CHESLEY LAKE 1991
WATER QUALITY ASSESSMENT AND MANAGEMENT IMPLICATIONS

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CHESLEY LAKE 1991 WATER QUALITY ASSESSMENT AND MANAGEMENT IMPLICATIONS

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PREFACE

Chesley Lake was one of 12 southwestern Ontario lakes investigated by the Ministry of Environment and Energy as part of the Water Resources Branch Inland Lakes Program (1987 - 1991). The program was designed to monitor a selection of hardwater inland lakes with, or with the potential for, nuisance blue green algae blooms and other water quality programs. Lakes were chosen in response to complaints regarding surface scums, algal blooms and seasonal hypolimnetic anoxia, received from organized public associations, MOE regions and/or Conservation Authorities Based on these surveys, several lakes were assessed as having a good potential to respond to experimental treatment programs.

Hypolimnetic aeration could be a successful remedial measure for Chesley Lake, if combined with shoreline enhancement and reduced external nutrient loadings. A lack of funding currently limits such remedial works however, unless private funds or other cost sharing agreements between government agencies and the private sector can be reached. This report was prepared to ensure that the water quality information is made available to those agencies and individuals expressing an interest in the findings.

ABSTRACT

Chesley Lake is an eutrophic hardwater lake and experiences severe hypolimnetic anoxia. Anoxic hypolimnetic conditions were first noted in mid-May. The hypolimnetic oxygen depletion occurred at a maximum mass rate of 184.4 kg/day (AHOD: 280 $mg/m^2/day$) in 1991. The hypolimnion, which extended from 8 to 18 m, was completely anoxic by the end of June. Bottom water nutrient and metals levels increased as a result.

Total phosphorus concentrations increased in both bottom and surface waters as the season progressed reaching levels as high as 0.025 mg/L in mid-September. Soluble reactive phosphorus trends were similar, but levels increased from approximately 3.0 to 9.0 µg/L during the August to September period. Marked seasonal increases were also observed (particularly in hypolimnetic waters) for ammonium, iron, manganese, hydrogen sulphide and turbidity.

Hypolimnetic aeration could be a successful management technique used to reduce sediment nutrient regeneration and improve lake water quality and fish habitat.

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INTRODUCTION

Chesley Lake (Amabel Township) is located on the Bruce Peninsula (Lat. 44°'30', Long. 81° 14') near the town of Owen Sound in southern Ontario. The lake has a surface area of 178.4 ha with a maximum depth of 18.0 m (Figure 1). The mean depth is 5.0 m. The lake is primarily spring fed (Almond 1987) without a designated inflow, but some overland inflow is sourced from the watershed during the spring runoff. Consequently, flushing is limited. Lake physical data are presented in Table 1. The outflow at the northwestern end of the lake (Stoney Creek) is surrounded at its mouth by a marsh and only occasional trickle flow from the lake surface into the creek was observed during the summer of 1991. Stream gauging and outflow measurements, however, were not conducted.

The lake is recognized as eutrophic/mesotrophic and is under considerable stress from over development; 289 cottages at a density of 39 cottages/km of shoreline (1.4 cottages/ha of lake surface area), one summer seasonal resort and trailer park are within 500 m of the shoreline. Approximately 81 percent of the shoreline has been developed (Almond 1987).

Fish populations have declined significantly since 1970 with greatly reduced numbers and individuals of smaller size. The sportfish community consists of several dominant species: walleye *(Stizostedion vitreum),* bass *(Micropterus* spp.), yellow perch *(Perca flavascens)* and Northern pike *(Esox lucius).* A variety of coarse and forage fish also inhabit the lake (Almond 1987).

Littoral zone degradation has been a major problem with significant reductions of macrophyte beds in developed areas. Substrate alteration by the addition of foreign substrates and construction of gabion walls for docks and supporting structures is also practiced by residents, and may be partly responsible for the fisheries decline (Almond 1987).

Public concern regarding water quality and a declining recreational fishery resulted in several water quality (Hawkins 1989) and fisheries (Almond 1987) assessments. This report describes the water quality, phytoplankton and zooplankton data gathered from Chesley Lake during 1991 and briefly describes a possible remedial measure, hypolimnetic aeration, previously mentioned as a potential management option for Chesley Lake by Hawkins (1989). Combined with increased public awareness of the implications of their nearshore activities and reductions of external nutrient sources, a high probability of reducing sediment nutrient regeneration and bottom water nutrient levels is likely and enhanced surface water quality and restored bottom water habitat are additional anticipated benefits.

METHODS

Water samples, biological samples and oxygen/temperature profiles were collected approximately bi-weekly between April 23 and October 16 during 1991. Samples were collected from one site (Figure 1), near the centre of the lake, above the lake's deepest point (18 m).

Water clarity was determined by the use of a Secchi disc. The Secchi disc is a 20 cm diameter steel plate suspended from a marked rope and is painted in alternating black and white quarters. The disc was lowered on the shaded side of the boat until it disappeared, then raised again until it reappeared. The average of these two depths was termed the Secchi disc depth.

Two narrow mouthed (2.5 cm) 1 Litre glass bottles were lowered and raised through the euphotic zone (twice the Secchi disc depth), collecting a composite sample of the water column. The rate of descent and ascent was timed so that the bottles were completely filled just as they reached the surface.

A 500 mL subsample was withdrawn into a separate bottle for chlorophyll analysis and stabilized with 2 mL of a 2% $MgCO_3$ suspension. A second 500 mL subsample for metals analyses was preserved with 2 mL of concentrated nitric acid to prevent metals precipitation. A third subsample was preserved with Lugol's iodine for phytoplankton analyses and a fourth sample for nutrient chemistry was unpreserved. Samples were kept cool and delivered to the Ontario Ministry of the Environment's laboratory the same day for analyses. Samples were refrigerated and analyzed within 48 hours of collection following standard procedures (MOE 1986). A second set of samples was drawn from 1 metre above the bottom of the lake (1 MOB), using a 6 Litre capacity PVC Van-Dorn bottle. 500 mL samples were withdrawn from the bottle for nutrient chemistry and metals analyses.

Phytoplankton analyses were conducted on a bi-weekly basis and were identified to the genus level according to Nicholls *et. al* (1977).

Oxygen and temperature measurements were taken at one metre intervals from the surface of the lake to the bottom, using a YSI 58 dissolved oxygen and temperature metre. The metre was calibrated against a standard hand held mercury thermometer for temperature accuracy and calibrated against water samples of known oxygen concentration determined by the micro-Azide modification of the Winkler technique. In addition, 60 mL Winkler samples from the lake surface and 1 MOB were taken to confirm the integrity of oxygen metre readings at these depths.

Zooplankton samples were collected using a modified, metered Clarke-Bumpus net with 80 gm mesh and a 15.5 cm mouth. The collection net was raised through a vertical column from 1 MOB to the surface with the net attached to collect the sample (A) and with the net removed (B) to determine net flow restriction. Efficiency was determined by the ratio A/B from the metre values of each haul. Zooplankton collections were analyzed according to Yan and Mackie (1987). Sample enumeration was conducted as described in Girard and Reid (1990).

Lake volumes were determined from areas digitized from the Chesley Lake map contours presented in Almond (1987) (Figure 1) and were digitized with a standard computer digitizing table.

Hypolimnetic oxygen depletion rates were calculated from the bi-weekly profiles and calculated as the difference in the sums of the total oxygen mass at each stratum between 8 and 18 m at two metre intervals. Mean oxygen concentrations at two metre intervals were utilized because of limiting matching information from the existing bathymetric chart.

RESULTS

Oxygen and Temperature

Isopleths and isotherms for 1991 are presented in Figure 2. Peak surface temperatures (24°C) were reached during the week of July 24, 1991. Strong westerly winds induced surface water mixing throughout the summer forming an epilimnion which oscillated between 5 and 7 m in depth. Water temperatures declined rapidly through the thermocline which was resident between 7 and 9 m. Bottom water temperatures through the ice-free season were approximately 10°C except during periods of turnover in the spring and fall.

Epilimnetic oxygen levels ranged as high as 11.1 mg/L during the spring and fall but were severely depleted in the hypolimnion with the onset of anoxic conditions at 1 MOB between the sample dates June 11 and June 25. The areal hypolimnetic oxygen depletion (AHOD) rate was calculated by regression from the first 5 sample dates and was determined to be 280 $mg/m^2/day$ (Figure 3). The maximum oxygen depletion rate was observed during the first 9 weeks of the summer, 184.4 kg oxygen/day (0.19 mg/L/day), but an exponential equation best described the oxygen depletion (Figure 3). The hypolimnion was standardized as extending from 8 m to 18 m based on the deepest thermocline penetration prior to Aug 1, 1991. Hypolimnetic oxygen values began to rebound in early September as cooling waters and increasing winds depressed the thermocline. Turnover was considered complete in early October.

Secchi Disc and Turbidity

Water clarity was seasonally consistent with an annual mean averaging 2.3 ± 0.221 m. Transparency ranged between 1.7 and 2.7 m (Figure 4). Euphotic zone turbidity ranged between 1 and 9 Formazin turbidity units (FTU) but no clear seasonal trend was observed. Turbidity at 1 MOB was unimodal increasing during the late season following the diatom decline, indicating that settling seston from the euphotic zone was the most likely source of the turbidity (Figure 5).

Chlorophyll a and Phytoplankton

Total chlorophyll a displayed two seasonal peaks, one in early spring and a second in late summer. The mean annual total chlorophyll a recorded was the second highest since 1974 (Hawkins 1989) at 4.0 ± 1.8 µg/L. Chlorophyll a values ranged between 3.0 and 4.0 µg/L through May and July and fell through August to their lowest levels of the summer (1.5 µg/L) before rising during fall peaks in September (Figure 6). Seasonal chlorophyll a levels and total nitrogen:total phosphorus (TN:TP) ratios (Figure 6) were highly correlated until July 24, 1991 ($r=0.95$). This association was not however significant during the August to October period. Linear regressions between chlorophyll (dependant variable) and either Secchi disc or mean annual total phosphorus true and log-transformed values were performed but were not found to be significant ($r<0.602$, $n=9$). As expected, seasonal chlorophyll values were correlated to total phytoplankton biovolume (Figure 7) with a significant relationship ($r = 0.520$, $P>0.05$).

Seasonal phytoplankton biovolumes and % community dominance are presented in Figure 8. The seasonal distribution was bimodal with spring and fall peaks surrounding a summer minimum. The total biovolume during the fall peak was the highest of the season (4.5 mm^3/L). This compared to an annual mean phytoplankton biovolume of 3.04 ± 1.8 mm^3/L. The limnetic phytoplankton genera identified in Chesley lake are presented in Table 3.

Bacillariophytes (diatoms) dominated the algal assemblage throughout the year. During mid-summer, however, the chlorophytes (green algae) and cyanophytes (blue-green algae) comprised as much as 20% of the total biovolume. The other taxa rarely exceeded 10% of the algal assemblage (Figure 8).

Diatoms were mainly represented by four genera: *Cyclotella, Fragillaria, Melosira* and *Synedra.* Early season peaks which exceeded 80% of the total algal biovolume through most of the first half of the summer season were dominated by *Cyclotella. Melosira* and *Fragilaria* grew in importance during the mid summer and comprised close to 50% of the diatom assemblage although diatom biovolume at this time only comprised 40% of the total phytoplankton. *Synedra* dominated the diatoms through September and October and comprised up to 80% of the diatom group and 90% of the total phytoplankton assemblage in October.

A significant mid-season shift among the dinophytes was observed when *Ceratium* replaced *Peridinium in* late July during a period of increasing dinophyte biovolumes. *Cryptomonas erosa* was the dominant cryptophyte. Several genera alternated dominance among the chrysophytes (Table 3), but *Chrysocromulina parva* (although not a true chrysophyte) was clearly the most important member of this group, present in all samples with a seasonal mean biovolume of 0.27 mm^3/L. *Chlamydomonas, Gloeocystis* and *Oocystis* were the dominant chlorophytes which constituted a regular but small percentage (<10%) of the phytoplankton.

Cyanophyte biovolumes peaked in late summer but did not exceed 1.0 mm^3/L. Their increase coincided with lower N:P ratios and decreased nitrate and increased phosphorus levels. Increasing blue-green levels as a percentage of the community followed the midsummer diatom decline (Figure 8). Limited formation of surface blooms ("scum") was observed during the late season. The late summer increase was dominated by *Anabaena.* Other dominants included *Microcystis, Aphanothece* and Lyngbya.

Phosphorus

Total phosphorus (TP) concentrations at 1 MOB and in the euphotic zone exhibited spring and fall peaks (Figure 9). The trend was better developed in 1 MOB samples where nutrient uptake by phytoplankton was limited and phosphorus regeneration from bottom sediments was likely. Seasonal 1 MOB and euphotic zone TP concentrations were correlated ($r= 0.543$, $a=0.10$) and 1 MOB total phosphorus levels usually exceeded surface water values. Mean annual euphotic zone total phosphorus (0.014 ± 0.005 mg/L) was within surface water quality guidelines (MOE 1987).

Seasonal euphotic zone soluble reactive phosphorus (SRP) concentrations showed little fluctuation with an annual mean of <0.001 mg/L. Seasonal 1 MOB levels, however, rose sharply from the end of July to just prior to turnover when concentrations increased from 0.001 mg/L to 0.009 mg/L (Figure 10).

Nitrogen

Total nitrate levels (NO_3 + NO_2) at both the surface and at 1 MOB fell from spring peaks as the season progressed (Figure 11). By late May, nitrate concentrations were reduced to values at or near the detection limit (<0.005 mg/L). Nitrite (NO_2) concentrations were generally below the detection limit (<0.0005 mg/L). 1 MOB concentrations generally exceeded euphotic zone values. Euphotic zone Kjeldahl nitrogen (TKN) displayed little seasonal fluctuation (Figure 12) with an annual mean of 0.411 ± 0.078 mg/L. 1 MOB concentrations increased sharply from the end of June until turnover probably as a result of accumulating organic matter. Decreasing nitrate levels during the same time period were strong indicators that anaerobic nitrate reduction may have been significant and may have also been responsible for increasing TKN levels at 1 MOB.

Euphotic zone ammonium levels were at or near the detection limit with an annual mean of 0.004 mg/L. Samples from 1 MOB showed marked seasonal increases peaking at just below 0.8 mg/L in September (Figure 13). Microbially-mediated anaerobic decomposition of sediment and settling organic matter was the likely source of the increase.

Alkalinity, Calcium, DIC and Silica

Mean annual euphotic zone alkalinity, calcium and DIC concentrations are presented in Table 2 along with values for other parameters. Seasonal trends were not marked (Figure 14), although a small mid-summer decline in these parameters was observed, probably related to increased summer photosynthetic rates. Silica levels (Figure 14) increased in the euphotic zone as the summer progressed and reflected the declining importance of the Bacillariophytes during the mid to late summer.

Spring levels at or near the detection limit (<0.005 mg/L) suggest that silica may have been a limiting nutrient at this time which resulted in the mid-summer shift to other algal groups.

Iron, Magnesium, Manganese and Conductivity

Magnesium concentrations were similar in the euphotic zone and at 1 MOB averaging 14 mg/L and displayed little seasonal fluctuation. Euphotic zone iron and manganese levels (Figure 15), however, responded to increasing seasonal 1 MOB levels (Figure 16) with late season concentration peaks. During the same period of anoxic hypolimnetic conditions, iron and manganese levels at 1 MOB began to rise immediately after stratification in mid-May and peaked in September at approximately 1.5 mg/L (Figure 16).

Conductance measurements varied seasonally (Figure 17). Euphotic zone conductivity decreased as the summer progressed and was likely related to the precipitation of ions such as Ca^{2+} to sestonic matter and their settling from the euphotic zone to the hypolimnion. Observed seasonal increases in hypolimnetic conductance were highly correlated to increasing iron and manganese levels (r = 0.94) caused by anoxic bottom water conditions. These ionic patterns in combination with ionic contributions from settling epilimnetic sexton rationalize the conductivity increase.

Hydrogen Sulphide

Hydrogen sulphide levels at 1 MOB first appeared in mid-June following several weeks of anoxia at the sediment-water interface. Between June 11 and Sept 17 concentrations increased dramatically with bottom levels rising by approximately 95 µg/L /day (Figure 18). Just prior to turnover hydrogen sulphide levels peaked at 760 µg/L.

Zooplankton

Zooplankton biomass was bimodal with spring and fall peaks (Figure 18) ranging between 7.4 and 233 mg/m^3. Biomass was less than 10.0 mg/m^3 throughout July and August. The zooplankton community was dominated by cyclopoid copepods during the early season and replaced by daphnid cladoceran in autumn (Figure 19). Table 4 presents the common zooplankton of Chesley Lake in 1991.

Daphnia retrocurva was the dominant species during the spring and fall peaks, contributing over 95% of the daphnid species. The largest peak occurred during September and was 70% *D. retrocurva. Bosmina longirostris* was the dominant non-daphnid cladoceran and ranged between 5 and 20% of the population between April and August. The calanoid copepods were dominated throughout the season by calanoid copepodids and *Diaptomus oregonensis.* They constituted approximately 20% of the zooplankton biomass until October when they dominated the zooplankton by contributing over 80% of the total. Cyclopoid copepodids and *Mesocyclops edax* were the most important species of the cyclopoid grouping.

DISCUSSION

Severe oxygen depletion was apparent in early June. An AHOD rate of 280 $mg/m^2/day$ describes the trend. Rich organic sediments with a measured loss on ignition (LOI) of 46% (460 mg/g dry weight) and a total phosphorus concentration of 750 µg/g (Hawkins 1989) indicated that sediment oxygen demand was expected to be high and were a potential source of phosphorus via internal nutrient loading. Unfortunately, only limited data from earlier studies exists. As cited by Almond (1987), they describe mid-summer oxygen conditions at 1 MOB between 1.6 mg/L in 1948 and 23 mg/L in 1963. The first appearance of an anoxic hypolimnion in Chesley Lake was recorded in 1975, so it appears that the anoxic hypolimnion "phenomenon" followed the increase in cottage development during the late 1950s and 1960s around Chesley Lake.

It should be stressed, however, that sediment oxygen demand is only a partial component of the total hypolimnetic oxygen demand. For example, Cornett and Rigler (1987) described the relative importance of water column processes which could impart up to 60% of the total oxygen demand and the low hypolimnetic oxygen levels of 1948 and 1963 indicate that there was already significant hypolimnetic oxygen demand prior to the shoreline development years.

The formation of a stable thermocline limited infusion of oxygen down to the hypolimnion but it also limited vertical nutrient transport. Regeneration of nutrients from anoxic profundal sediments is well documented (Syers 1973, Carignan and Lean 1991) and was likely a primary source of the dramatic increases in bottom water levels and euphotic zone increases which followed periods of strong westerly winds when the thermocline was depressed and metalimnetic erosion occurred.

Long term water quality trends were described by Hawkins (1989) and show significantly higher mean annual chlorophyll a levels (5.4 µg/L) in 1988 and from this study (4.01 µg/L), compared to levels of between 1.0 and 2.0 pg/L in the early 1970s. The mean annual chlorophyll a levels (4.01 µg/L) measured during this study compare favourably with those reported by Hawkins for 1988. Fluctuations in phytoplankton community dominance were also tabulated and showed a higher contribution to the phytoplankton by the Bacillariophyte (diatom) group which composed 36 and 45% of the algal assemblage in 1981 and 1988 compared to between 15 and 20% of the total between 1977 and 1980.

These values probably understated the importance of the diatom group as the analyses used were summer recombinations (pooled samples) on sample sets which often excluded the critical spring and fall periods when diatoms are dominant. The phytoplankton data presented here show that the diatoms comprised up to 80% of the algal assemblage with reduced importance during the mid-summer when they comprised approximately 40 to 60% of the algal community. Increasing seasonal chlorophyll a levels combined with the 1991 seasonal mean algal biovolume of 3.04 mm^3/L (the highest recorded) suggests that Chesley Lake trophic conditions are continuing to degrade. Because of the inconsistent sampling approach in the earlier studies, a conclusive trend cannot be established based on chlorophyll values alone.

Mean annual phosphorus levels at 1 MOB were variable from year to year and those reported for 1991 were significantly lower than in previous years. Between 1977 and 1988, mean annual 1 MOB TP concentrations ranged between 0.033 and 0.062 mg/L. Mean annual 1 MOB SRP levels ranged between 0.006 and 0.025 mg/L (Hawkins 1989). In 1988 for example, 1 MOB TP reached 0.146 mg/L in early September, compared to a 1991 1 MOB mean of 0.018 mg/L, and peak of 0.026 mg/L. The 1991 1 MOB TP levels may have been reduced by high iron levels and co-precipitation with calcium during a lake-whiting effect which was observed in late August but may have been occurring throughout the late summer. Under these conditions, decreases in TP occur and as the particles subsequently sink TP is removed from the water column. High 1 MOB turbidity and silica levels suggest that sedimentation was high in 1992.

Two typical algal community responses in relation to changing environmental variables were observed on a seasonal basis; the inverse relationship between euphotic zone silica levels and diatom numbers, with silica levels increasing during the mid-summer diatom decline (Hecky *et al.* 1986), and the late season increase in blue-green algae as a result of falling TN:TP ratios (Smith 1986) and increasing phosphorus levels during the fall. Peak blue-green levels were observed on Sept 17 when annual TN:TP ratios were at their lowest and a peak in phosphorus values was observed. Nuisance surface blue-green blooms were not observed in 1991, probably because of a combination of factors (wind, temperature, seasonal variation), but TN:TP ratios remained above the critical level of 20:1.

Zooplankton numbers were weakly correlated to phytoplankton biovolumes displaying early summer and fall biomass peaks. The sudden mid-summer decline in numbers has been suggested as an artifact of the absence of a hypolimnetic refuge (Tessier *et al.* 1991), and anoxic hypolimnia have been observed to result in a clustering of zooplankton and predators, in the metalimnion where the zooplankton would be most susceptible to predation from young of the year fish (Gemza unpubl. data). Decreasing zooplankton lengths and weights during the mid-summer period of declining zooplankton biomass supports this observation. Of note was the appearance of

Leptodora kindtii, a crustacean predator, during the mid-summer period.

Peak spring euphotic zone nitrate levels were typical of southern Ontario lakes and a result of increased runoff and groundwater recharge (Gemza 1991, Vandermeulen and Gemza 1991). Values fell as the season progressed due to increased algal assimilation and reduced inputs, and at 1 MOB primarily as a result of nitrate reduction.

Late season increases in 1 MOB turbidity values were most likely a result of several determinants. The sedimentation from epilimnetic seston would certainly have been important but with anoxic hypolimnetic conditions present, turbidity in bottom waters would also have been increased by the precipitation of ferric iron in colloidal and finely suspended forms.

The prolonged anoxia below 6 m clearly effected sediment nutrient release as indicated by dramatic seasonal increases in the typical indicator variables: TP, NH_3 SRP, H_2S, iron and manganese as well as conductivity.

Hypolimnetic Aeration as a Management Option for Chesley Lake

Hypolimnetic aeration is a technique used to oxygenate lake and reservoir hypolimnia without lake destratification. Although it is beyond the scope of this report to outline in detail the techniques, results and recent developments in hypolimnetic aeration as applied to Ontario lakes, the references quoted will provide a good background to the reader unfamiliar with this technology.

Reviews of the wide variety of hypolimnetic aeration techniques either successful or experimental are well documented (McQueen and Lean 1986, Fast and Lorenzen 1976, Speece 1973). The intent as applied to Chesley Lake would be to maintain bottom water oxygen levels through the late-May to September period, primarily for nutrient control when anoxic conditions lead to increased sediment nutrient release with the additional benefits of a restoration of cold water fisheries habitat.

Results from other studies show that the aeration of bottom waters has affected control of nutrient levels (Ashley 1983, Gemza unpubl. data) even at hypolimnetic oxygen concentrations maintained at less than 2.0 mg/L, but the impact on euphotic zone phytoplankton communities is less well understood.

While some hypolimnetic aeration experiments utilizing compressed atmospheric air have been successful in reducing nutrient levels and enhancing oxygen concentrations, they required surface venting to remove accumulated nitrogen gas from the treated water. This methodology could result in hypolimnetic warming and offers relatively low oxygen transfer efficiencies. Utilizing a system which introduces pure oxygen (95% min. by weight) offers increased oxygen transfer efficiency and lower capital and operating costs.

In basins exceeding 30 m in depth, oxygen may be introduced to the hypolimnion by simply bubbling the gas into the bottom of the hypolimnion through fine bubble diffusers. Oxygen from the rising gas bubbles diffuse into the water so that by the time the plume reaches the thermocline, bubble size and rise energy are negligible, the bubbles uncouple from the entrained water and only a few remnant bubbles reach the surface without disrupting the stratification (Speece 1973). In shallower reservoirs, because of the reduced pressures involved, additional structures (Fast and Lorenzen 1976, Ashley 1983) are required to confine the energy of the bubble network while circulating and oxygenating the bottom waters.

The advent of small scale on-site oxygen generation facilities has made the use of pure oxygen in hypolimnetic aeration systems cost effective from a practical long term lake restoration standpoint (Gemza in prep). While the use of stored liquid oxygen has been shown to be effective (Fast *et al.* 1975, Speece 1973, Gemza unpubl. data), the ongoing annual cost for treatment of Chesley Lake would have been between $35,000 to $55,000 (Can. 1992) in oxygen costs alone. Oxygen generation on-site requires only a small capital cost and the electrical cost of running air compressors is approximately 10% the price of purchased oxygen. The technology is currently under development by the Ministry of the Environment (Aquatic Plant Unit) with an advanced design with potential application to Chesley Lake available in the near future. It is feasible that a cost effective application at under $60,000 in capital start up costs may be ready in the near future.

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Table 1: Chesley Lake hypsometry and physical data.

Depth (m)	Interval Area (ha)	Contour Interval (m)	Interval Volume ($x10^4$ m^3)	Sediment Area (ha)
0	178.4			
2	135.5	0-2	313	43
4	92.4	2-4	227	43
6	55.3	4-6	146	37
8	31.4	6-8	86	24
10	18.2	8-10	49	13
12	11.0	10-12	29	7
14	6.0	12-14	17	5
16	3.1	14-16	9	3
18	1.0	16-18	4	2.5

Total volume:	877.9×10^4 m^3
Hypolimnetic Volume:	106.9×10^4 m^3
Max. depth:	18.0 m
Mean depth:	5.0 m
Shoreline:	7.4 km

ha - Hectares

Table 2: Chesley Lake mean annual ice-free euphotic zone concentrations for the year 1991.

	Units	Mean	Standard Dev.
Secchi Disc	(m)	2.321	0.221
Total Chlorophyll a	(µg/L)	4.008	1.777
Phytoplankton Biovolume	(mm^3 /L)	3.040	1.790
% Blue-greens		10.461	9.685
Zooplankton Biomass	(mg/m^3)	72.413	60.590
Total Phosphorus	(mg/L)	0.014	0.005
Soluble Reactive Phosphorus	(µg/L)	< 1.0	1.0
Total Nitrates	(mg/L)	0.013	0.031
Ammonium	(mg/L)	0.004	0.007
Total Kjeldahl	(mg/L)	0.411	0.078
TN:TP Ratio		30.041	12.287
pH		8.380	0.061
Calcium	(mg/L)	36.517	2.548
Hardness	(mg/L)	153.083	6.570
Chloride	(mg/L)	5.917	0.362
Conductivity	(µmhos/cm)	279.333	8.475
Turbidity	(FTU)	2.009	1.140
DIC	(mg/L)	30.883	1.556
DOC	(mg/L)	4.333	2.052
Iron	(mg/L)	0.010	0.014
Silica	(mg/L)	0.718	0.548
Sulphates	(mg/L)	9.345	0.675

Table 3: List of the common phytoplankton genera of Chesley Lake.

Cyanophytes	Chlorophytes
Anabaena	*Botryococcus*
Aphanothece	*Carteria*
Chroococcus	*Chlamydomonas*
Coelosphaerium	*Chodatella*
Lyngbya	*Closterium*
Merismopedia	*Coelastrum*
Microcystis	*Cosmarium*
Oscillatoria	*Crucigenia*
	Euastrum
Dinophytes	*Gloeocystis*
	Golenkinia
Ceratium	*Kirchneriella*
Peridinium	*Micractinium*
	Monoraphidium
Cryptophytes	*Oocystis*
	Pediastrum
Cryptomonas	*Planctonema*
Katablepharis	*Quadricula*
Rhodomonas	*Scenedesmus*
	Staurastrum
Chrysophytes	*Tetraedron*
Bitrichia	**Bacillariophytes**
Chromulina	
Chrysochromulina	*Asterionella*
Chrysolykos	*Cyclotella*
Dinobryon	*Eunotia*
Mallomonas	*Fragilaria*
Salpingoeca	*Melosira*
Spiniferomonas	*Navicula*
Uroglena	*Nitzschia*
	Rhizosolenia
	Synedra
	Tabellaria

Table 4: List of the common zooplankton species of Chesley Lake 1991.

Cladocerans

Acroperus harpae
Alona sp.
Bosmina longirostris
Ceriodaphia sp.
Chydorus sphaericus
Daphnia galeota mendotae
Daphnia parva
Daphnia retrocurva
Leptodora kindtii
Macrothrix laticornis
Diaphanosoma birgei

Calanoid copepods

Calanoid copepodid
Calanoid nauplii
Diaptomus minitus
Diaptomus oregonensis
Epischura lacustris
Epischura lacustris copepodid

Cyclopoid copepods

Cyclopoid copepodid
Cyclopoid nauplii
Cyclops bicuspidatus thomasi
Cyclops vernalis
Mesocyclops edax

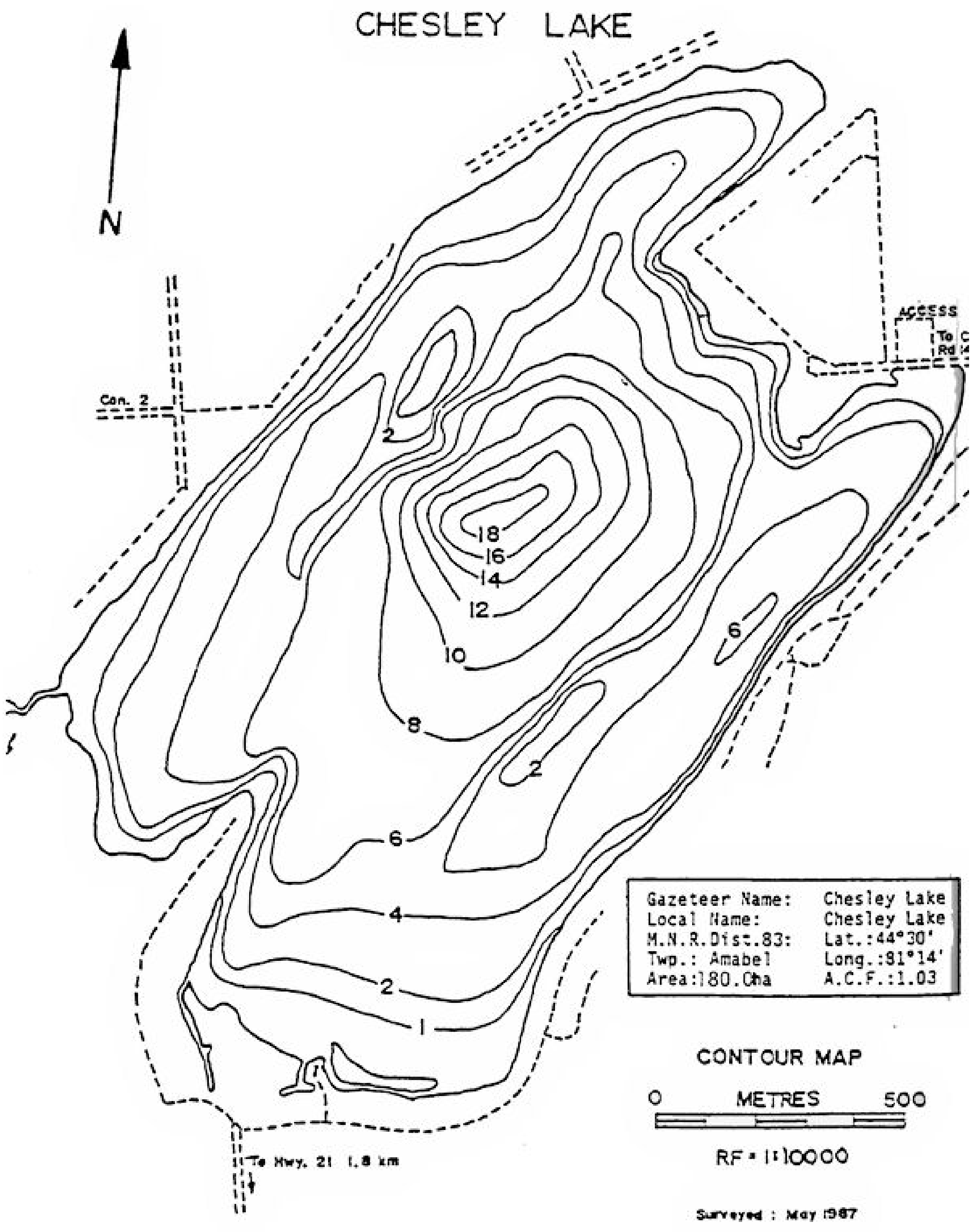

Figure 1: Bathymetric chart of Chesley Lake.

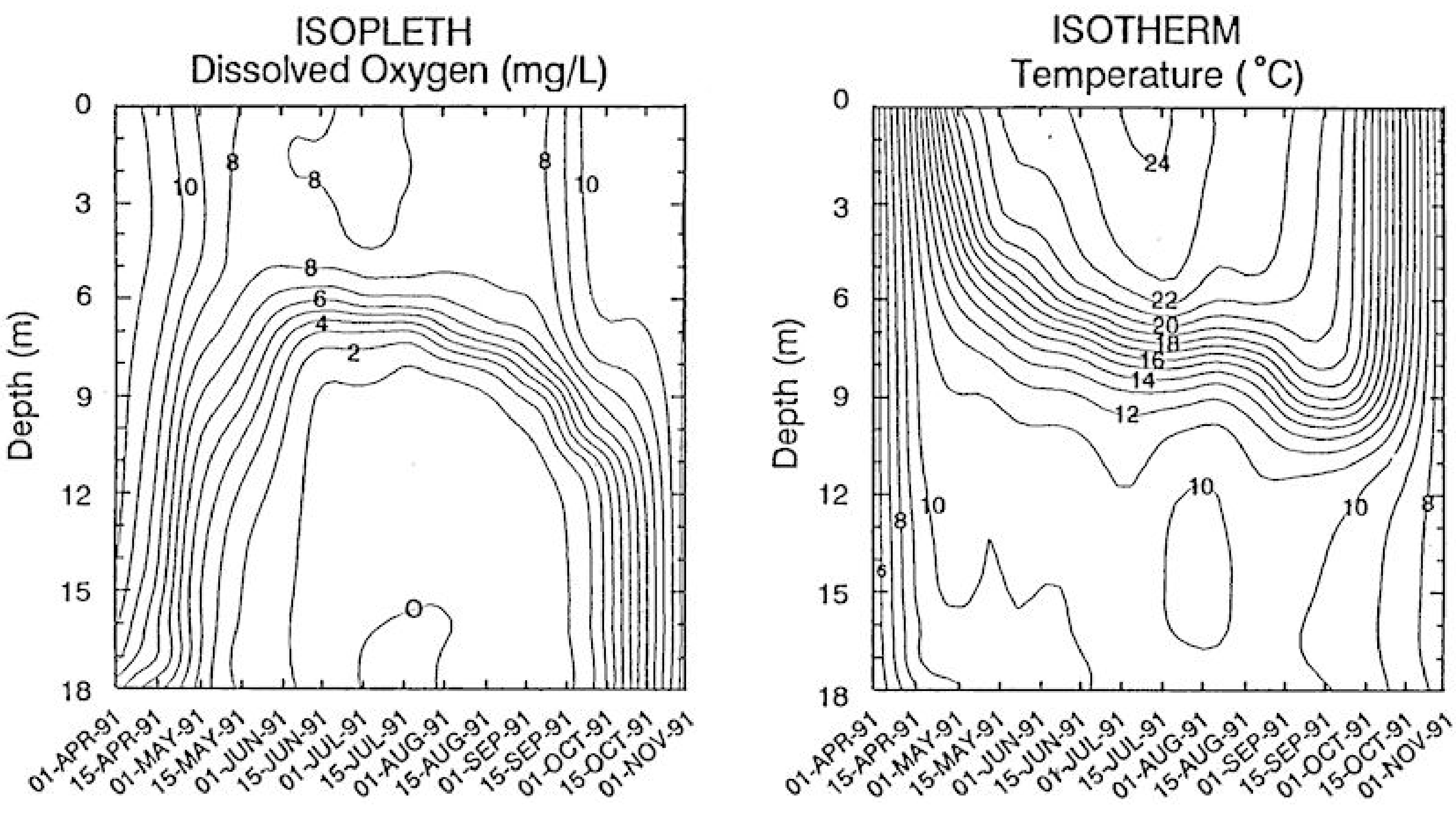

Figure 2: Isotherms and isopleths for Chesley Lake, 1991.

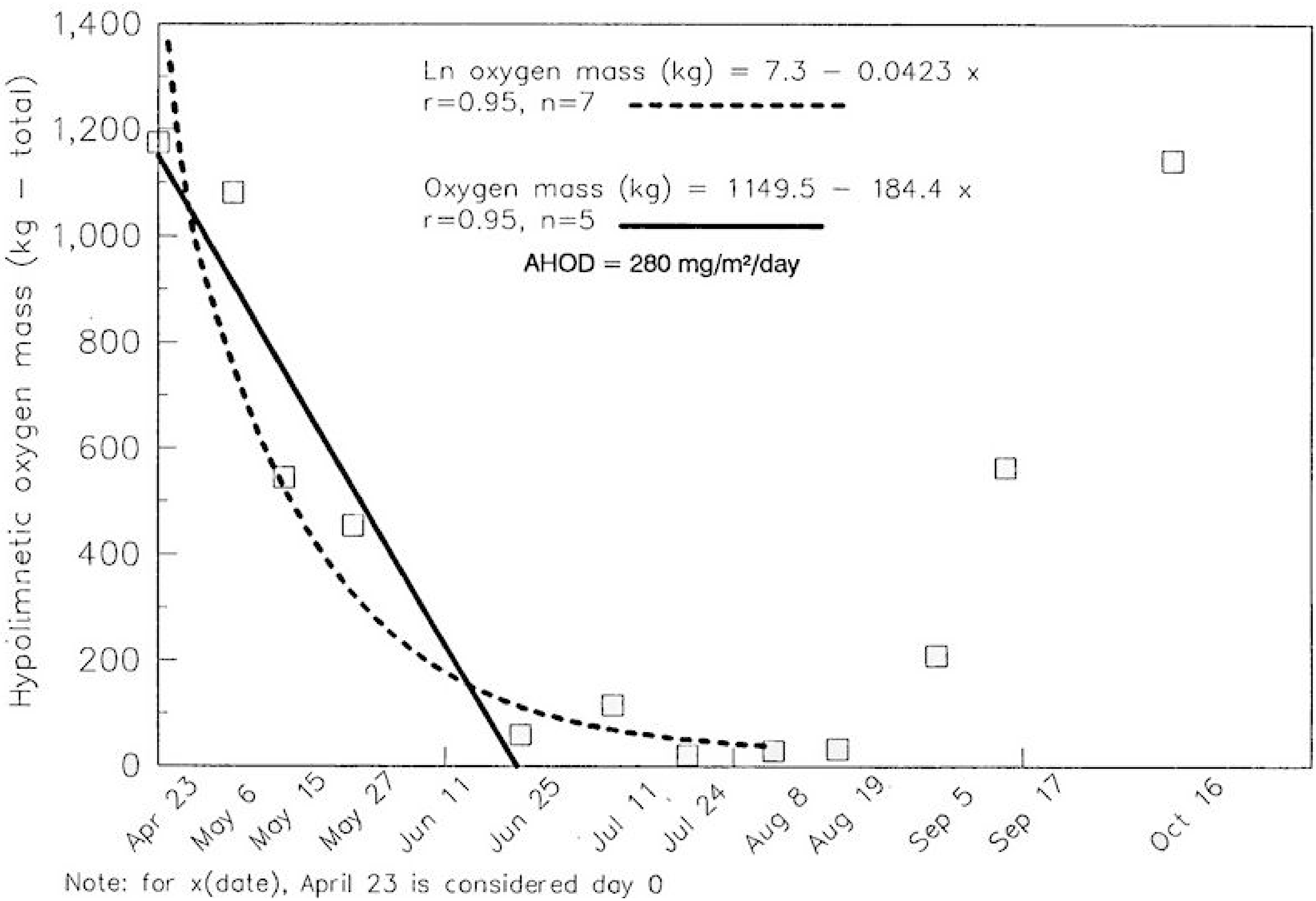

Figure 3: 1991 Chesley Lake hypolimnetic oxygen depletion rate curves during the period of maximum oxygen depletion.

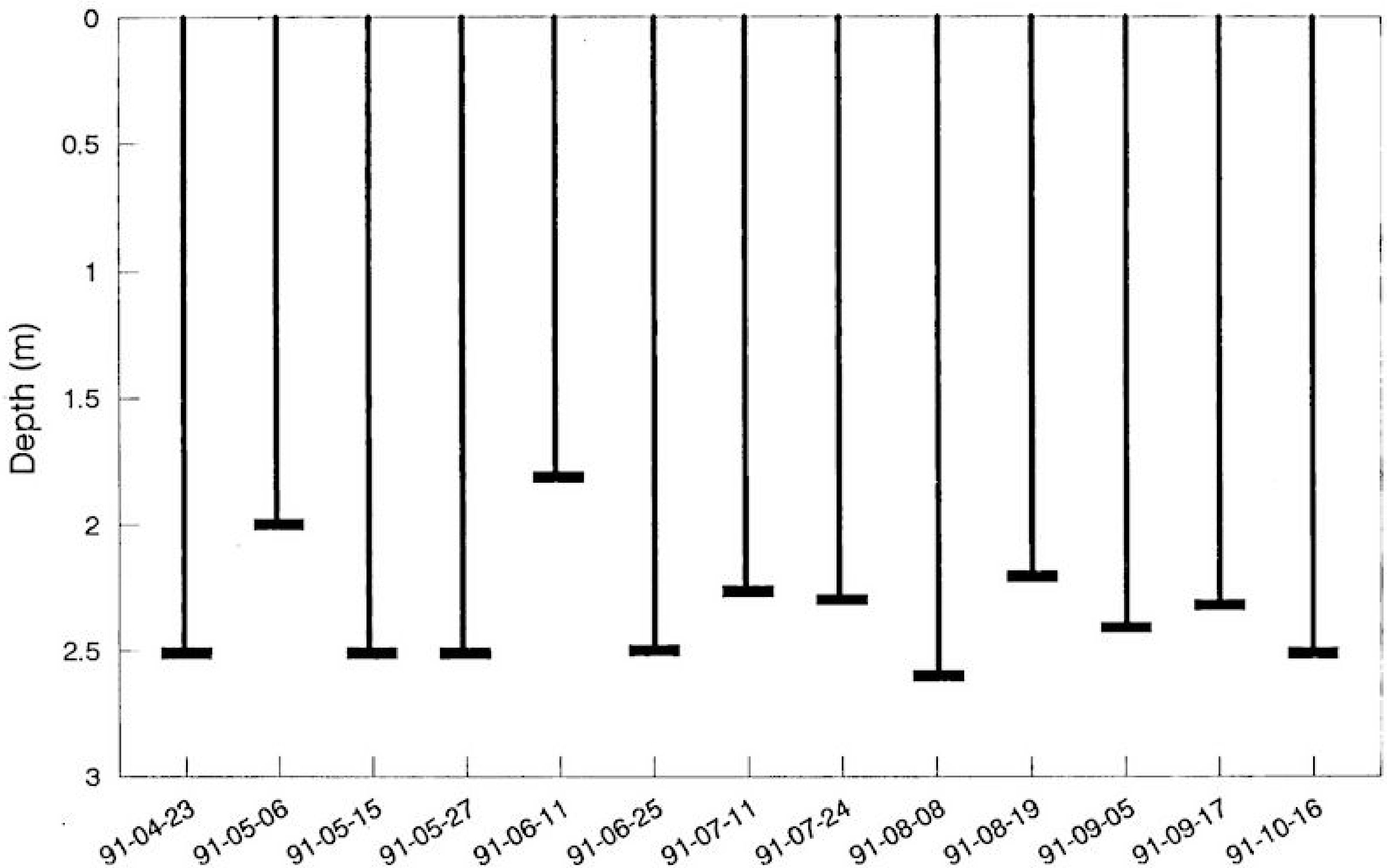

Figure 4: Seasonal secchi disc visibility for Chesley Lake during 1991.

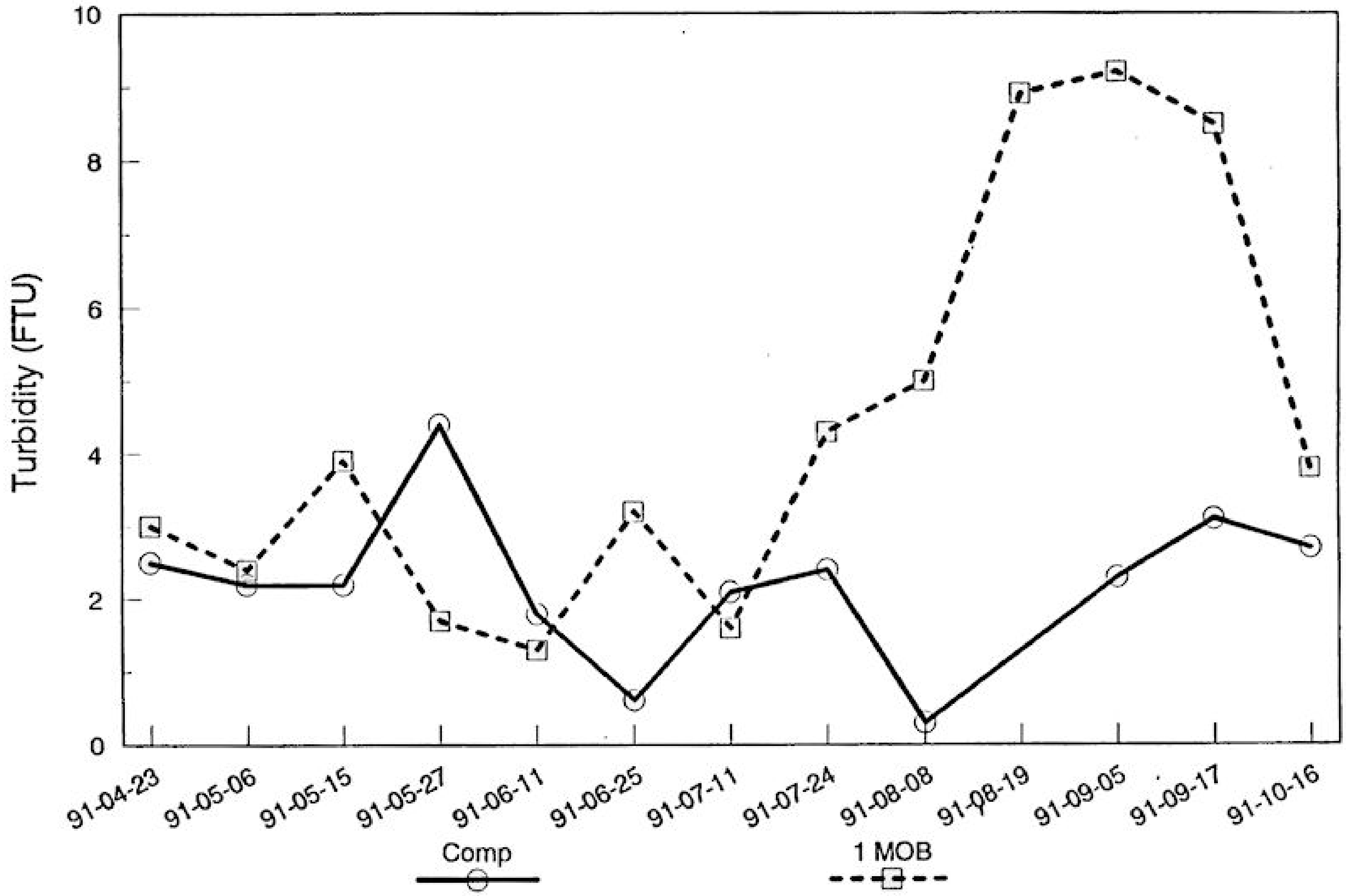

Figure 5: Seasonal euphotic zone (comp) and 1 MOB turbidity readings.

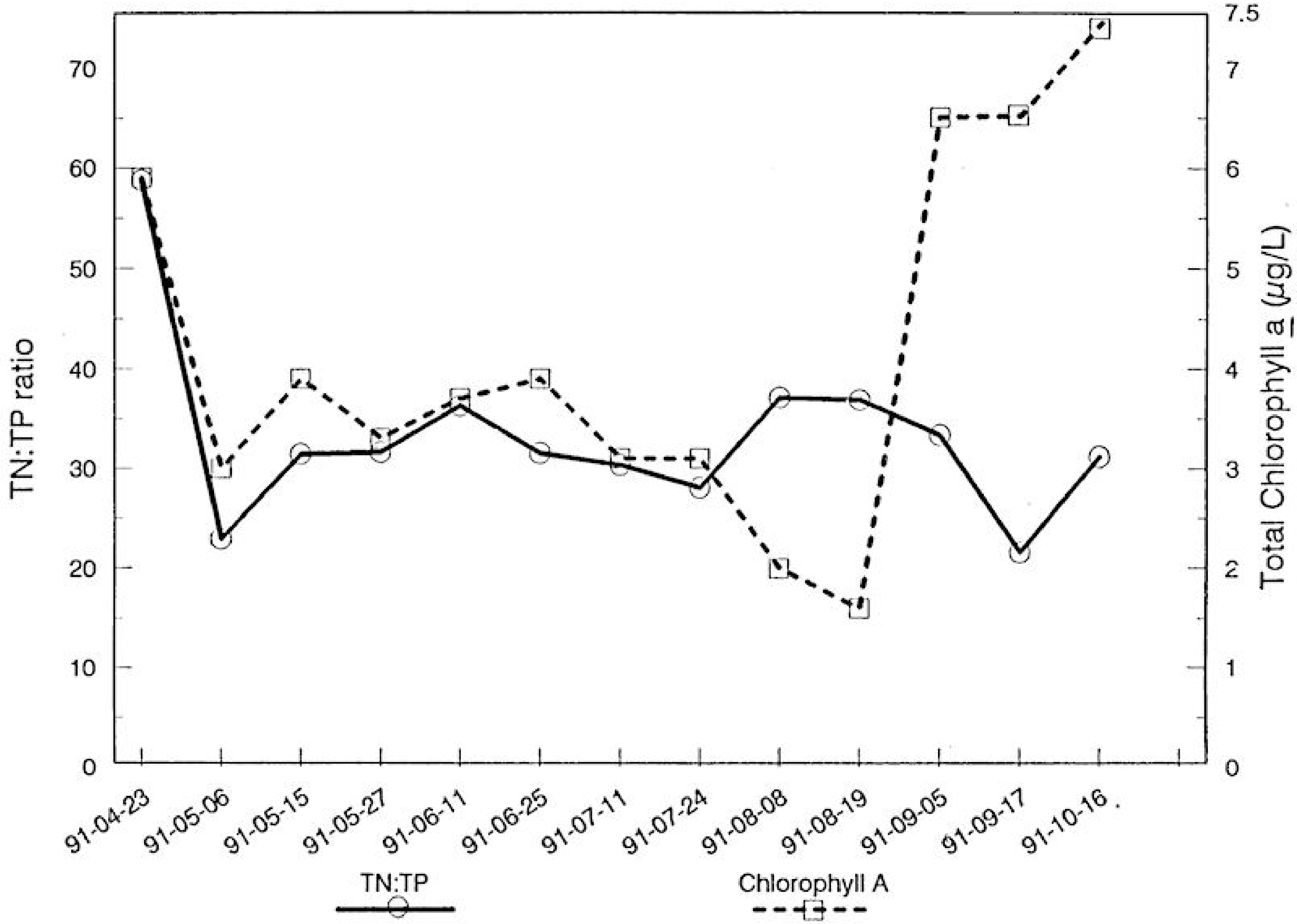

Figure 6: 1991 seasonal euphotic zone total chlorophyll a and total nitrogen : total phosphorus ratios.

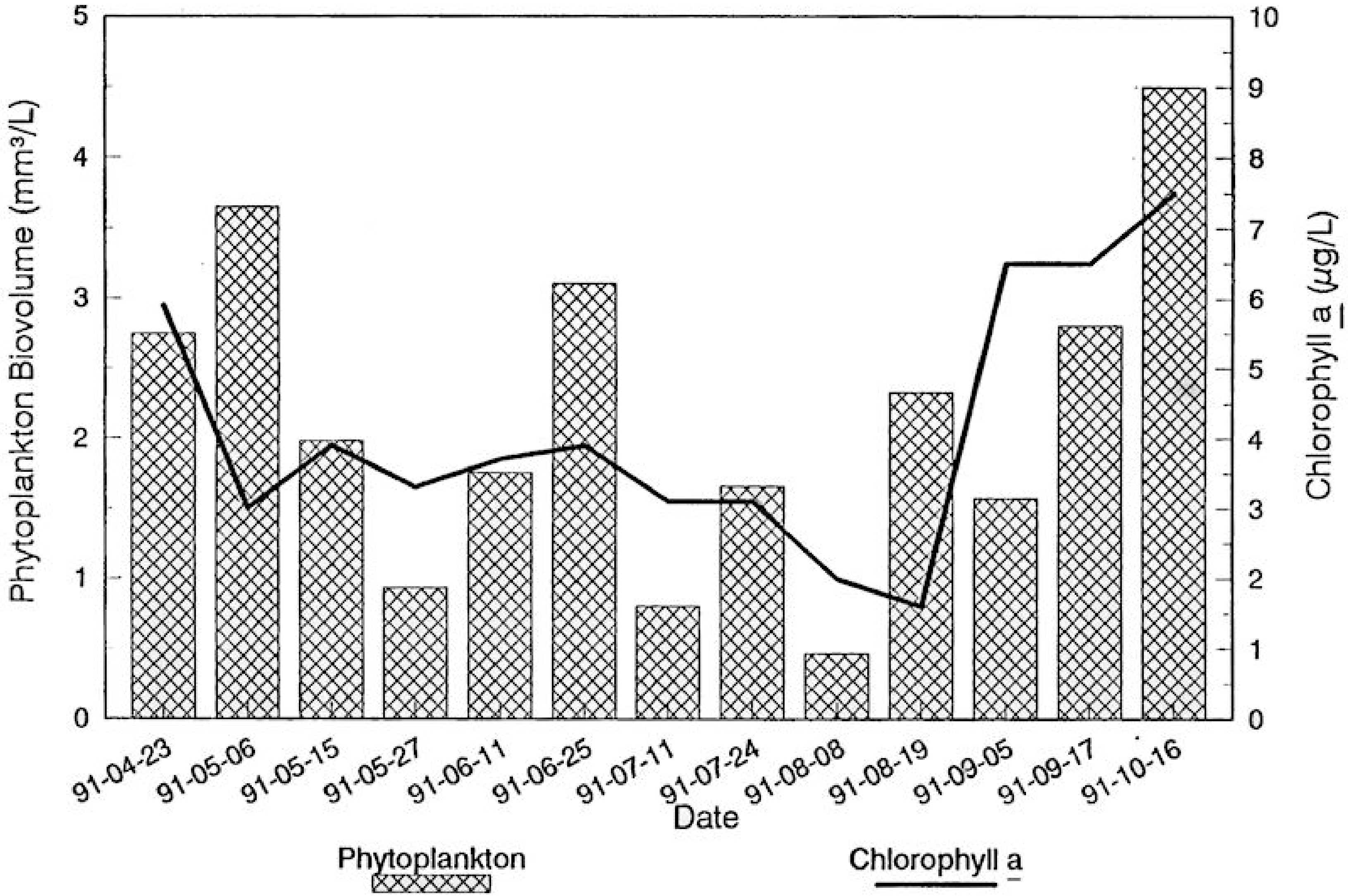

Figure 7: 1991 seasonal phytoplankton biovolume and total chlorophyll a concentrations.

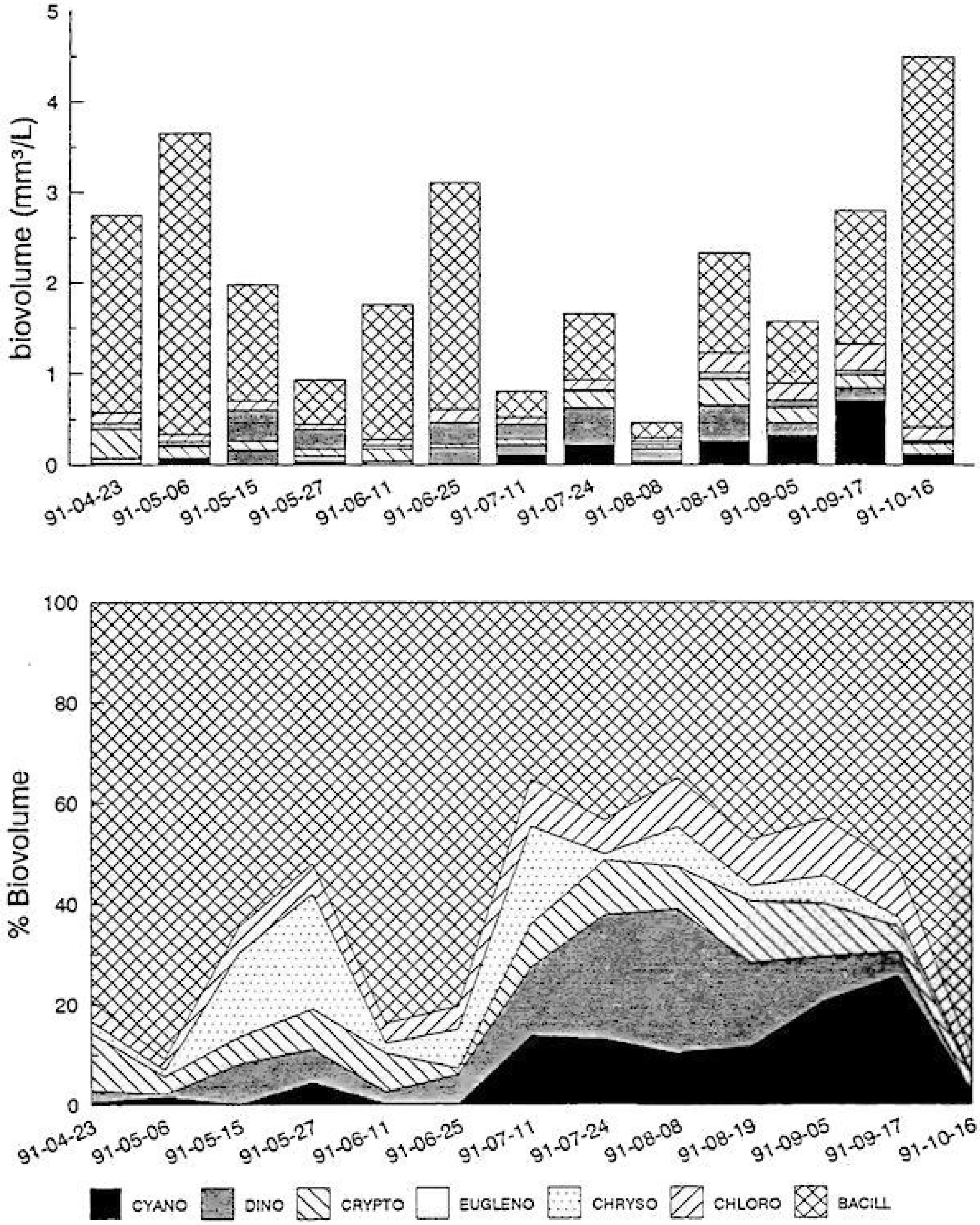

Figure 8: Seasonal Phytoplankton succession at Chesley Lake during the year 1991.

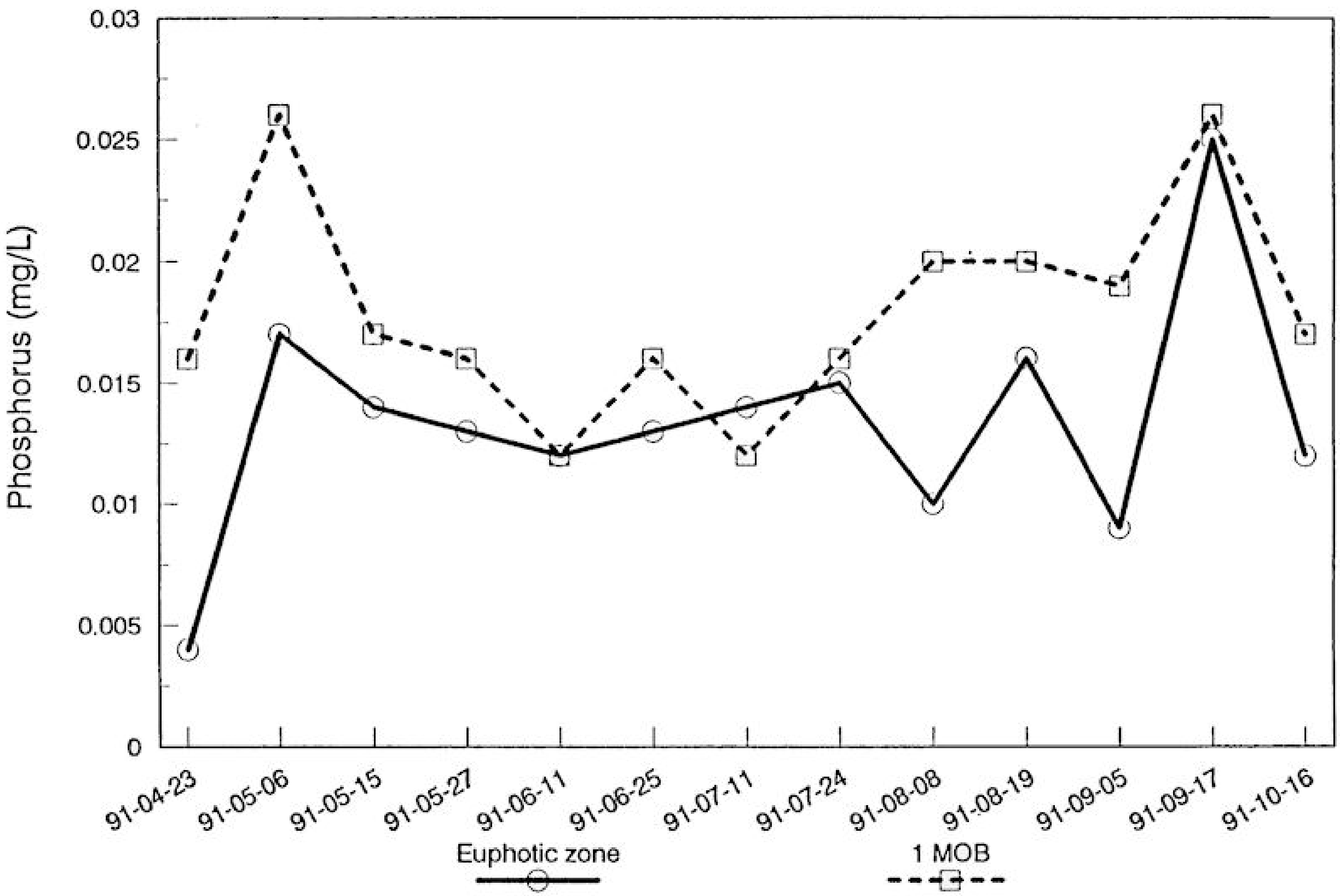

Figure 9: Seasonal euphotic zone and 1 MOB total phosphorus concentrations.

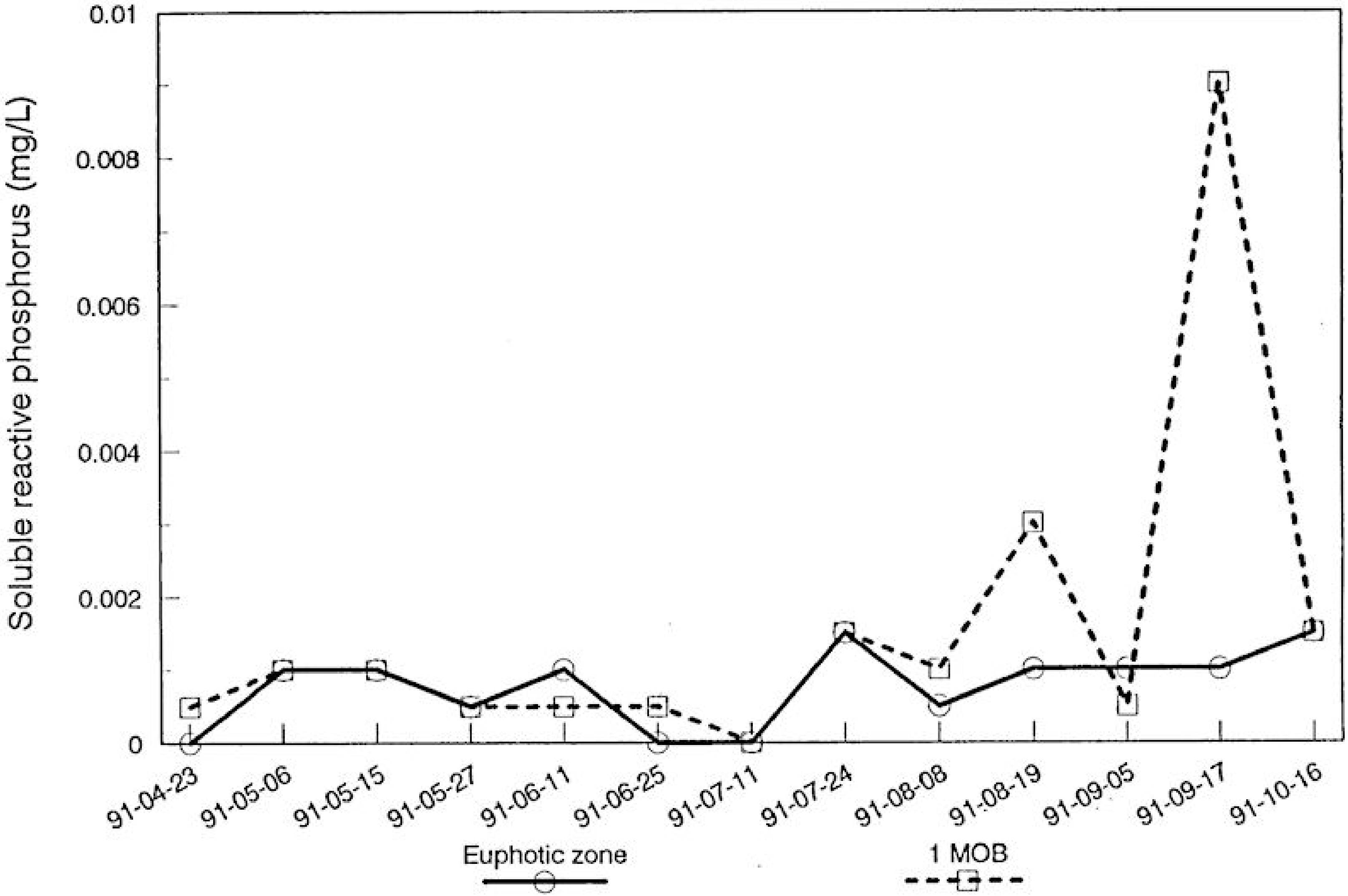

Figure 10: Seasonal soluble reactive phosphorus concentrations in the euphotic zone and at 1 MOB.

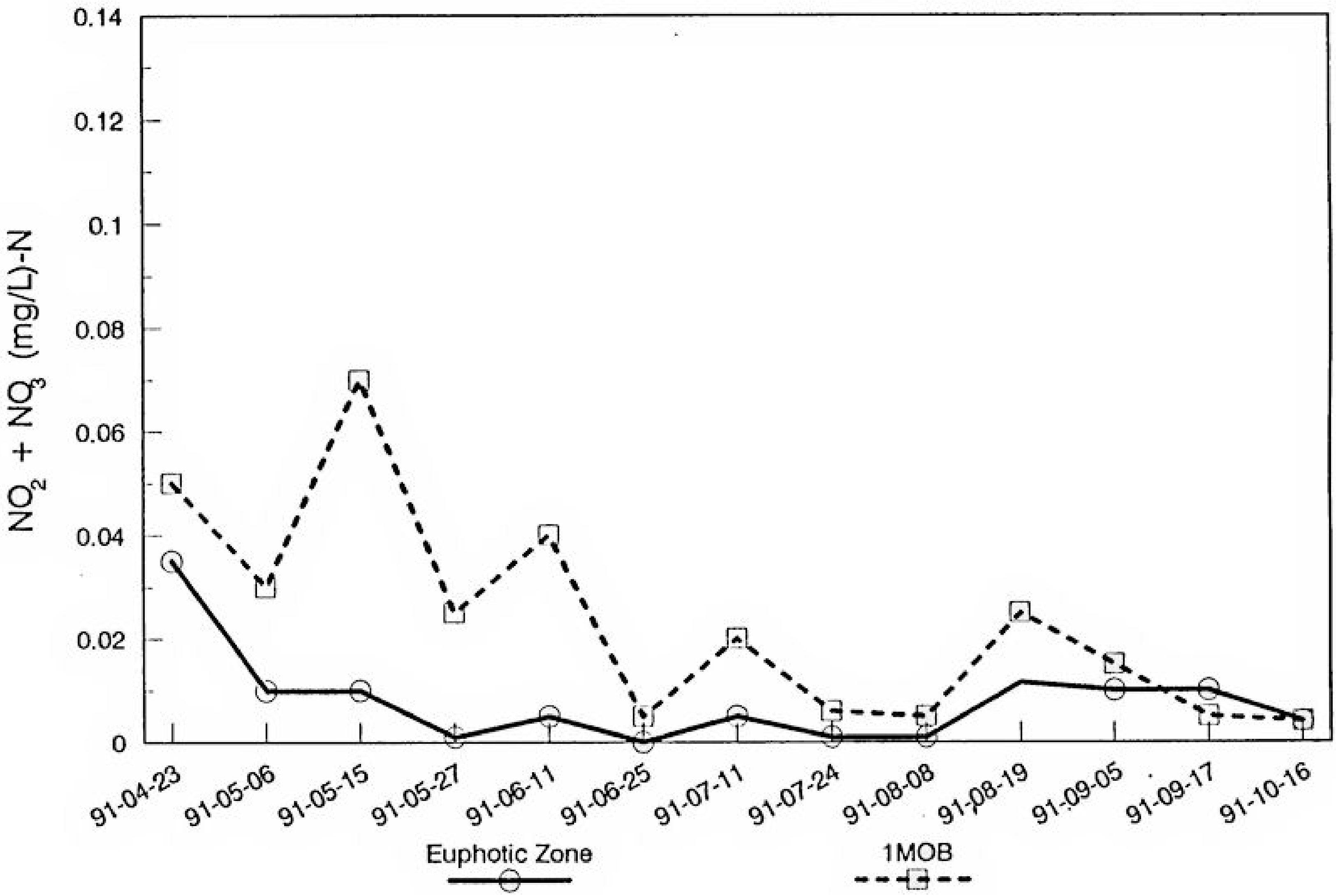

Figure 11: Seasonal total nitrate nitrogen concentration (NO_2 +NO_3), in the euphotic zone and at 1 MOB.

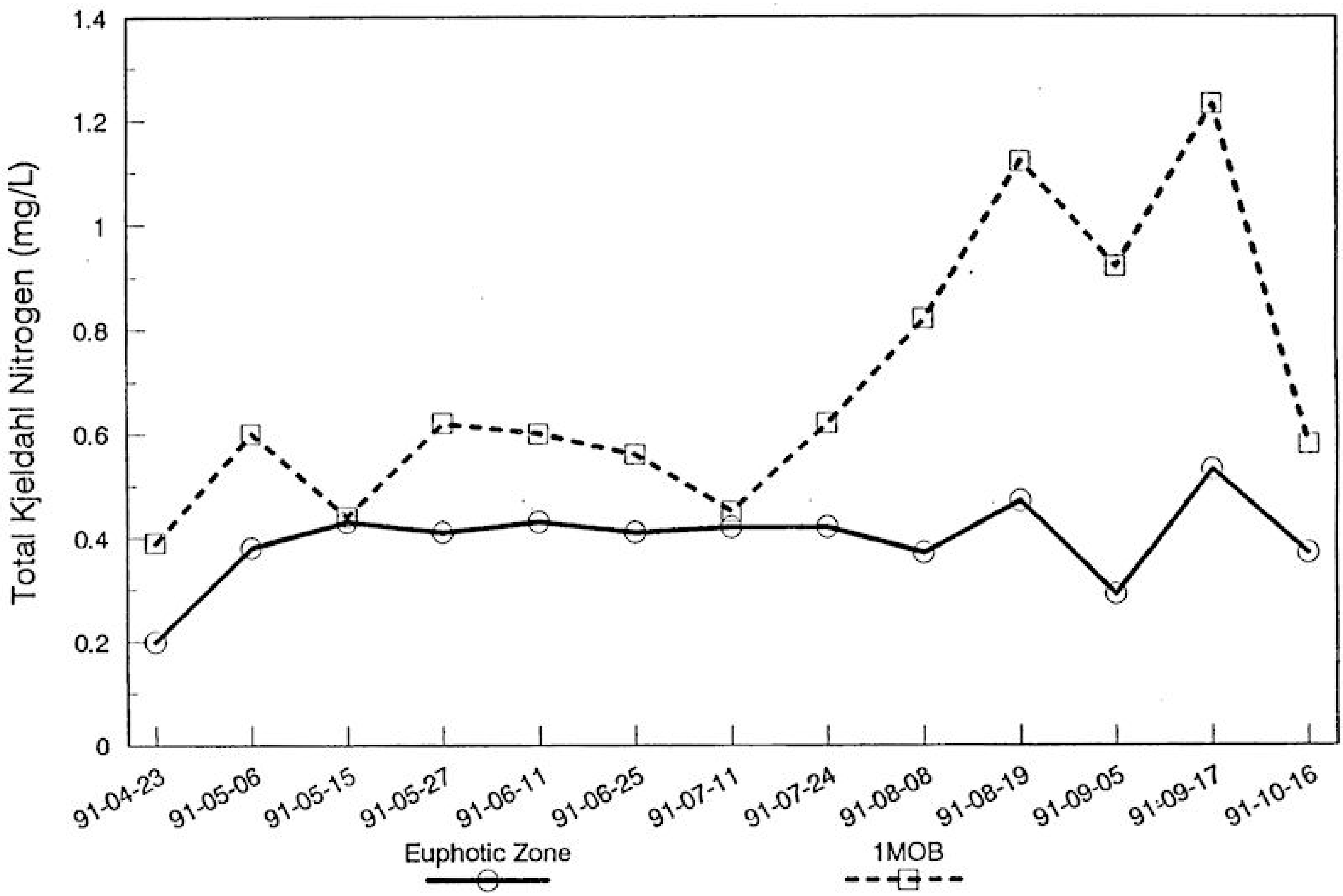

Figure 12: Seasonal total kjeldahl nitrogen levels in the euphotic zone and at 1 MOB.

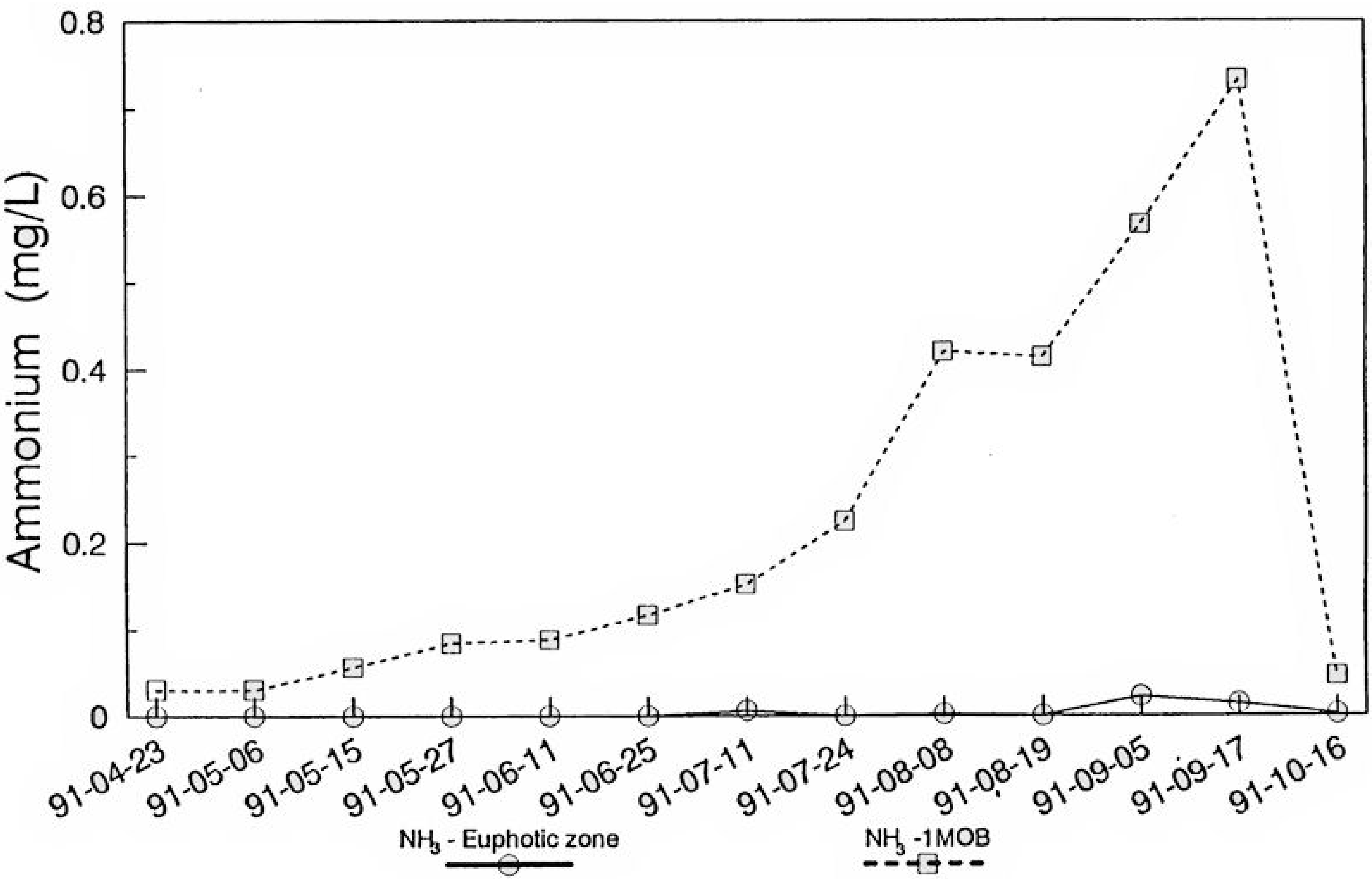

Figure 13: Seasonal ammonium levels in the euphotic zone and at 1 MOB.

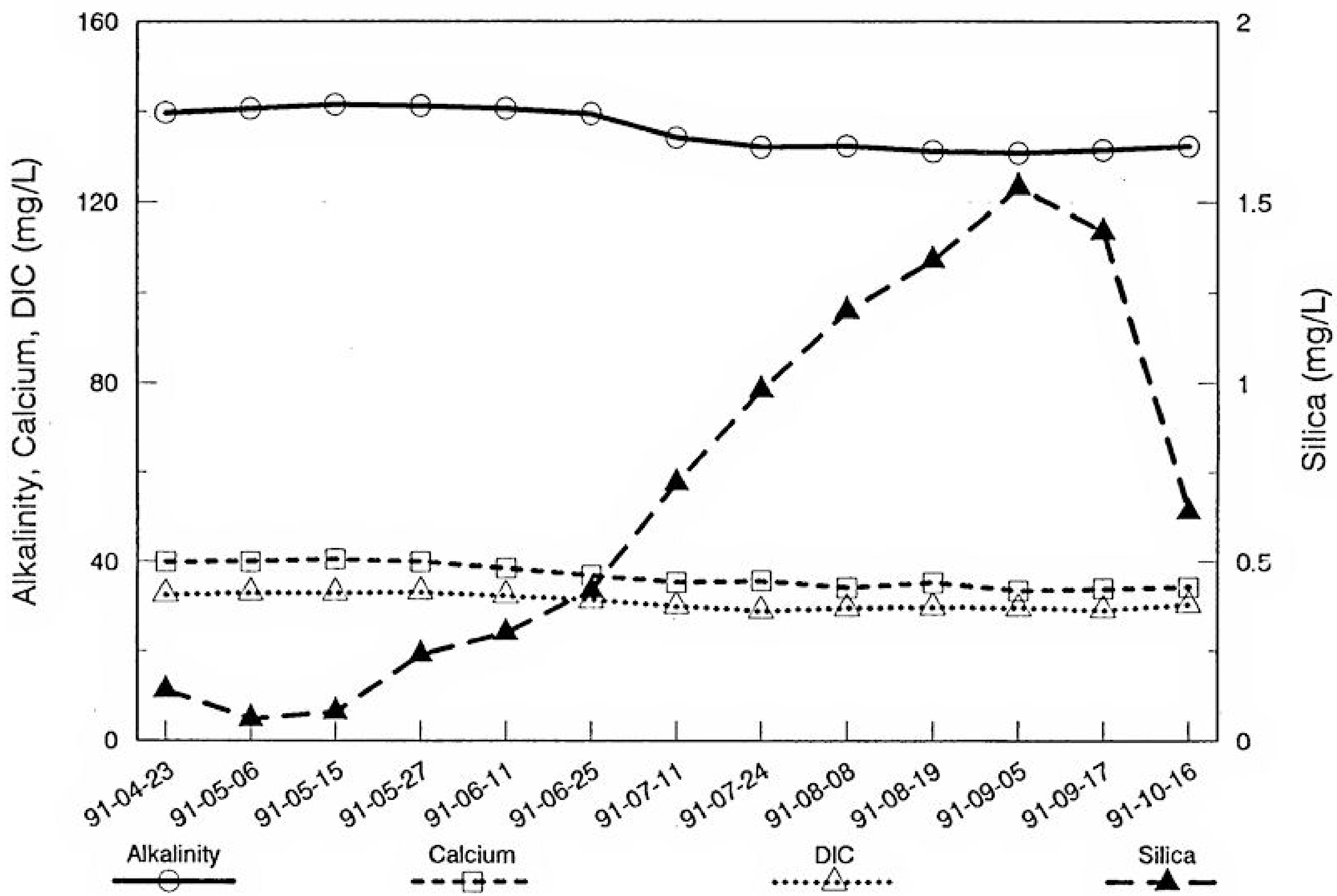

Figure 14: Seasonal euphotic zone alkalinity, calcium, DIC and silica levels.

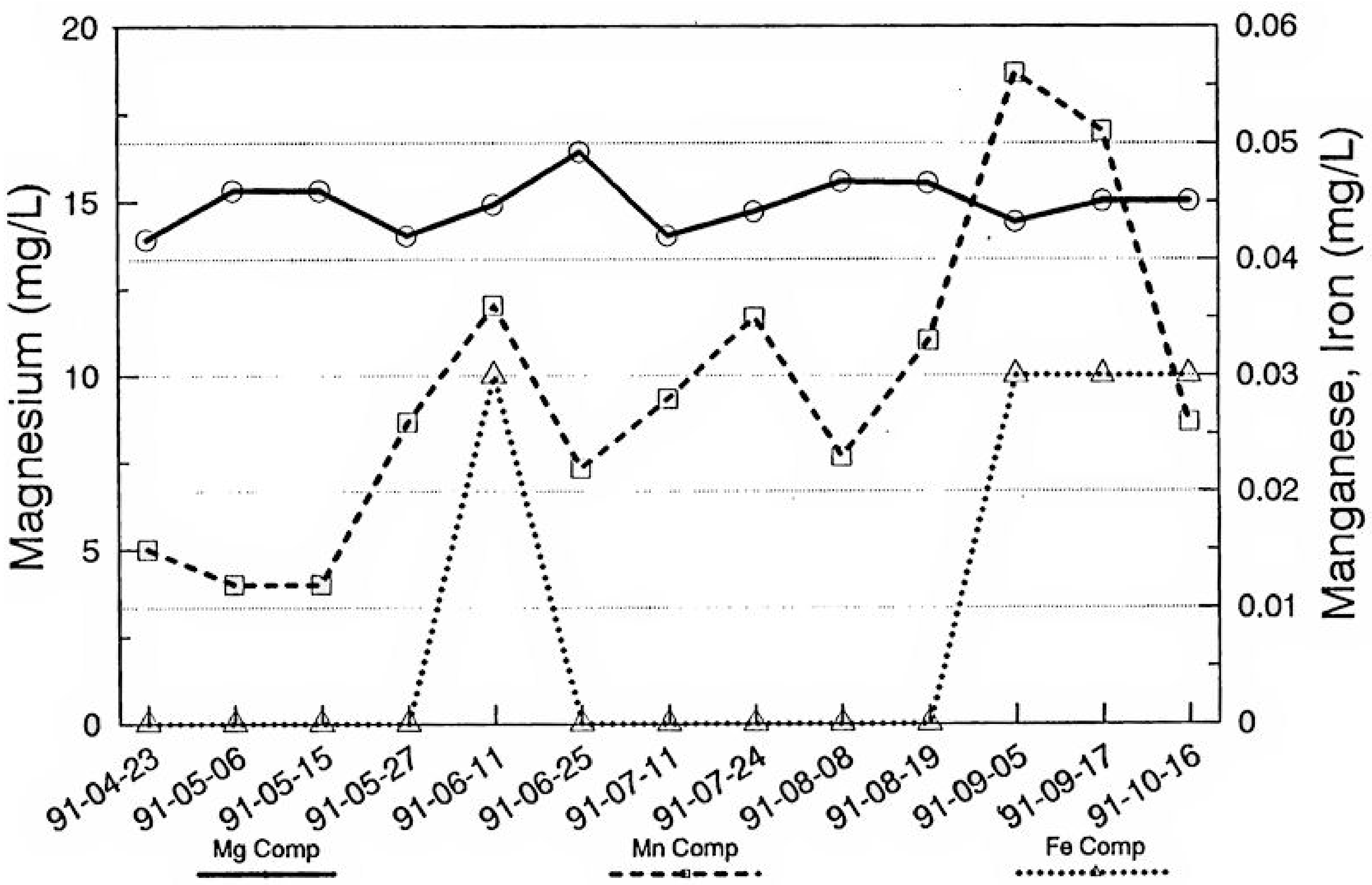

Figure 15: Seasonal euphotic zone iron, magnesium, and manganese concentrations.

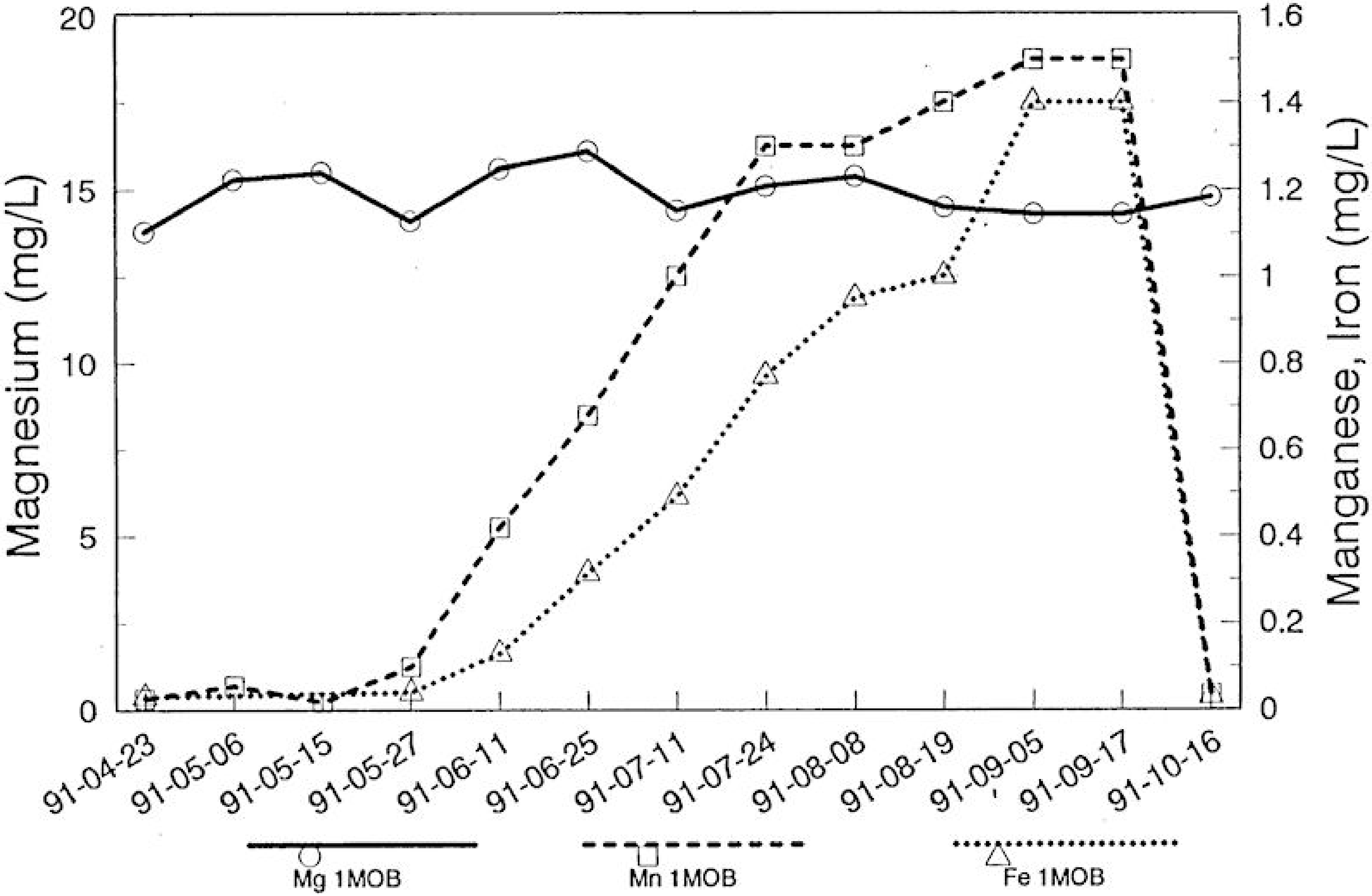

Figure 16: Seasonal iron, magnesium and manganese trends at 1 MOB.

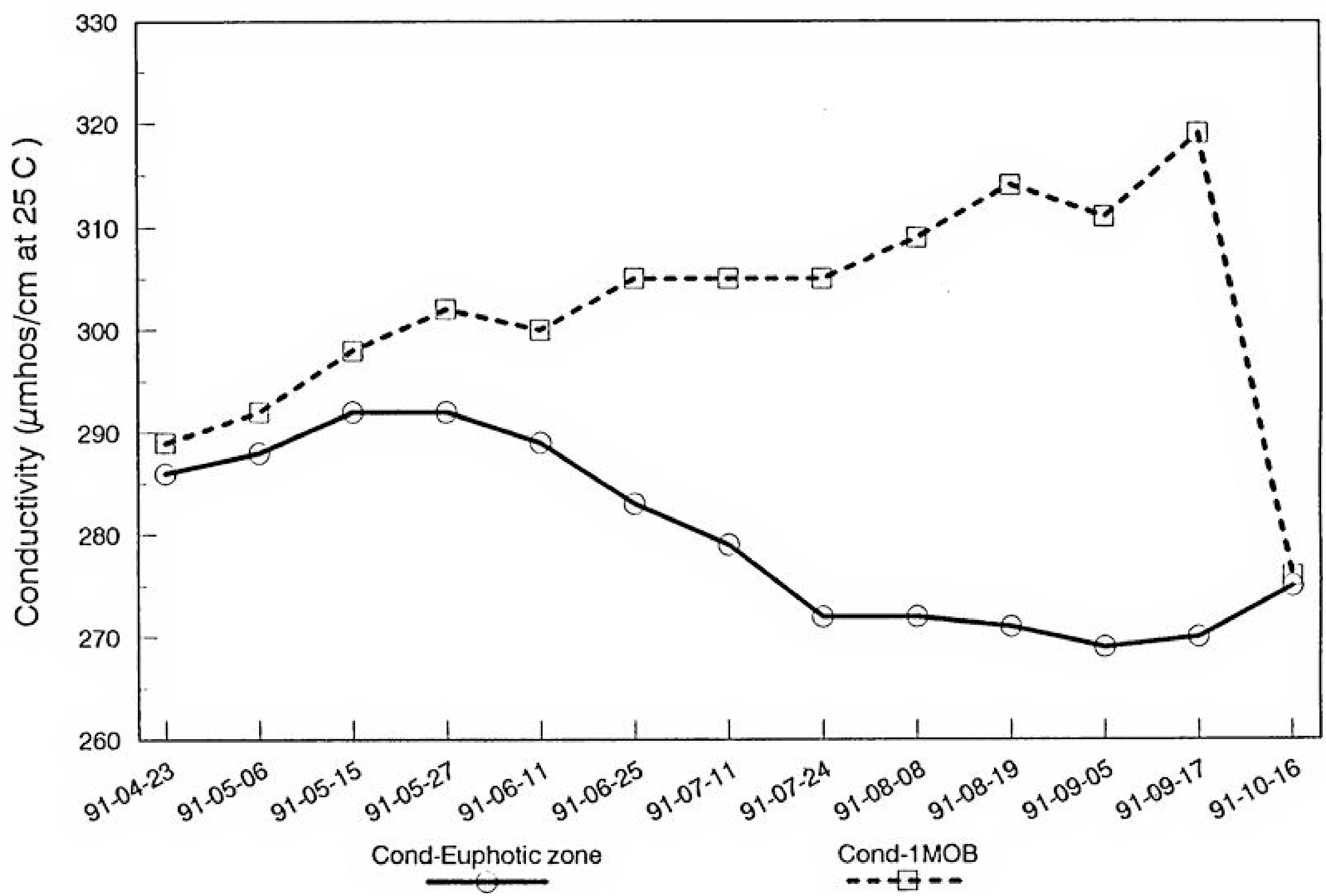

Figure 17: Seasonal euphotic zone and 1 MOB conductivity readings.

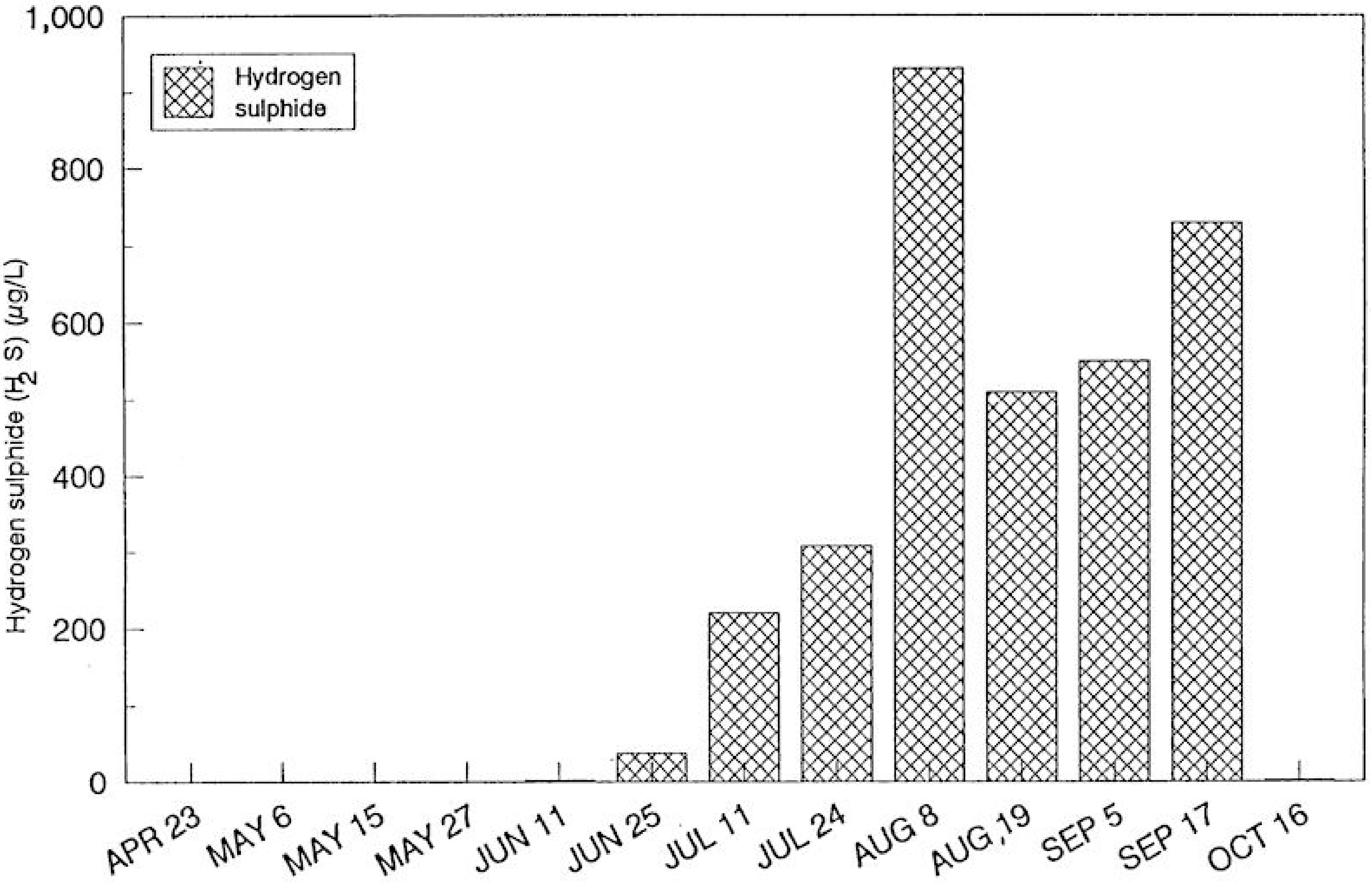

Figure 18: Seasonal hydrogen sulphide trends at 1 MOB.

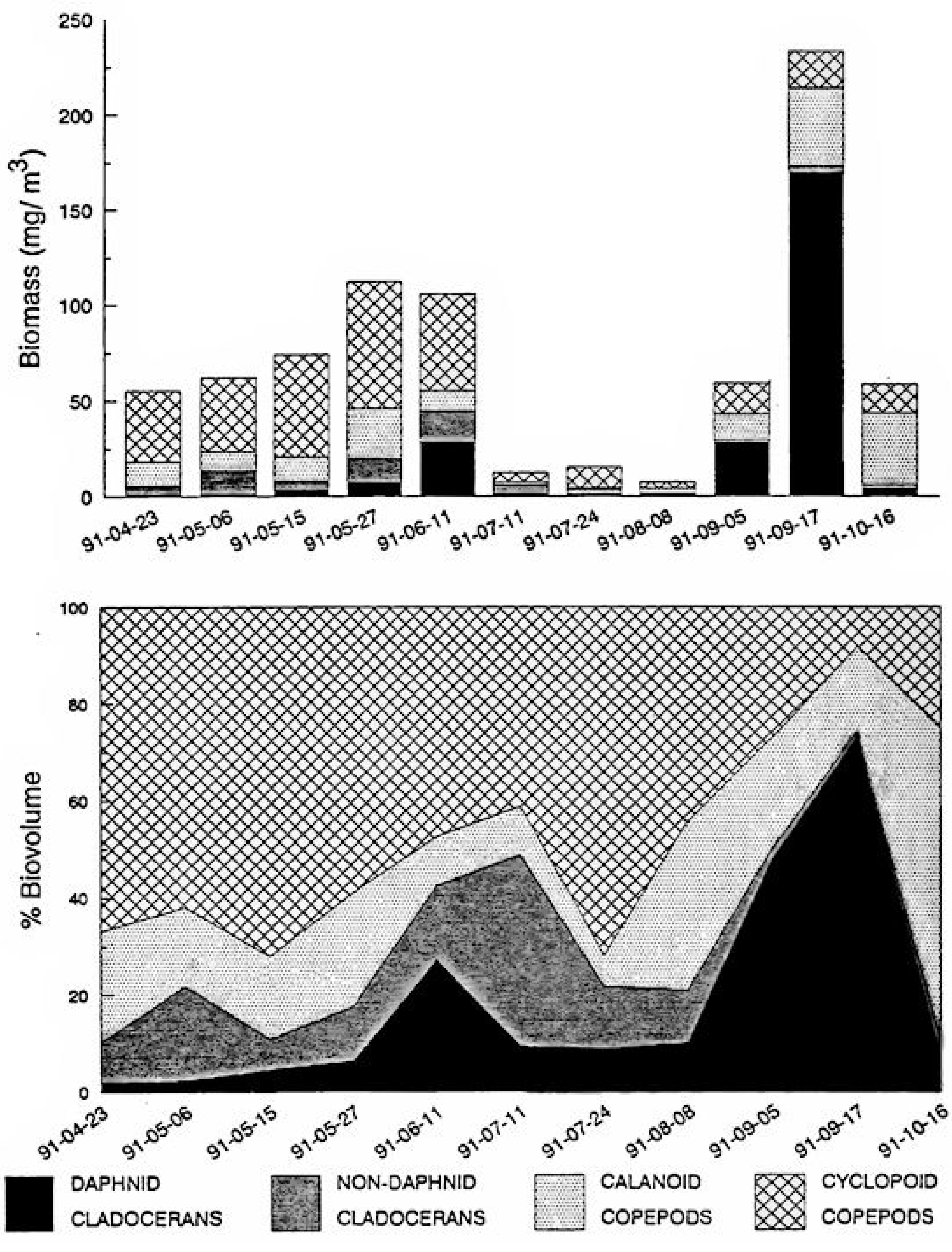

Figure 19: Seasonal zooplankton succession at Chesley Lake, 1991.

Appendix A: Raw water chemistry results of key water quality variables for the ice-free season of 1991. Euphotic Zone.

Date	Secchi disc (m)	Total chloro a (ug/L)	Total P (mg/L)	Soluble reactive P (mg/L)	Ammonium (mg/L)	Total Nitrate (mg/L)	TN:TP Ratio	Conductiv (µmho/cm)	Iron (mg/L)
Apr 23	2.5	5.9	0.004	0.0005	0.002	0.035	58.9	286	0.02
May 6	2.0	3.0	0.017	0.001	0.002	0.01	22.9	288	0.02
May 15	2.5	3.9	0.014	0.001	0.002	0.01	31.4	292	0.02
May 27	2.5	3.3	0.013	0.0005	0.002	0.005	31.9	292	0.02
Jun 11	1.8	3.7	0.012	0.001	0.002	0.005	36.3	289	0.03
Jun 25	2.5	3.9	0.013	0.0005	0.002	0.005	31.9	283	0.02
Jul 11	2.25	3.1	0.014	0.0005	0.006	0.005	30.4	279	0.02
Jul 24	2.3	3.1	0.015	0.0015	0.002	0.005	28.3	272	0.02
Aug 8	2.6	2.0	0.010	0.0005	0.002	0.005	37.5	272	0.02
Aug 19	2.2	1.6	0.016	0.001	0.002	0.12	36.9	271	0.02
Sep 5	2.4	6.5	0.009	0.001	0.022	0.01	33.3	269	0.03
Sep 17	2.3	6.5	0.025	0.001	0.014	0.01	21.6	270	0.03
Oct 16	2.5	7.5	0.012	0.0015	0.002	0.005	31.3	275	0.03
1 meter off bottom (1 MOB)									
	H_2S (mg/L)								
Apr 23	0		0.016	0.0005	0.03	0.05	27.5	289	0.03
May 6	0		0.026	0.001	0.03	0.03	24.2	292	0.02
May 15	0		0.017	0.001	0.056	0.07	30.0	298	0.02
May 27	0		0.016	0.0005	0.084	0.025	40.3	302	0.04
Jun 11	2		0.012	0.0005	0.088	0.04	53.3	300	0.13
Jun 25	38		0.016	0.0005	0.116	0.005	35.3	305	0.315
Jul 11	220		0.012	0.0005	0.152	0.02	39.2	305	0.49
Jul 24	308		0.016	0.0015	0.224	0.005	38.8	305	0.77
Aug 8	930		0.02	0.001	0.42	0.005	41.0	309	0.95
Aug 19	510		0.02	0.003	0.414	0.025	57.3	314	1
Sep 5	550		0.019	0.0005	0.566	0.015	49.2	311	1.4
Sep 17	730		0.026	0.009	0.732	0.005	47.5	319	1.4
Oct 16	0		0.017	0.0015	0.046	0.005	34.1	276	0.03