</sup>).

<sup>b</sup> additional phosphorus (0.25 mg P l<sup>-1</sup>) added to inflow water after 6 days.

<sup>\*</sup> Incubation for first 3 days at ambient temperature of broad sediment and then temperature increased.

<sup>+</sup> Only one sediment core was counted.

ND Not determined

The seasonality in sediment phosphorus release was not purely temperature dependent, as increasing the incubation temperature of the cores sampled in February from the ambient temperature of 3.5°C to 20°C did not bring about phosphorus release. The high Fe(II):P ratio (5.6:1) in the upper sediment layer in February could be responsible for the high phosphorus adsorption rate of the sediment (65 mg m<sup>-2</sup> day<sup>-1</sup>). However, in May the Fe(II):P ratio was also high (7.6:1), yet significant phosphorus release was measured (62 mg m<sup>-2</sup> day<sup>-1</sup>). The maximum phosphorus release rate was obtained from cores sampled in June, coinciding with a minimum in interstitial phosphorus in the sediment (Figure 11a). These factors indicate that the processes involved in phosphorus release are not purely physical or chemical.

The high release rate obtained from the June cores also coincided with the presence of large numbers of 4th instar *Chironomus plumosus* larvae in the sediment. Unfortunately the possible importance of chironomid larvae in phosphorus release was not recognised until mid-July and only the animals from one sediment core were counted in this experiment. Lower release rates in cores sampled in July and September coincided with lower numbers of chironomid larvae.

The experimentally determined phosphorus release rates from the Barton Broad sediment cores were plotted against time (Figure 15). The area under this graph gives the maximum total phosphorus release for the broad sediment and is equal to  $8.0 \text{ g m}^{-2}$ ; which is very close to the value of total phosphorus release ( $8.8 \text{ g m}^{-2}$ ) estimated for Barton Broad from the regression of net sedimentation rate against river flow.

### Bioturbation

The data from the experiments with Barton Broad sediment indicated that phosphorus release could be related to the density of chironomid larvae. Six cores were taken from Barton Broad and 30 large chironomid larvae and 30 tubificid worms were added to each of three cores after the cores had been incubated for 19 hours. There was a significant increase in release of soluble phosphorus from the sediment immediately following the addition of the animals (Figure 16). Table 2 shows the average release rate for each core and the numbers of chironomids and tubificids. There was a significant correlation ( $P < 0.002$ ) between the numbers of 4th instar *C. plumosus* larvae and the rate of phosphorus release, as was also found by Granéli (1979).

It was not possible to do any detailed work on chironomids in 1988 but in 1978 a survey was made of the numbers of chironomid larvae in Barton Broad sediment. Figure 17 shows the numbers of 4th instar *C. plumosis* larvae and the soluble reactive phosphorus concentration in the overlying water throughout 1978. There was a significant correlation ( $r = 0.89$ ,  $P > 0.001$ ) between the larval numbers and the phosphorus concentration.

**Table 2. Average release rates of soluble phosphorus from Barton Broad sediment cores compared with numbers of benthic animals. The surface area of each sediment core was =  $37.4 \text{ cm}^2$**

Core	Release rate ( $\text{mg m}^{-2} \text{ d}^{-1}$ )	<i>C. plumosus</i> larvae (4th instar)	Chironomid larvae (small)	Tanypoid larvae	Tubificid worms
1	19.1	1	6	7	31
2	38.1	2	7	1	23
3	43.6	6	20	10	32
4	78.1	24	6	2	32
5	72.0	25	8	10	54
6	60.0	29	21	10	70
Correlation (r)					
between release rate and number of animals per core		0.88	0.04	0.02	0.45

### Interstitial phosphorus profiles and experimentally determined phosphorus release rates from other Broads.

Sediment cores were collected from a number of sites in Broadland during August and September 1988. Sediment profiles of interstitial phosphorus and ferrous iron concentration and redox potential were measured for each site and the rate of sediment phosphorus release determined experimentally. A considerable range in release rates (Table 3) and interstitial phosphorus and ferrous iron concentrations (Figure 18) were found. The potential rates of diffusive release of phosphorus were calculated from the concentration gradients across the sediment-water interface and, as with Barton Broad, little correlation was found between these and the experimentally determined release rates.

The greatest phosphorus release rates were obtained not from sediments containing the highest concentration of interstitial phosphorus, ie those from Wroxham Broad and Hunsett Mill, but from sediments which contained high concentrations of hydrogen sulphide and hence no ferrous iron and had low redox potentials near the sediment surface, ie those from South Walsham and Ranworth Broads. The exception to this was the sediment from Upton Broad which adsorbed phosphorus

**Table 3. Average experimentally determined phosphorus release rates from various broads, compared with: calculated potential diffusive release rates determined from the concentration gradients of phosphorus across the sediment/water interface; the ratio of ferrous iron to interstitial phosphorus; and the density of 4th instar *Chironomus plumosus* larvae. Release rates are the mean daily release rate of 3 cores. Values in parentheses = 1 standard deviation of the mean.**

Site*	Date	Ambient temp. (°C)	Release rate (mg m <sup>-2</sup> d <sup>-1</sup> )	Diffusive flux (mg m <sup>-2</sup> d <sup>-1</sup> )	Fe:P	<i>C plumosus</i> larvae (No. m <sup>-2</sup> )
River Ant	13/9/88	16.0	3.3 (1.1)	28.1	1.7	0
Belaugh	17/8/88 <sup>a</sup>	18.5	0.7 (0.5)	1.2	3.0	0
Belaugh	17/8/88 <sup>b</sup>	18.5	-33.2 (3.5)			
Hoveton Gt	8/8/88	21.0	2.6 (15.2)	1.8	2.6	180 (540)
Ranworth	2/8/88	18.0	17.8 (9.2)	2.4	0	1340 (270)
South Wal.	2/8/88	18.0	194.3 (74.8)	6.7	0	800 ( 0)
Upton	17/8/88 <sup>a</sup>	21.0	-0.8 (0.2)	0.1	0	0
Upton	17/8/88 <sup>b</sup>	20.0	-62.8 (0.7)			
Wroxham	8/8/88	21.0	14.4 (2.1)	17.2	1.9	0

\* All sites are broads unless otherwise stated

<sup>a</sup> Phosphorus concentration in the inflow water = concentration in surface water from Barton Broad (<0.001 mg P l<sup>-1</sup>).

<sup>b</sup> Additional phosphorus (0.25 mg P l<sup>-1</sup>) added to inflow water after 5 days

rather than releasing it; although it was very anoxic, as shown by the redox profile, and contained no ferrous iron. The sediment from this broad is highly unusual; the top 8-10 cm is green and fluid being composed of a thick layer of the blue-green algae *Aphanothece stagnina* (Spreng.) A.Br. This algal layer contained virtually no soluble phosphorus; presumably the algae act as a phosphorus sink, which explains why this anoxic sediment adsorbed phosphorus rather than releasing it.

The sediments with the highest phosphorus release rates also contained the greatest numbers of 4th instar *Chironomus plumosus* larvae, which have already been shown to increase the rate of phosphorus release.

Sediments with high interstitial ratios of ferrous iron to phosphorus, particularly near the sediment surface, such as those from Belaugh Broad and Hoveton Great Broad, released very little phosphorus. The high Fe(II):P ratio (7.5:1) in the top 2-3 cm of Belaugh Broad sediment probably accounts for phosphorus uptake by this sediment. The very low interstitial phosphorus concentration in Belaugh Broad sediment is presumably due to the fact that this broad has had the top, more recently deposited sediment removed as part of a restoration programme. The highest interstitial phosphorus concentrations were measured in sediment cores from Wroxham Broad and from Hunsett Mill on the River Ant. The sediments from these two sites contained high concentrations of ferrous iron and had more oxidised surface layers, which probably explains the relatively low level of phosphorus release from these sediments.

## DISCUSSION

Sediment phosphorus release was shown to occur in a number of sites throughout Broadland. Incubation experiments with sediment cores indicated that sediment from some of the broads could release substantial quantities of phosphorus over short periods of time, ie a few days. Frequently, this sediment-released phosphorus was not detected in the surface water of the broads because of rapid uptake by phytoplankton. Since sediment phosphorus release occurs in the spring and summer months, it is obviously an important contributory factor in the development of large algal populations.

One of the sites, Barton Broad, was studied in more detail. Sediment phosphorus release determined directly from incubated sediment cores, and indirectly from the difference between the soluble phosphorus concentration above the sediment surface and the concentration in the surface water, was shown to be a seasonal process, occurring only during the spring and summer. Evidence from these experiments and from the budget for Barton Broad suggests that during the autumn and winter the sediment adsorbs phosphorus. The experimentally determined phosphorus release rates were plotted against time and the total spring-summer release estimated. This value ( $8.0 \text{ g m}^{-2}$ ) was very close to that estimated from the regression of net sedimentation rate against river flow ( $8.8 \text{ g m}^{-2}$ ); whilst, both these estimates are likely to be inaccurate, owing to insufficient data points, they do, however, give some indication as to the extent of phosphorus release in Barton Broad. If all the sediment in the broad released phosphorus at this rate, this would represent an average annual gross sediment loss of  $17.1 \text{ kg P day}^{-1}$  for the whole broad, compared with the average

annual net loss of 1.5 kg P day<sup>-1</sup> from the budget calculation, indicating that most of the sediment-released phosphorus is returned to the sediment.

The restriction of phosphorus release to the spring and summer and the lack of release in the autumn and winter was not dependent on temperature alone: increasing the incubation temperature of cores sampled during the autumn and winter months did not increase phosphorus release. Nor was the difference in release between the two periods of the year due to the interstitial phosphorus concentration: high concentrations of interstitial phosphorus were measured near the sediment surface in autumn when no release was detected and the highest release rate from Barton Broad was obtained from sediment which contained the lowest interstitial phosphorus concentration. This implies that diffusion is not a controlling factor in phosphorus release in Barton Broad.

During the winter, the sediment tended to adsorb phosphorus as shown by the experiment in February and the phosphorus budget for Barton Broad (Osborne & Phillips, 1978). High concentrations of interstitial iron were determined during this period, which probably accounts for the phosphorus adsorption. Wilkinson (1985) measured a minimum in total sulphide during the winter, presumably due to a minimum in microbial activity; this would result in minimal precipitation of ferrous iron as ferrous sulphide, and hence explain the winter maximum concentrations of ferrous iron. Also, the redox profiles of this study and that of Wilkinson (1985) show that the surface sediment was more oxidised during the winter months and this would act as a barrier to phosphorus release due to precipitation of phosphorus by ferric iron.

During the summer, the surface sediment of Barton Broad became increasingly more reduced, and hence more anoxic, as shown by the redox potential profiles. Wilkinson (1985) showed that this increasing anoxia was correlated with temperature and sulphide concentration and therefore indicates increased microbial activity. This enhanced microbial oxygen demand could be sufficient to destroy the oxidised surface layer and allow anaerobic phosphorus release into the overlying water. Also, increased microbial activity would increase the degradation of organic matter and therefore increase the release of phosphorus into the pore water (Sinke et al., 1990). It is unlikely that phosphorus release from the sediment particles into the pore water was pH-mediated because the pH of the pore water did not exceed 8 (Lijklema, 1977), except in the surface sediment. However, there could have been pH-mediated release of phosphorus from resuspended sediments in overlying water of high pH (pH 8-9).

Bioturbation, particularly in the form of 4th instar *Chironomus plumosus* larvae, was shown to have a significant effect on sediment phosphorus release. Andersen & Jensen (In press) concluded that chironomid larvae affected phosphorus release because their bioturbating activity stimulated microbial decomposition of organic matter, as shown by increased sediment oxygen demand. Enhancement of phosphorus release could also be due to their pumping action moving phosphorus in interstitial water into the overlying water (Granéli, 1979). Andersen & Jensen (In press) estimated the irrigation rate of chironomids in summer to be 5.3 mg hr<sup>-1</sup>. Assuming an interstitial

concentration of  $0.5 \text{ mg P l}^{-1}$  and a chironomid larval density of  $1870 \text{ m}^{-2}$  (June 1988); this would give a phosphorus release rate of  $119 \text{ mg m}^{-2} \text{ day}^{-1}$  compared with the experimentally determined release rate of  $131 \text{ mg m}^{-2} \text{ day}^{-1}$  for June 1988. Chironomid larvae have been observed to burrow 5-10 cm into the sediment and could therefore penetrate the anaerobic sediment underlying the aerobic layer; thus crossing the barrier at the anaerobic-aerobic interface and allowing transfer of phosphorus-rich anaerobic water interstitial water into the water overlying the sediment surface.

Incubation experiments with cores from a number of sites throughout Broadland showed that maximum phosphorus release occurred from sediments which contained no ferrous iron and high numbers of chironomid larvae. Low phosphorus release rates were obtained from sediments which contained high interstitial concentrations of ferrous iron relative to soluble phosphorus and no chironomid larvae.

It would therefore appear that the most significant factors affecting phosphorus release from the sediment of the Norfolk Broads were the presence or absence of ferrous iron in the pore water and the degree of bioturbation by large chironomid larvae.

#### ACKNOWLEDGEMENTS

This work forms part of a wider investigation into the restoration of Broadland jointly funded by Anglian Water, the Nature Conservancy Council and the Broads Authority. R H Jackson was supported by a research contract awarded by the Nature Conservancy Council. We would like to thank Dr R A Watson for data collected in 1979-80 and other colleagues at the Anglian Water biology laboratory for assistance with sampling and numerous helpful comments. The views expressed in this report are those of the authors and should not be attributed to Anglian Water or the Nature Conservancy Council.

#### REFERENCES

- Andersen F.O. & Jensen H.S. (In press) The influence of chironomids on decomposition of organic matter and nutrient exchange in a lake sediment.
- Andersen J.M. (1975) Influence of pH on release of phosphorus from lake sediments. *Archiv für Hydrobiologie*, 76, 411-419.
- Berner R.A. (1980) *Early Diagenesis, A Theoretical Approach*. Princeton University Press, Princeton, N.J.
- Boers P.C.M & Van Hese O. (1988) Phosphorus release from the peaty sediments of the Loosdrecht Lakes (The Netherlands). *Water Research*, 22, 355-363.
- Boström B., Jansson M. & Forsberg C. (1982) Phosphorus release from lake sediments. *Archiv für Hydrobiologie: Ergebnisse der Limnologie*, 18, 5-59.
- Dillon P.J. & Rigler F.H. (1975) A simple method for predicting the capacity of a lake for development based on lake trophic status. *Journal of the Fisheries Research Board of Canada*, 32, 1519-1531.

Ellis E.A. (1965) *The Broads*. Collins, London.

Fillos J. (1977) Effect of sediments on the quality of the overlying water. *Interactions between Sediments and Freshwater* (Ed. H.L. Golterman), pp. 266-271. W. Junk, The Hague.

Granéli W. (1979) The influence of *Chironomus plumosus* larvae on the exchange of dissolved substances between sediment and water. *Hydrobiologia*, 66, 149-159.

Hilton, J. & Phillips, G.L. (1982) The effect of boat activity on turbidity in a shallow Broadland river. *Journal of Applied Ecology*, 19, 143-150.

Holdren G.C. & Armstrong D.E. (1980) Factors affecting phosphorus release from intact sediment cores. *Environmental Science and Technology*, 14, 79-87.

Holdren G.C. & Armstrong D.E. (1986) Interstitial ion concentrations as an indicator of phosphorus release and mineral formation in lake sediments. *Sediments and Water Interactions* (Ed. P.G. Sly), pp. 133-147. Springer, New York.

Jacoby J.M., Lynch D.D., Welch E.B. & Perkins M.A. (1982) Internal phosphorus loading in a shallow eutrophic lake. *Water Research*, 16, 911-919.

Lambert, J.M. Jennings, J.N. Smith, C.T. Green, C. & Hutchinson, J.N. (1960) The making of the broads: a reconsideration of their origin in the light of new evidence. *Royal Geographic Society Memoirs*, 3, 1-242.

Lijklema L. (1977) The role of iron in the exchange of phosphate between water and sediments. *Interactions between sediments and freshwater* (Ed. H.L. Golterman), pp. 313-317. Dr. W. Junk B.V., The Hague.

Marsden, M.W. (1989) Lake restoration by reducing external phosphorus loading: the influence of sediment phosphorus release. *Freshwater Biology*, 21, 139-162.

Mortimer, C.H. (1941) The exchange of substances between mud and water in lakes. I. *Journal of Ecology*, 29, 280-329.

Mortimer, C.H. (1942) The exchange of dissolved substances between mud and water, II. *Journal of Ecology*, 30, 147-201.

Moss B. (1983) The Norfolk Broadland: experiments in the restoration of a complex wetland. *Biological Review*, 58, 521-561.

Murphy, J. & Riley, J.P. (1962) A modified single solution method for the determination of phosphate in natural waters. *Analytica Chimica Acta*, 27, 31-36.

Nürnberg, G. (1984) Iron and hydrogen sulphide interference in the analysis of soluble reactive phosphorus in anoxic waters. *Water Research*, 3, 369-377.

OECD (1982) *Eutrophication of Waters: Monitoring Assessment and Control*. Organisation for Economic Co-Operation and Development, Paris.

Osborne P.L. (1981) Phosphorus and nitrogen budgets of Barton Broad and predicted effects of a reduction in nutrient loading on phytoplankton biomass in Barton, Sutton and Stalham Broads, Norfolk, United Kingdom. *Internationale Revue der Gesamten Hydrobiologie*, 66, 171-202.

Osborne, P.L. & Phillips, G.L. (1978) Evidence for nutrient release from the sediments of two shallow and productive lakes. *Verhandlungen internationale der Vereinigung für theoretische und angewandte Limnologie*, 20, 654-658.

Phillips. G.L. (1984) A large scale field experiment in the control of eutrophication in the Norfolk Broads. *Water Pollution Control*, 83, 400-407.

Phillips, G.L. Eminson, D.F. & Moss, B (1978) A mechanism to account for macrophyte decline in progressively eutrophicated freshwaters. *Aquatic Biology*, 4, 103-126.

Rippey B. (1977) The behaviour of phosphorus and silicon in undisturbed cores of Lough Neagh sediments. *Interactions between Sediments and Freshwater* (Ed. H.L. Golterman), pp. 348-353. Dr. W. Junk B.V., The Hague.

Sas H. (1989) *Lake restoration by reduction of nutrient loadings: expectations, experiences, extrapolations*. Academia Verlag Richarz, Sankt Augustin.

Schindler D.W., Hesslein R. Kipphut G. (1977) Interactions between sediments and overlying waters in an experimentally eutrophied Pre-Cambrian Shield Lake. *Interactions between Sediments and Freshwater* (Ed. H.L. Golterman), pp. 235-243. Dr. W. Junk B.V., The Hague.

Sinke A.J.C. & Cappenberg T.E. (1988) Influence of bacterial processes on the phosphorus release from sediments in the eutrophic Loosdrecht Lakes, The Netherlands. *Archiv für Hydrobiologie*, 30, 5-13.

Sinke A.J.C., Cornelese A.A., Keizer P., Van Tongeren O.F.R. & Cappenberg T.E. (1990) Mineralisation, pore water chemistry and phosphorus release from peaty sediments in the eutrophic Loosdrecht Lakes, The Netherlands. *Freshwater Biology*, 23, 587-599.

Standing Committee of Analysts (1981) Phosphorus in waters, effluents and sewages, 1980. *Methods for the examination of waters and associated materials*. HMSO, London.

Vollenweider R.A. (1968) *Scientific fundamentals of the eutrophication of lakes and flowing waters, with particular reference to phosphorus and nitrogen as factors in eutrophication*. OECD Technical Report, DAS/CSI/68.27, Paris.

Wilkinson D.E. (1985) *Chemical and biological studies of a shallow freshwater sediment interface*. Ph. D. Thesis, University of East Anglia, Norwich.

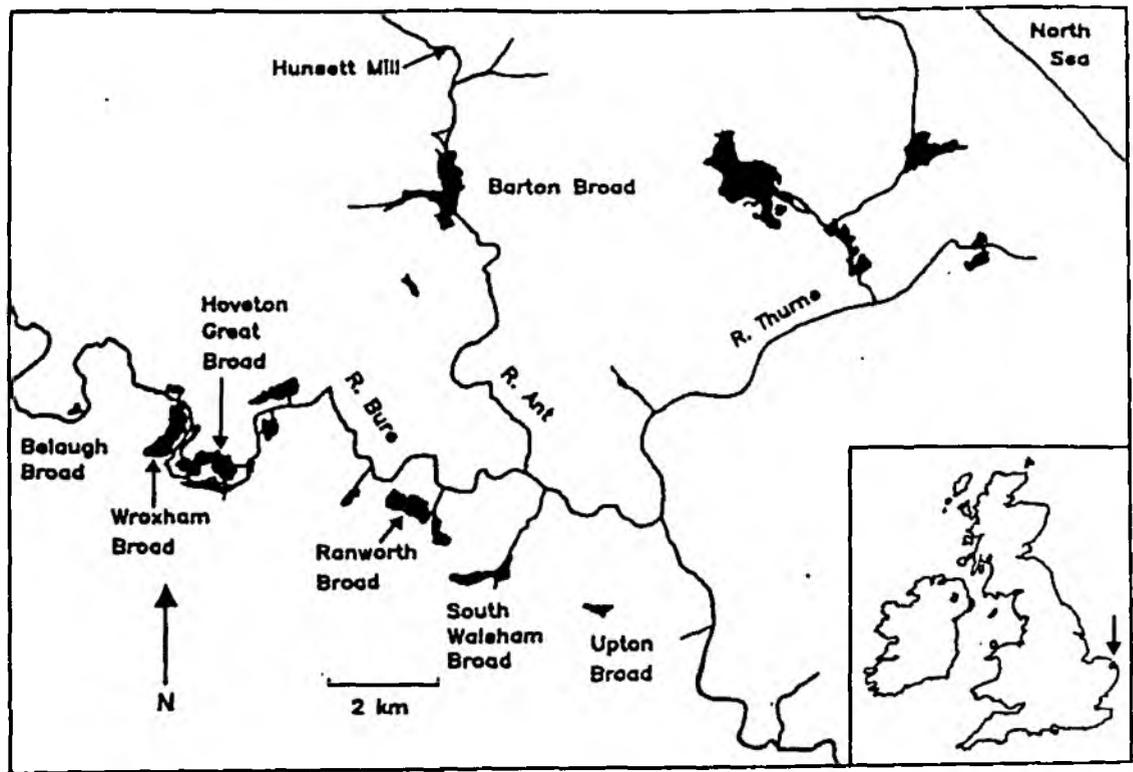


Figure 1. Map of the Norfolk Broads showing the sampling sites. The arrow on the inset map marks the position of the detailed map.

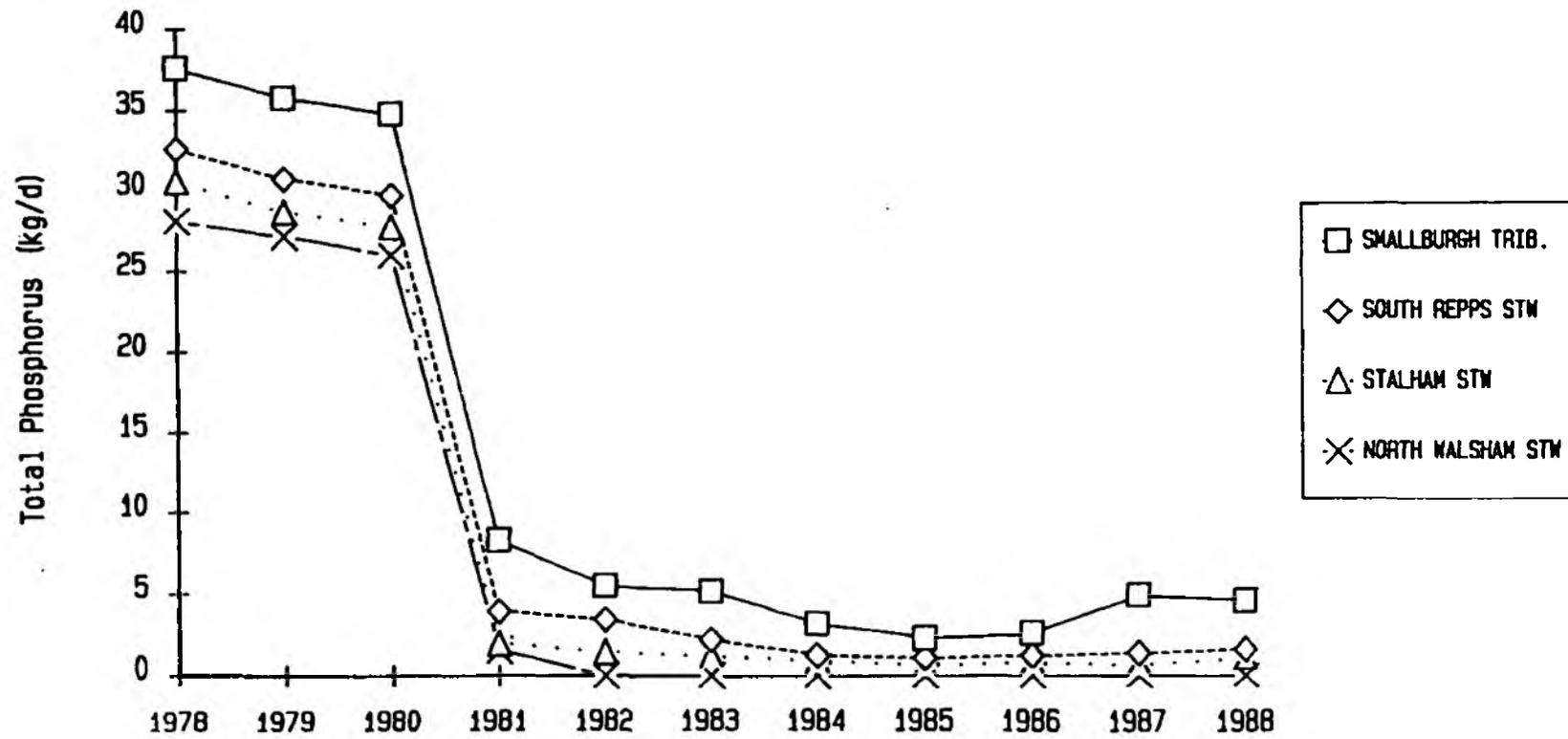


Figure 2. Average total phosphorus discharged to the River Ant from the principle sources between 1978 and 1988

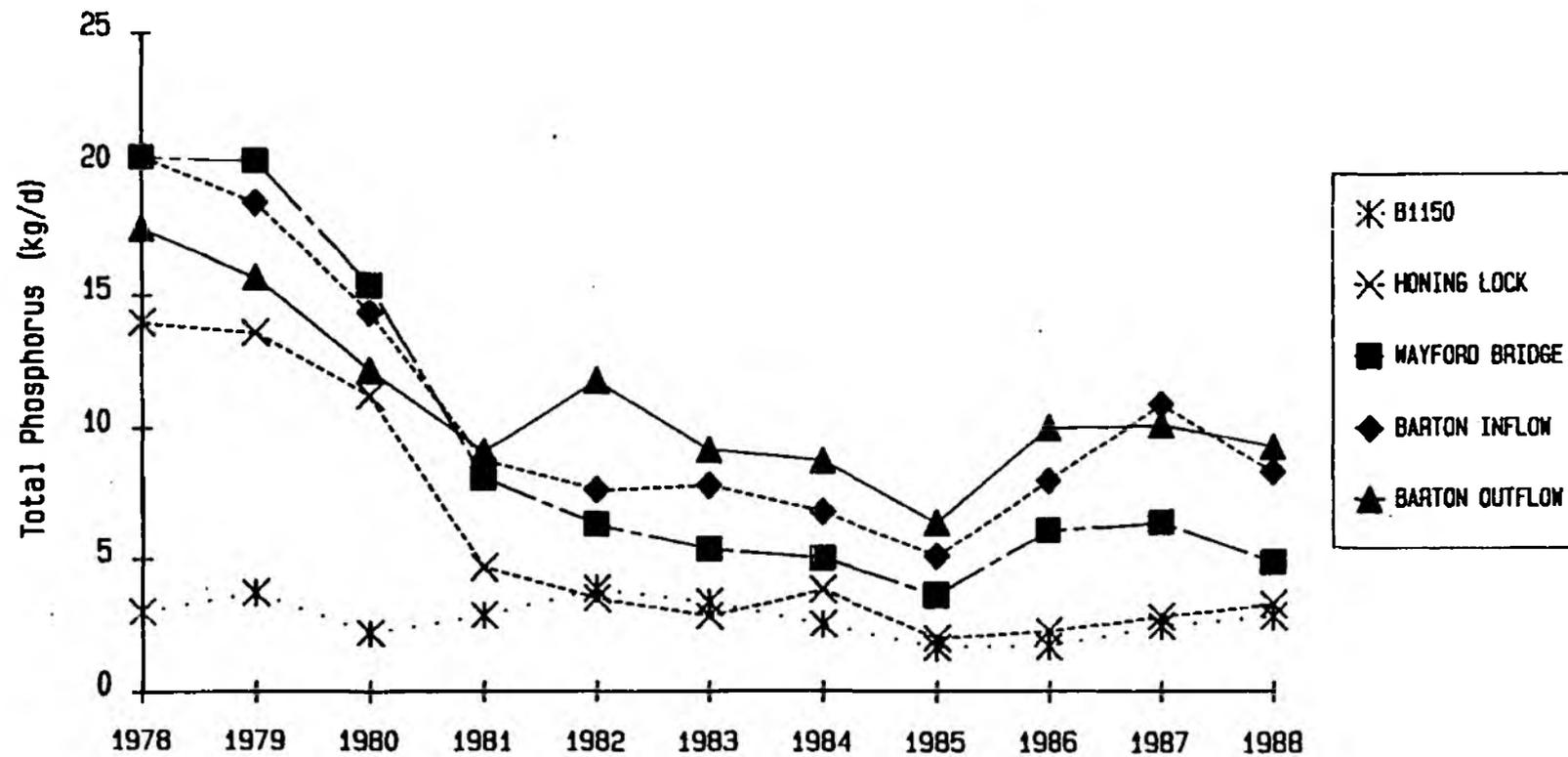


Figure 3. Average total phosphorus load measured at several points along the River Ant between 1978 and 1988

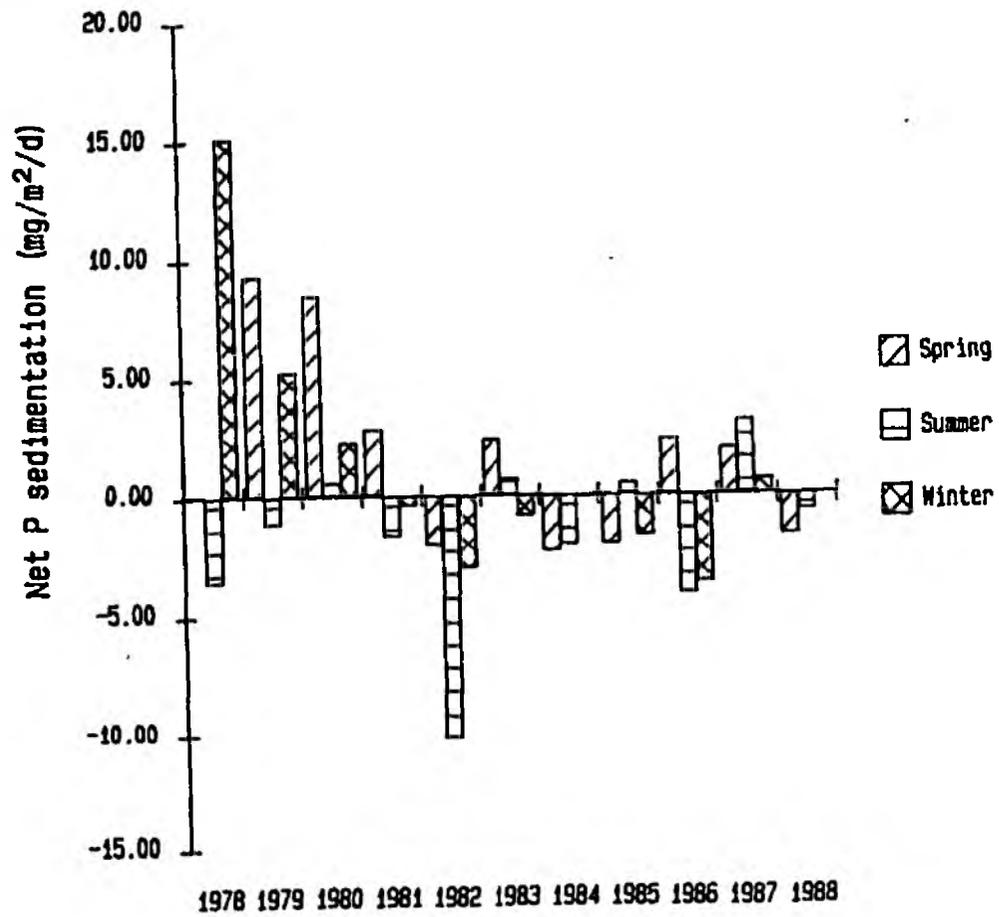


Figure 4. Net seasonal gain or loss of total phosphorus from Barton Broad between 1978 and 1988

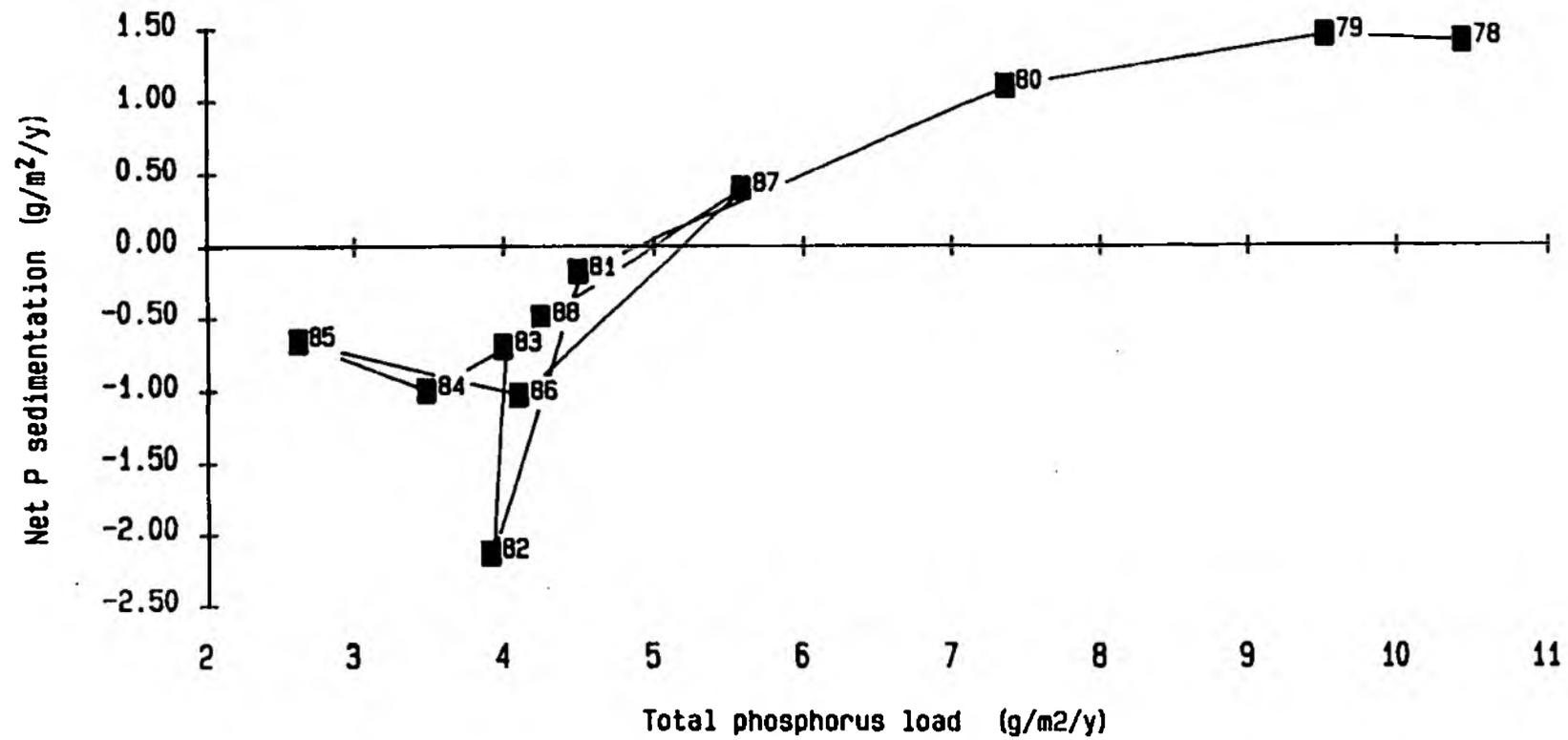


Figure 5. Net phosphorus sedimentation rate versus total phosphorus load for Barton Broad between 1978 and 1988

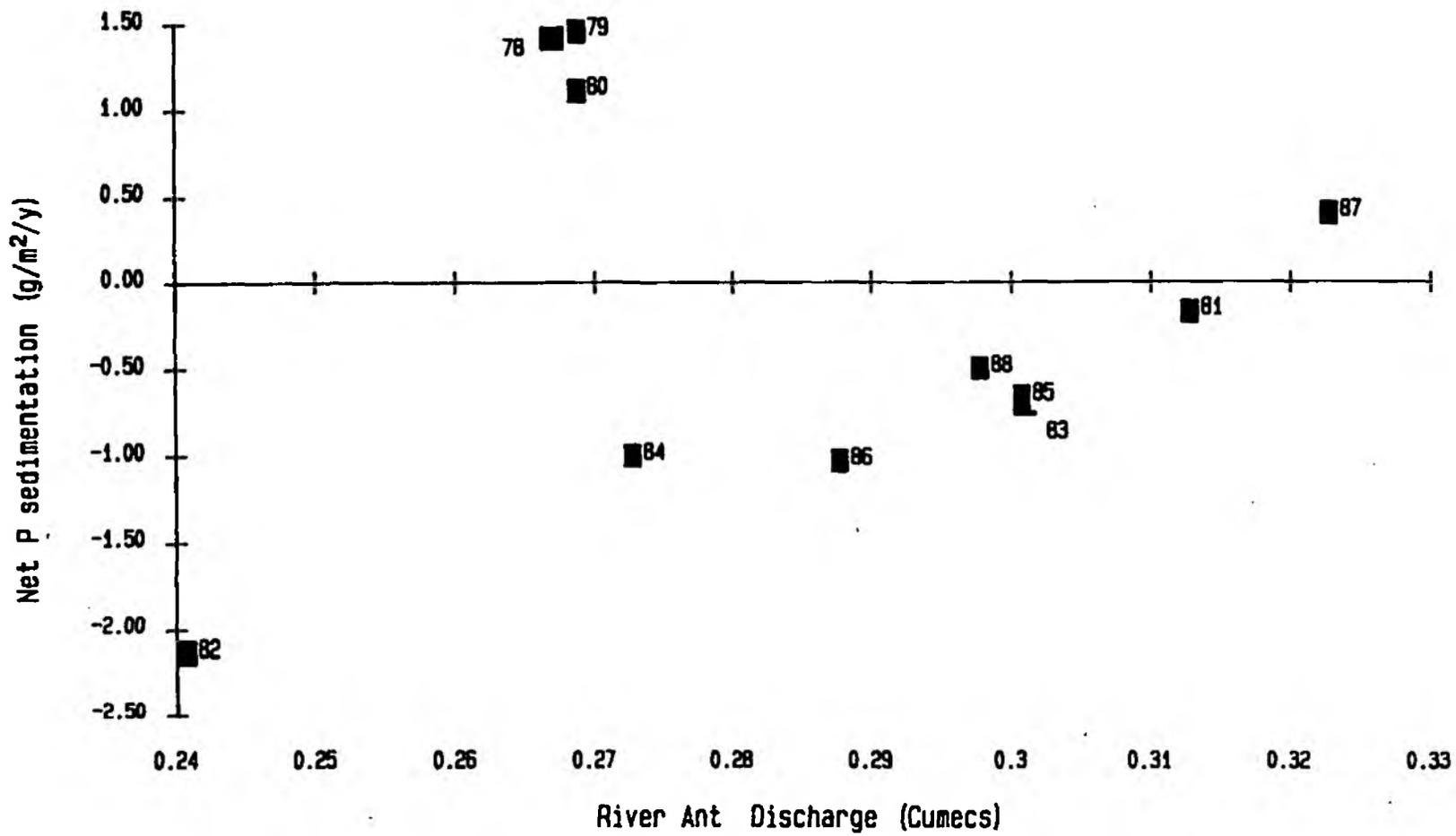


Figure 6. Net phosphorus sedimentation rate in Barton Broad between 1978 and 1988 as a function of the discharge to the broad from the River Ant.

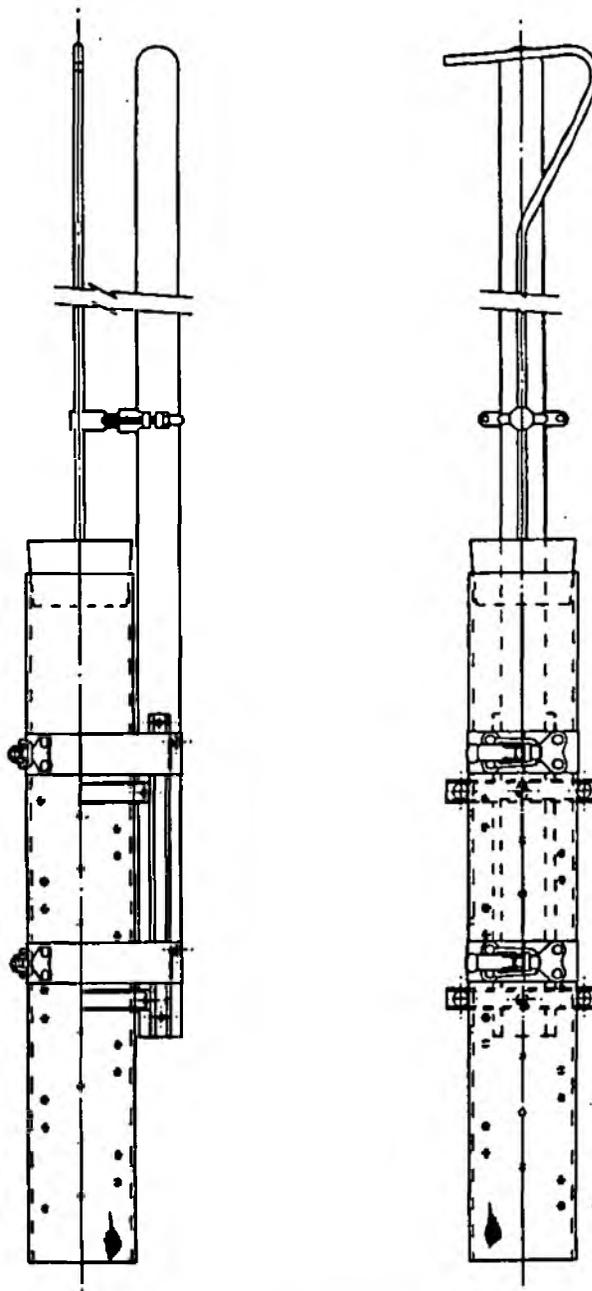


Figure 7. Diagram of sediment core sampler.

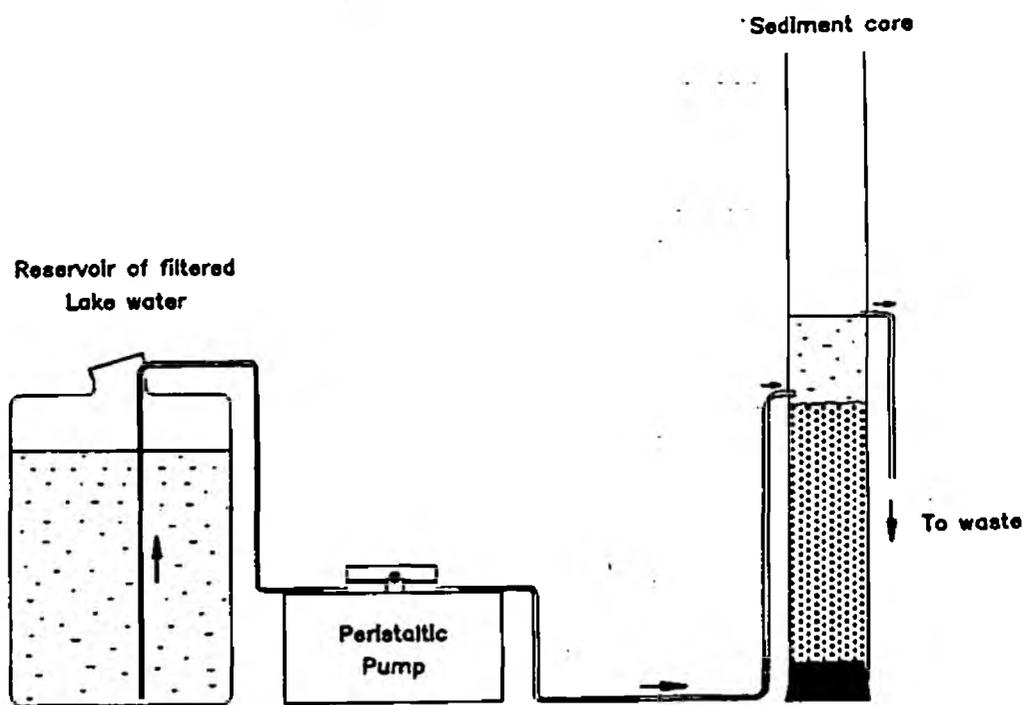


Figure 8. Experimental set-up, for incubating sediment cores.

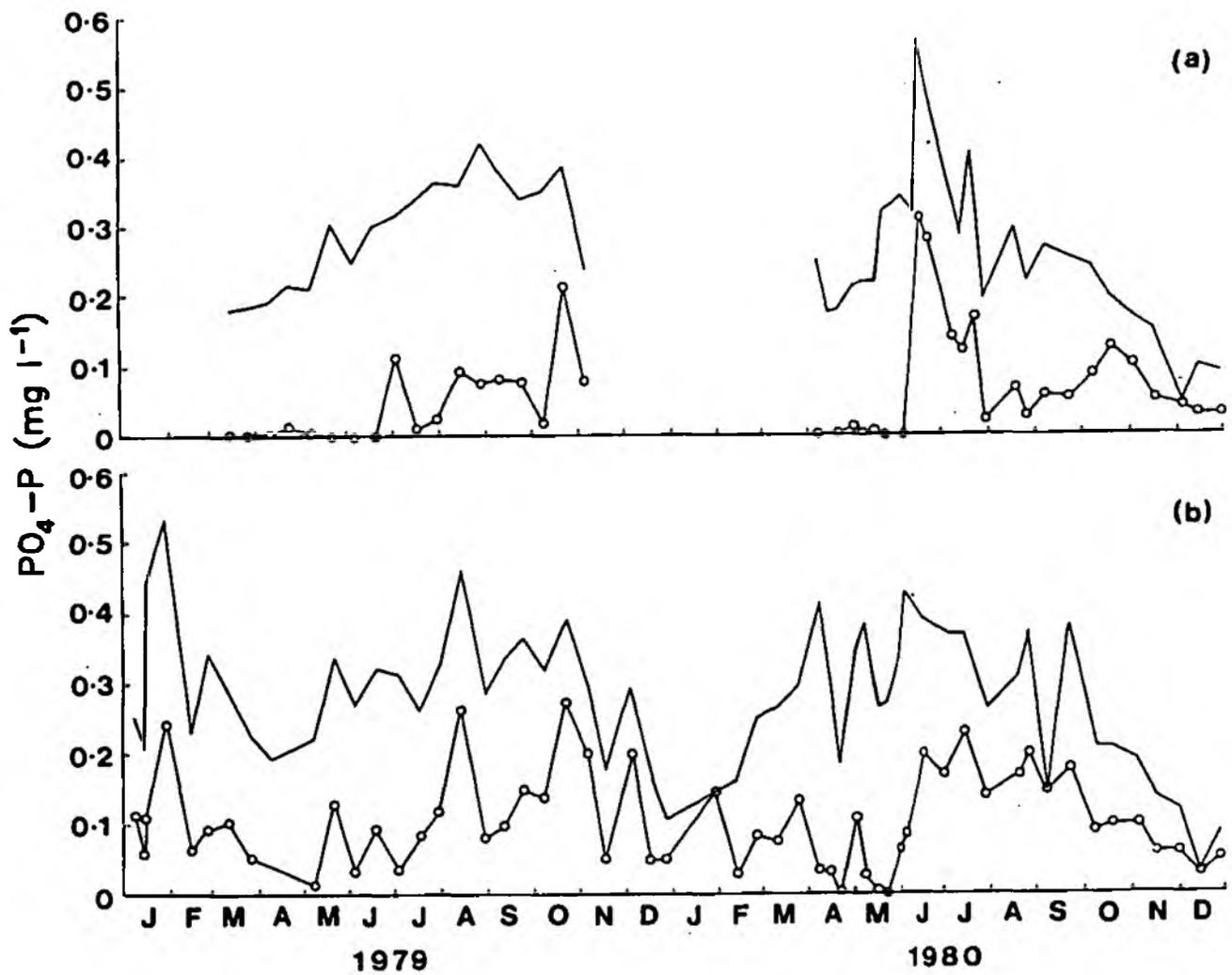


Figure 9. Total (—) and soluble reactive (—○—) phosphorus concentration in the surface water of (a) Barton Broad and (b) the inflow to Barton Broad during 1979-80.

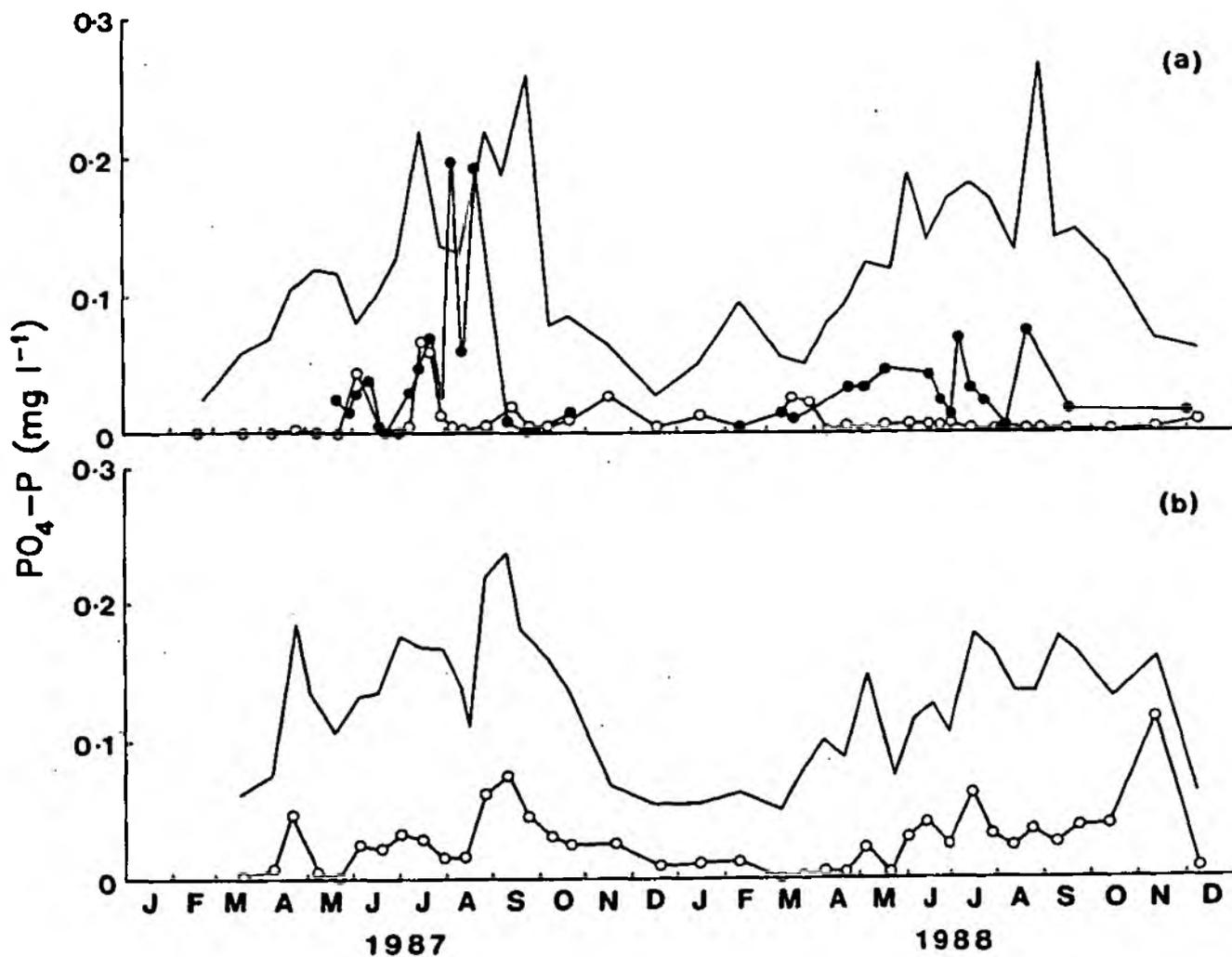


Figure 10. Total (—) and soluble reactive (—○—) phosphorus concentration in the surface water, and soluble reactive phosphorus concentration in the water just above the sediment (—●—) of (a) Barton Broad and (b) the inflow to Barton Broad during 1987-88.

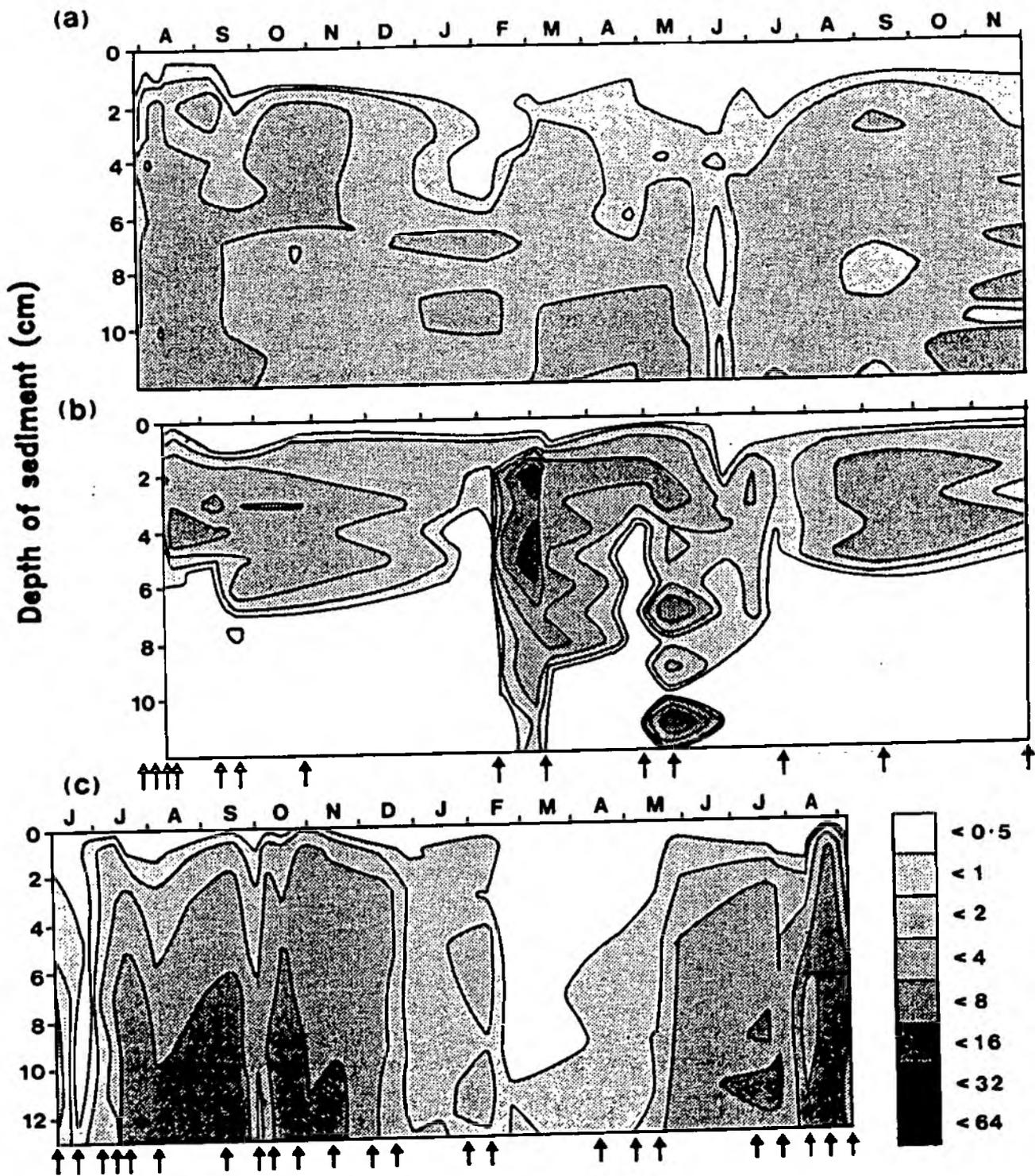


Figure 11. Temporal variation in concentration with sediment depth of (a) soluble phosphorus during 1987-88; (b) ferrous iron during 1987-88; and (c) soluble phosphorus during 1979-80 in sediment cores taken from Barton Broad. All concentrations expressed in  $\text{mg l}^{-1}$ . Arrows indicate sampling dates.

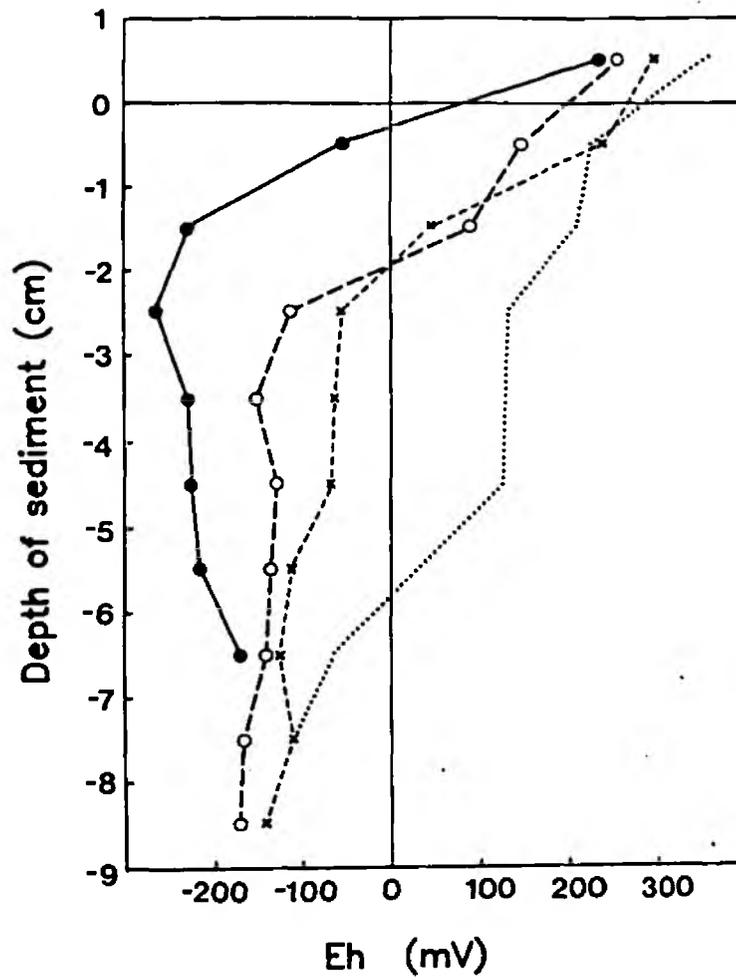


Figure 12. Depth profiles of redox potential (Eh,) in sediment cores taken from Barton Broad on 9/3/88 (.....), 3/5/88 (-x-), 20/6/88 (-o-) and 13/9/88 (-●-).

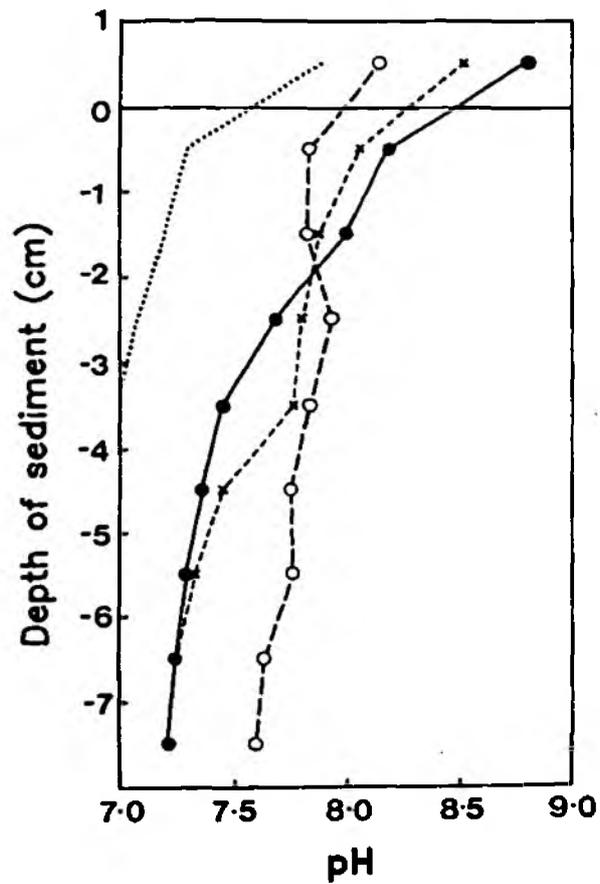


Figure 13. Depth profiles of pH in sediment cores taken from Barton Broad on 9/3/88 (.....), 3/5/88 (---x---), 20/6/88 (---o---) and 13/9/88 (---●---).

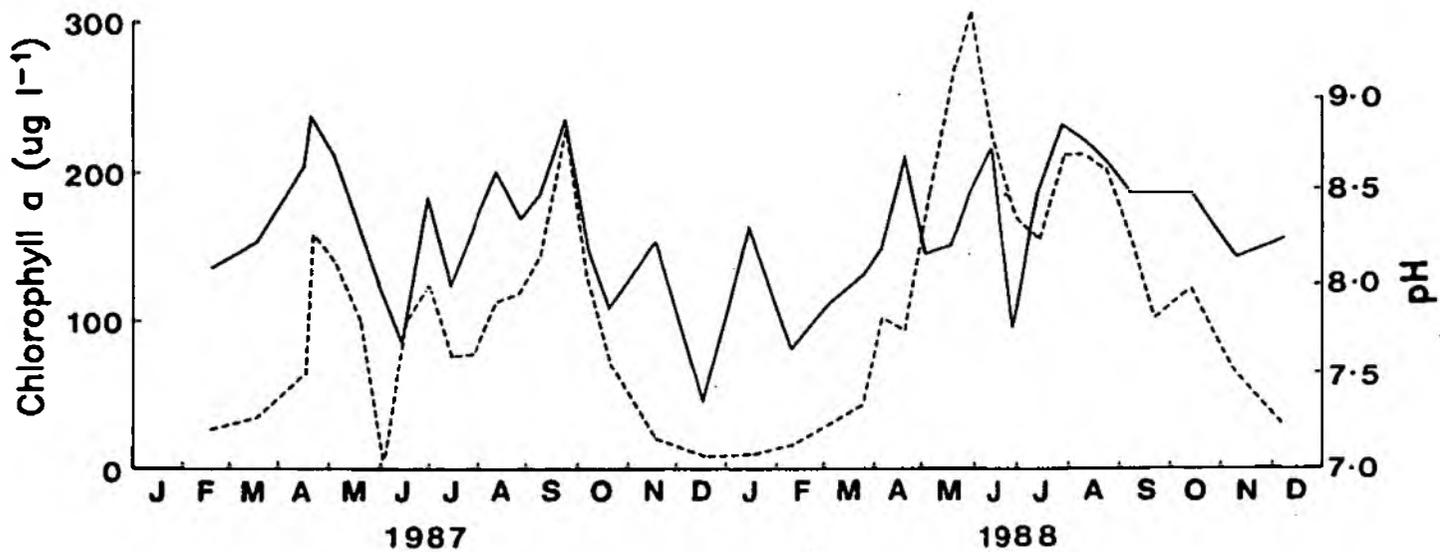


Figure 14. pH (—) and Chlorophyll a (----) concentration in the surface water of Barton Broad during 1987-88.

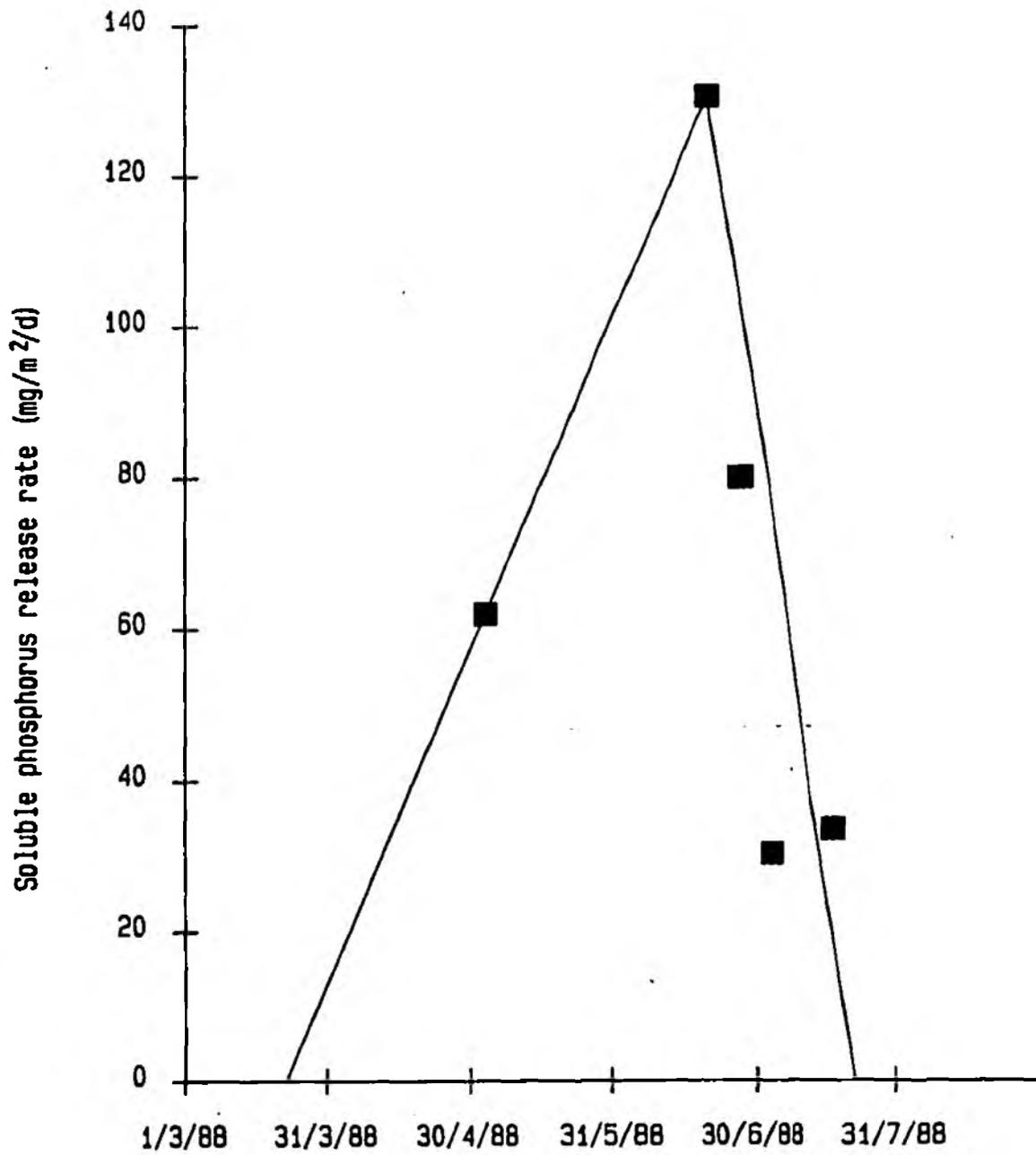


Figure 15. Experimentally determined phosphorus release rate from Barton Broad sediment during 1988.

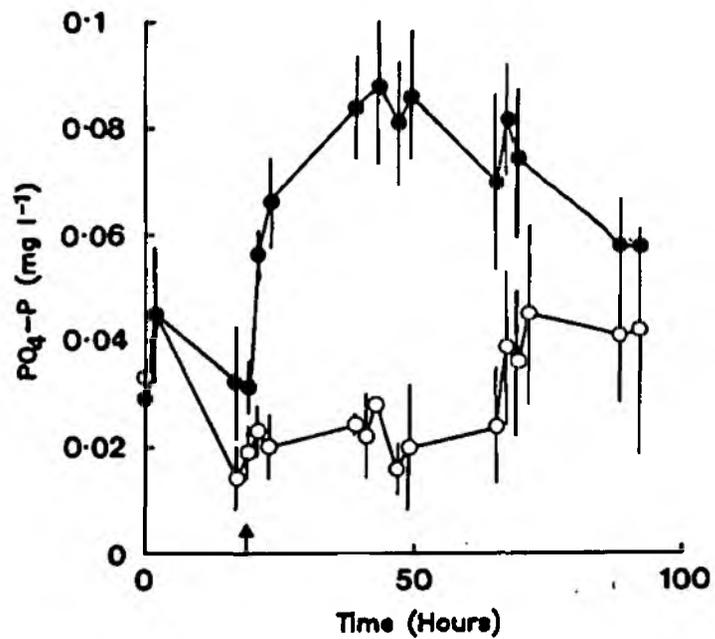


Figure 16. Release of soluble reactive phosphorus from Barton Broad sediment cores with (●) and without (○) the addition of 30 chironomid larvae and 30 tubificid worms to each core. The arrow denotes the addition of the animals. The error bars denote the standard deviation from the mean for the release from three cores.

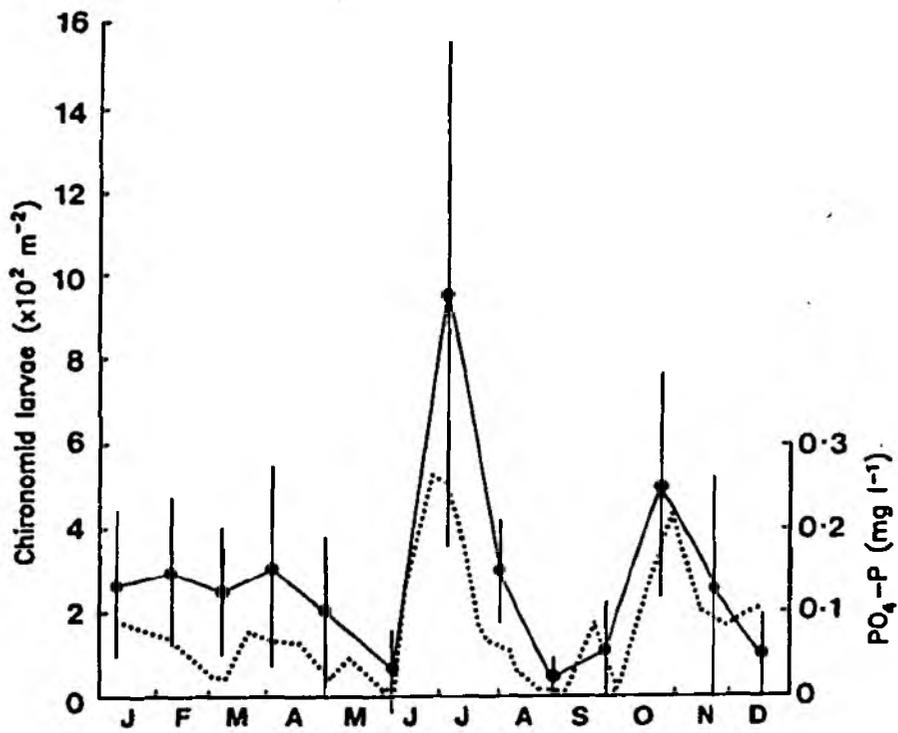


Figure 17. Concentration of 4th instar *Chironomus plumosus* larvae (—●—) in the sediment of Barton Broad in 1978 in comparison with the soluble reactive phosphorus concentration (.....) in the overlying water. Error bars denote the standard deviation from the mean larval density of 10 Eckmann grab samples.

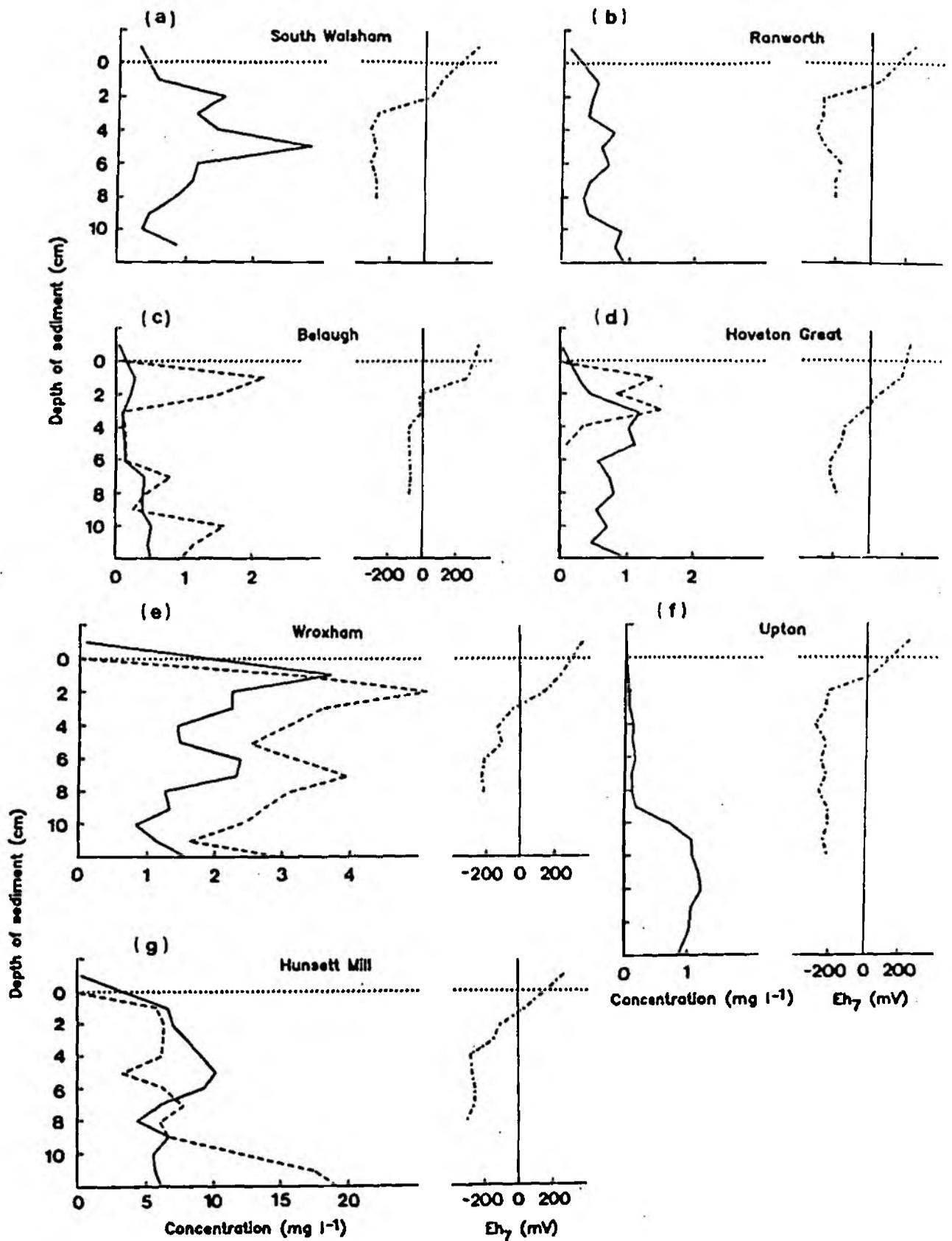


Figure 18. Profiles of soluble phosphorus (—) and ferrous iron (----) concentration and  $Eh_7$  (· · ·) from sediment cores from (a) South Walsham Broad, (b) Ranworth Broad, (c) Belough Broad, (d) Hoveton Great Broad, (e) Wroxham Broad, (f) Upton Broad and (g) Hunsett Mill on the River Ant upstream of Barton Broad.