Ministry of Water, Land and Air Protection
Province of British Columbia

CHEMICAL AND PHYTOPLANKTON CHANGES ASSOCIATED WITH COAL MINING IN THE VICINITY OF LONG LAKE, CAMPBELL RIVER, BC

Bernie Taekema
Environmental Protection, Vancouver Island Region

December 2002

TABLE OF CONTENTS

LIST OF FIGURES

1.0 Executive Summary

Quinsam Coal Corporation has operated a coal mine in the vicinity of Middle Quinsam Lake, Long Lake, No Name Lake and the Quinsam River since 1986. This mine includes two underground workings, several open pit operations and a coal processing area. At present, only the underground workings located near Middle Quinsam Lake are being mined. Both groundwater and surface water originating from the mine site are directed to settling ponds to remove settleable solids prior to being discharged to Long Lake and Middle Quinsam Lake.

Sampling of both Middle Quinsam Lake and Long Lake occurred prior to mine operation. Results indicated both lakes were oligotrophic, were slight to moderately nitrogen limited and had phytoplankton communities representative of low productivity. Sulphate concentrations were negligible while the pH level was near neutral.

Several years after the mine began operating, sulphate concentrations which had background levels around 2.0 mg/L began to rise in both lakes, but most substantially in Long Lake where it peaked near 200 mg/L in 1999. Although the sulphate concentration rose at all depths, the most dramatic rise was in the hypolimnion layer of the lake during the summer months, once a thermocline was established. During these months, flushing of the lake is confined to the epilimnion layer resulting in comparatively lower sulphate concentrations at and near the surface. Sources of sulphate in Long Lake included the surface discharge from a settling pond and groundwater likely originating from underground workings to the south of the lake.

The pH in Long Lake also increased significantly at all depths since the spring of 1997, the probable cause being the disturbance of overburden and soil by mining activities. Nutrient levels have remained relatively constant.

This report focuses on the effect the rise in sulphate concentrations and pH levels has had on the Long Lake phytoplankton community. The data indicate that changes to the phytoplankton community in Long Lake have occurred. The most obvious change was the appearance of the cryptophyte species, *Chryptomonas obovoidea*, which appeared in large concentrations in the deeper waters of Long Lake. The literature suggests that this organism prefers water with high sulphate concentrations. Although some phytoplankton species increased in number and frequency, others disappeared or had a significant reduction in their counts. Trends in diversity, as measured by a diversity index, did not indicate any substantial changes to the phytoplankton community. Excluding the appearance of *C. obovoidea*, the Long Lake phytoplankton community was similar to that found in Middle Quinsam Lake.

2.0 Introduction

Quinsam Coal Corporation operates a coal mine in the Middle Quinsam Lake area, west of Campbell River, British Columbia (See Figure 1). The mine began as a small open pit operation in 1986, and subsequently expanded to a second open pit, both of which have since had underground workings developed in association with them. At present, only the underground workings associated with the original pit (north pit) are in operation.

The mining area has the potential to generate acid rock drainage (ARD) although the acid base accounting tests indicate that there is significant buffering capacity offered by native calcareous rock (*Sturm Environmental Services, Inc. 1996*). At present, any ARD is immediately neutralized.

This report focuses on the increasing concentration of sulphate and rising level of pH in Long Lake and its effect on the abundance and diversity of the phytoplankton community. Typically, impacts on the phytoplankton community can negatively impact the zooplankton community, which in turn impacts the fisheries resource. In order to achieve a comprehensive understanding of the changes occurring to the phytoplankton community, the lymnology of the lake, pH, sulphate sources and nutrient concentrations were also considered. Each of these factors has a considerable influence on the biology of the lake.

The overall objective of this report is to note any significant changes in the phytoplankton community of Long Lake since mining activities began and to determine if there is a correlation between these changes and the rising pH levels and sulphate concentrations in the lake.

Sulphate concentrations in Middle Quinsam Lake and No Name Lake have also risen, but to a lesser extent than in Long Lake. Similar to Long Lake, Middle Quinsam Lake receives a substantial sulphate loading. However, it is subject to a higher degree of water renewal and concentrations are much lower than in Long Lake. To date, sulphate concentrations in No Name Lake are lower than those found in Middle Quinsam Lake. Based on this information, assessments of Middle Quinsam Lake and No Name Lake were not included in the scope of this report although the phytoplankton community in Middle Quinsam Lake was compared to that in Long Lake.

When the mine was first proposed, an inquiry was held to assess potential impacts. The primary concerns at the time were increased total suspended solid, increased nutrient concentrations and ARD potential. Baseline data, including water quality, hydrology and fisheries information was generated in the 1983 – 1985 period prior to the mine operation.

Shortly after the mine began operation, provisional water quality objectives for waters in the Middle Quinsam Lake sub-basin were set for a number of parameters, including total phosphorus, non-filterable residue, total ammonia, nitrate, nitrite, pH and several metals (*Kangasniemi 1989)*. Nutrients were included because of the expectation that explosives and land disturbance could increase the concentrations of nitrogen compounds. This could result in a doubling of algal productivity in Long Lake, which would put the lake at a significant risk of serious hypolimnetic oxygen depletion and attendant impacts on salmonids and their supporting food webs (*MacIsaac and Stockner 1985*).

As a condition of the mine's original waste management permit, the permittee was required to follow a limited sampling and monitoring program. The data generated was primarily associated with discharges from the mine site. In 1994, the permit was amended as a result of proposed mine expansion. Because of concerns over potential impacts on the receiving environment, the amended permit required a more extensive monitoring program involving Long Lake, Middle Quinsam Lake, No Name Lake and the Quinsam River. This report focuses on the data generated in the period from 1994 to 1999. Parameters analysed during this period included pH, conductivity, metals, sulphate, nutrients, periphyton and phytoplankton.

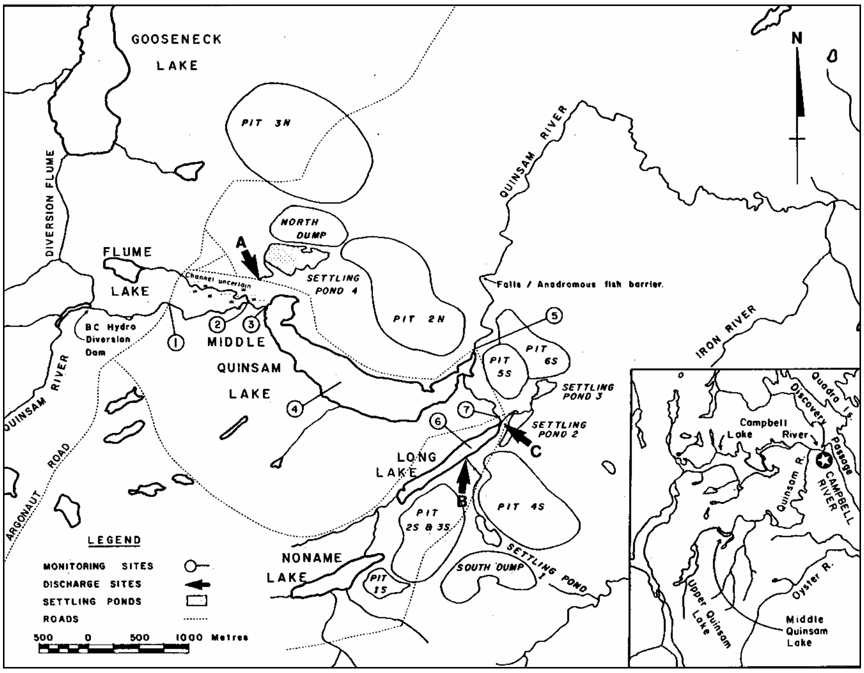
The Ministry of Water, Land and Air Protection (WLAP) has also sampled these water bodies routinely over the past several years. WLAP, in partnership with

Environment Canada, also operates a long-term monitoring station on the Quinsam River near its confluence with the Campbell River to monitor changes in water quality, including sulphate concentrations.

3.0 Site Description

The mine site is located 27 kilometers southwest of Campbell River, British Columbia. Middle Quinsam Lake, Long Lake and No Name Lake are part of the larger Quinsam River drainage basin (Figure 1). No Name Lake lies just to the west of Long Lake, both of which are south of Middle Quinsam Lake. No Name Lake discharges to Long Lake, which in turn discharges to the eastern end of Middle Quinsam Lake near its outflow to the Quinsam River. Middle Quinsam Lake is the largest of the three lakes.

Figure 1. Map of Middle Quinsam Lake Sub-Basin



Long Lake is approximately 1.3 kilometers in length and has an average width of 150 metres. It has a southwest to northeast orientation with the lake inlet located at the southwest end and the outlet at the northeast end. The mean depth of Long Lake is seven metres while the maximum depth of 18 metres is found in a small pocket just to the north of the centre of the lake. The littoral zone, < 5 metre water depth, encompasses approximately 35 percent of the lake area.

Water flows vary greatly throughout the year with monthly percent water renewals of <5 percent common during summer months of May to September. During this period, the renewals are primarily confined to the epilimnion, which ranges from four to six metres in depth because of thermal stratification. Winter flushing and the associated vertical mixing results in percent water renewals ranging from 90 – 140 percent, accounting for an annual residence time of 0.07 years (*MacIsaac and Stockner 1985*). During this time, the water will be relatively homogeneous.

Long Lake is typical of many low altitude lakes on Vancouver Island. These lakes can be ice-covered for short periods of time during the winter months. As such, Long Lake varies between being monomictic and dimictic. A monomictic lake typically has water that does not fall below 4°C, circulates freely in the winter at or above 4°C, stratifies during the summer months and has turnover only once a year, typically in late fall. A dimictic lake has turnover twice a year, once in the fall and once in the spring shortly after the ice melts.

The lake is adjacent to the area of the mine site known as the south pits. This area has three, relatively small, open pits and two underground mining areas. Run-off water and water associated with the underground workings from these four areas is directed to a settling pond prior to discharging to the lake near its outlet.

4.0 Methods

Over the history of the mine, several different groups sampled the lakes, streams and works associated with the mine. Baseline sampling and the resulting data were primarily produced by Environment Canada, although some phytoplankton data was produced by Quinsam Coal. A majority of the data included in this report from 1994 onward was produced by Quinsam Coal as part of their permit requirement, although some limited WLAP data was also included. All non-baseline data in this report was identified and counted by Munro Environmental Consulting. All raw data reported as less than the minimum detection limit was recorded at the minimum detection limit concentration in the figures associated with this report.

Quality Assurance data was incomplete in the baseline data. That which was supplied as well as that for the period of 1994 -1999 was found to be acceptable and is available upon request.

Environment Canada personnel used a Hydrolab to measure depth, temperature, pH, conductivity, dissolved oxygen and light transmission at one metre intervals. Discrete water chemistry and phytoplankton samples were collected in a six litre Van-Dorn water bottle. Portions of each discrete sample were distributed to various sample containers. The phytoplankton sample was preserved with Lugol's solution and stored in the dark until analyses occurred. All samples were kept cool until arrival at the lab.

Methodology practiced by Quinsam Coal staff for water quality sampling was similar to that of Environment Canada. All discrete samples were obtained using a two litre Wildco Van-Dorn sampler. Portions of each sample in the Van Dorn were poured into a 250 millilitre amber wide mouth bottle for phytoplankton analysis and a clean, polyethylene bottle for the remaining parameters. The phytoplankton samples were preserved with Lugol's solution. No preservatives were added to the polyethylene bottle. Samples were cooled by icepacks to 4 C° and shipped to the lab the same day for analysis.

Vertical profiles of physical chemistry data, including pH, dissolved oxygen, water temperature, redox potential and conductivity were measured on site using a Hydrolab. The information collected was used in part to confirm the presence of a thermocline in the lake.

WLAP staff followed a similar methodology while collecting samples.

With the exception of some baseline samples, all samples were obtained from one representative multidepth sampling location (EMS # E206619) on Long Lake. In almost all cases, samples were taken at the 1 m, 4 m and 9 m depths. Typically, samples were taken between April and September.

5.0 Results

With the exception of pH (November 1981), all data assessed in this report was generated during two periods of time. The initial data set was collected from 1983 to 1985, prior to the start up of the mine while the second, from 1994 to 1999, was collected during mining operation. The latter period was considered representative of lake conditions during the time when sulphate concentrations were increasing. Baseline data was obtained from Environment Canada and International Environmental Consultants Ltd. (*Environment Canada, 1983; Environment Canada, 1984; Environment Canada, 1985; International Environmental Consultants, 1982)* while, as noted previously, the permittee provided the data from 1994 - 1999.

In general, only data for the months of May, July and September were reviewed as these months were considered representative of the conditions of the lake during its most productive phase. In cases where data for these months were not available, data from the preceding or following month was used. All data assessed in this report was based on samples collected by Quinsam Coal with the exception of the pH, nutrient and sulphate baseline data, generated by Environment Canada. Data generated by WLAP was used as a comparison for certain parameters and time periods.

As previously stated, sulphate is used as an indicator of mining impacts at the mine site. To confirm that this was the appropriate parameter, samples were obtained from both the No. 4 settling pond discharge and the outlet of Long Lake in January 2000 and an anion/cation balance requested. Results indicated that sulphate is the dominant anion, while calcium is the dominant cation, although potassium is also relatively abundant. Sulphate is a stable anion in solution and only turns into an insoluble salt, such as sodium sulphate, potassium sulphate, magnesium sulphate or calcium sulphate, when it or one of the cations it complexes with exceeds its saturation level (equilibrium constant) in water. The saturation level of sulphate depends on a number of factors, including temperature and pH, but typically ranges between 2000 and 3000 mg/L (*Clark, 2000, per. com.*).

Baseline data from Long Lake (Figure 2) showed the sulphate concentrations to be approximately 2 mg/L at all three depths. By June 1994, sulphate concentrations had increased to 25 mg/L at the 1 m and 4 m depths and then increased further to 90 mg/L in June 1995. Concentrations remained at or below this level until July 1997 at which time they increased again to a peak of 130 mg/L at 1 m in September 1998 and 170 mg/L at 4 m in April 1999. It should be noted that, with one exception (April 1999), sulphate concentrations at 1 m and 4 m were almost identical but varied considerably from those at 9 m. The data also showed that sulphate concentrations at these two depths were usually lowest in the spring and the highest in the fall of any given year.

Sulphate concentrations at the 9 m depth increased consistently from June 1994 to June 1999 when they peaked at 200 mg/L. They remained at or near this concentration through September 1999. Of interest is that in both 1995 and 1996, the increases occurred prior to the first set of samples being obtained in the spring as concentrations during the sampling period did not increase. With the exception of 1997 when concentrations increased throughout the sampling period, there was no seasonal variation in the concentrations at this depth as was the case at the 1 m and 4 m depths.

WLAP data generated during the 1999 – 2000 period confirms the rise in sulphate concentrations.

Based on the sulphate concentrations at the 9 m depth, it was suspected that sulphate rich groundwater was discharging to Long Lake. This was confirmed during an April 2000 field inspection when a large groundwater seep was observed discharging to Long Lake along its south shore. Samples from the seep, taken monthly from May to September 2000, ranged from 576 mg/L to 653 mg/L. Groundwater monitoring wells have since confirmed an underground hydraulic connection between the south pit works and Long Lake.

The pH of Long Lake had also changed since baseline data was obtained. Baseline data in November 1981 and the 1984 – 1985 periods indicated the pH at all three sampling depths was close to 7.0 although the pH at 9 m tended to be slightly lower. This tendency, with only a couple of exceptions, was noted during the entire study period. The June 1983 data showed pH levels at all three depths above 7.5. As there was no other data for 1983 to support these levels, it was not considered to be significant.

Figure 3 shows that the pH remained around 7.0 until May 1995 at which point it rose to an average of 7.5 at 1 m and 4 m before returning to background levels in September 1995. It should be noted that field data for the 1994 period showed a pH increase to 9.0. A review of the lab data for the same period indicated no increase during this period. It is the writer's opinion that the lab data is correct and that data generated in the field was likely the result of an incorrectly calibrated meter. After September 1995, the pH at all three depths had risen substantially and in 1999, was approximately 7.7 at all three depths.

Baseline data for total nitrogen indicated that concentrations ranged between 70 and 150 μg/L at 1 m and 4 m while concentrations at 9 m ranged between 60 and 120 μg/L (Figure 5). With only a couple of exceptions, concentrations at all three levels rose slightly, peaking during the period between July 1996 and May 1998 before receding to baseline concentrations in July and September 1998. The two exceptions were spikes of total nitrogen at 9 m in May 1994 (240 μg/L) and May 1997 (270 μg/L). Concentrations at all three depths in July and September 1999 ranged between 70 and 90 μg/L, below baseline levels.

Baseline data for nitrate showed concentrations $\leq$ 20 μg/L at all three sampling depths in Long Lake (Figure 6). If the June 1983 data is excluded, the average concentration is around 5 μg/L. Pre-mine sampling completed by the Department of Fisheries and Oceans indicate nitrate concentrations in the epilimnia at or near detection limits during the summer months (*MacIsaac and Stockner 1985*). From July 1993 through September 1999 concentrations at 1 m and 4 m continued to be at or below detection limits while those at the 9 m depth were consistently higher (it is important to note that the detection limits ranged from $\leq$5 to $\leq$ 20 μg/L during this period). May 1994 had a nitrate concentration of nearly 80 μg/L while those for the same month in 1996, 1997 and 1998 were approximately half of that.

Total phosphorus concentrations ranged from 2 to 4 μg/L during the 1984 - 1985 baseline years. As was the case with nitrate, the June 1983 concentration was much higher than all other total phosphorus data (the June 1983 data was not supported by any other data during that period of time and is considered to be data of poor quality assurance and not reliable. It was not considered in the context of this study). Baseline data in the 1985 Department of Fisheries and Oceans report states that phosphorus concentrations ranged from 2 to 4 μg/L in the winter through early summer and 1 to 3 μg/L during August and September (*MacIsaac and Stockner, 1985*).

Figure 7 indicates that concentrations during the 1993 – 1994 period had increased to an average 4.0 μg/L at all three depths. From 1995 through 1998, concentrations varied considerably, ranging from <1.0 mg/L in May 1997 at all three depths to 11 mg/L and 10 mg/L at the 4 m depth and 9 m depth respectively in 1998. Concentrations decreased to near baseline levels in 1999.

Phytoplankton was assessed through tracking both species abundance and species diversity from 1984 to 1999 (Figures 8 and 27). Phytoplankton abundance during the baseline years, with the exception of September 1985 and September 1986, was low compared to the 1994 to 1999 period although species diversity was higher, with the exception of 1999. The difference in abundance is partially due to the fact that Environment Canada did not sample the 1 m depth in 1984, the depth which typically has the highest phytoplankton abundance.

From 1994 to 1999, phytoplankton abundance varied considerably from month to month and from year to year. Typically the month of May had the highest phytoplankton abundance with September typically being the next most productive month. This correlates with the spring and autumn lake turn-over events that normally occur during these months. This also indicates that Long Lake is dimictic.

Figure 8 also shows the abundance of phytoplankton at the three depths over the study period. Baseline data was only available for the 4 m and 9 m depths in 1984 but was generated for the 1 m depth in 1985 (a number of May and July samples were obtained from a slightly shallower or deeper depth but, for the purpose of this study, were considered representative of what would be found at the four and nine metre depths). Phytoplankton was most often more abundant at the 1 m and 4 m depths.

Shannon's index of diversity was used to assess the diversity of the phytoplankton community over the time period of the study. Calculations were made for each month that the study was assessed, as well as for each depth (Figure 26). An average of these indices was also calculated for each year (Figure 27). The data from the yearly averages indicates that the baseline diversity index ranged between 0.35 and 0.4. The index then dropped to 0.27 in 1994, the first year that phytoplankton data was available, after the mine began operation. It is not known if the index had started dropping prior to 1994. From 1994 onward, the index gradually rose and in 1999 was at 0.4.

Long Lake supports a significant number of phytoplankton species. These species are included in the Chrysophyceae, Cyanophyceae (blue-green algae), Bacillariophyceae (diatoms), Chlorophyceae (green algae), Cryptophyceae, Dinophyceae and Euglenophyceae families. A number of species or genera from these different families were tracked over the study period. These are considered indicators of good diversity and typical assemblages for oligotrophic lakes on Vancouver Island (*Munro*, *1999 per com.*)

The predominant phytoplankton species in Long Lake during the course of the study were the chrysophytes *Ochromonas/Chromulina* spp and *Chrysochromulina parva.* These were most numerous in May and September and, although found in numbers at all three depths, were most abundant at the 1 and 4 m depths (Figures 9 and 10). The abundance of *Ochromonas/Chromulina* spp. has not changed but that of *C. parva* has been increasing steadily since July 1994. Because these two species are considered ultra-nannoplankton (2 – 5 μm), their numbers typically contribute very little to the overall algae biomass (*Munro 1998*). Two other larger chrysophytes, *Dinobryon bavaricum* and *Dinobryon cylindricum* were also tracked but were much less abundant (Figures 11 and 12). *D. bavaricum* counts have been almost negligible since May 1996 at which time it appeared to have undergone a bloom . *D. cylindricum* was present in baseline and 1994 data but not noted from then until September 1998. In September 1999, concentrations were the highest noted during the study, likely due to a bloom event.

Of the Cyanophyceae (blue-green algae) family, *Aphanocapsa sp* and *Aphanothece sp* were assessed (Figures 13 and 14). These genera are counted as colonies, and because of their relatively large size, their abundance, as measured in colonies/mL, will typically be less than other recurring, but smaller, phytoplankton species in Long Lake. They can, however, contribute significantly to the overall algal biomass. *Aphanocapsa* sp. was noted intermittently, in low concentrations, at all three depths during the baseline years. In 1994, it was only found at 4 m, while it was counted at all three depths in September 1995. It has not been observed since then. Similarly, *Aphanothece* sp. was most prevalent during the base line years but has not been noted since September 1997. Other Cyanophyceae species have been counted over the study period, including the 1998/1999 period, and were found to be most abundant in September.

Of the Bacillariophyceae (diatom), three species were tracked. *Melosira italica* was noted at all three depths although it decreased in abundance since September 1996. *M. italica* was most abundant in July and September (Figure 15). *Cyclotella bodanica* was found, primarily in July and September, until September 1997 after which it was noted only once more (Figure 16). *Synedra radians* was not found during the baseline study but since then has been present (Figure 17). It is most abundant in the month of May and is found at all three depths. *Synedra radians* counts declined significantly in the spring of 1999, but this is likely because samples were taken in April rather than May, when it typically blooms. Several other Bacillariophyceae species were also noted in the data base. These were typically most abundant in May and were found at all depths.

Four species of Cryptophyceae were assessed. Cryptophytes are associated with deeper, darker water due to their photosynthetic pigments that improve light absorption at depth (*Munro Dec. 1999*). The abundance of *Rhodomonas minuta*, *Cryptomonas marssonii*, *Cryptomonas obovoidea* and, to a lesser degree *Kathablepharis ovalis* at the nine metre depth bear this out (Figures 18 – 21). C*ryptomonas marssonii* increased substantially in abundance in 1998, but in 1999, returned to levels noted in previous years. *Rhodomonas minuta* was relatively constant throughout the study period although there was a large increase in numbers in July 1998, indicating a bloom event.

C. obovoidea (Figure 20) underwent the most substantial change. It was noted in the 1984 baseline survey but at very low abundance. It did not appear again until May 1997 when it was noted at the 4 m and 9 m depth. In July 1997, *C. obovoidea* had all but disappeared and was not noted in September 1997 samples. It followed a similar trend in 1998 although concentrations were much higher. In 1999, samples were taken in mid-April rather than in May, and as a result, concentrations were about half of that counted in 1998. However, because April 1998 data were similar to that of April 1999, it is probable a bloom occurred in May 1999 leading to concentrations similar to those noted in May 1998. This bloom, which only occurs in the spring, appears to be becoming a routine event and may be a long-term change.

Three species of Chlorophyceae (green algae) were assessed. *Schroderia setigera*, *Elakatothrix gelatinosa* and *Botryococcus brauni* have been present during all the years encompassing this study (Figures 22 – 24). The latter two species were less numerous in 1999 than in previous years. They were found at all three depths, although they were most numerous at the 1 m and 4 m depths. These three species were more predominant in the summer and fall.

Only one genus of Dinophyceae was tracked. *Gymnodium sp* was noted at all three depths over the course of the study period (Figure 25) and was prevalent throughout the growing season.

6.0 Discussion

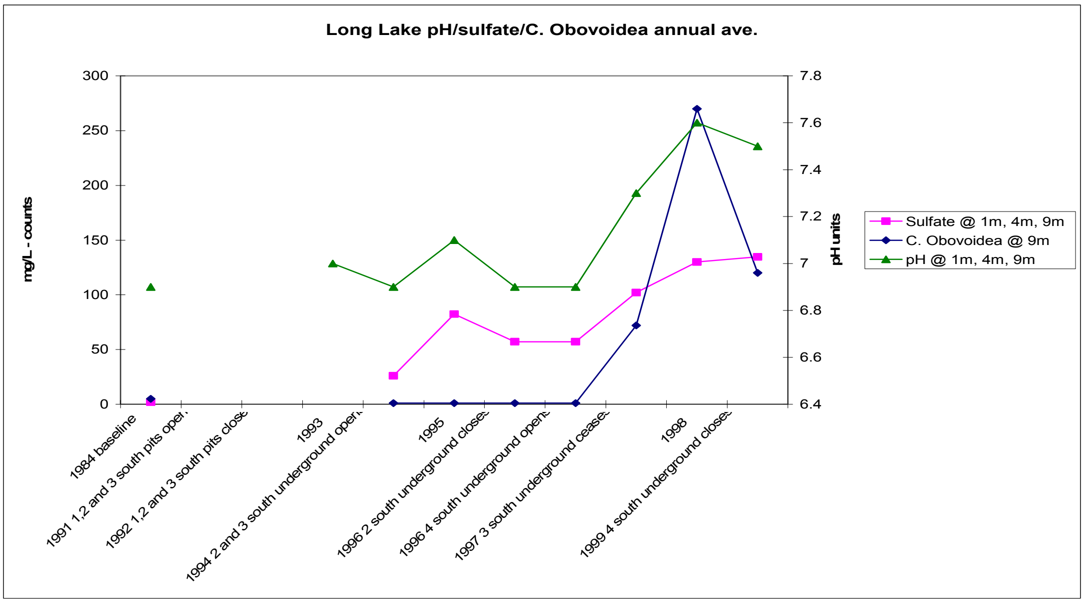
No objective for sulphate was in place at the time the mine began operating. Sulphate has since been noted to have detrimental effects on the receiving environment, particularly in soft water typically found along the coast. As a result, an ambient water quality guideline of 100 mg/L has been established (*WLAP, November 2000*).

Sulphate is a product of the oxidation of sulphur, a non-metallic element that occurs naturally in numerous minerals, including iron pyrite and gypsum (*WLAP, November 2000*). These sources of sulphate are associated with shales and other overburden disturbed during the mining process, and exposed coal seams in the open pits and underground areas. In addition, sulphate may be a byproduct of the sulphide-reducing bacteria acting on sulphide ores potentially occurring at the mine. Sulphate is commonly associated with coal mining operations and, as such, is used as an indicator of potential impacts on the receiving environment by mining activity.

Based on pre-operational monitoring data, there was very little sulphate associated with the waters of Long Lake or the other lakes and rivers in the vicinity of the mine. However, once the mine began operating, sulphate concentrations increased consistently and, in the case of Long Lake, dramatically. Figure 4 indicates that sulphate concentrations appear to be linked to mine development activity. Concentrations in Long Lake were well above background levels in 1994, the first year that sulphate data for the lake was available and three years after open pit operations began at south pits one, two and three. The initial development of the No. 2 south pit underground operation also occurred in 1994.

The only known source of sulphate at that point in time was the settling pond discharge near the outlet end of the lake.

Figure 4. pH/Sulphate/C. Obovoidea vs. Mining Activity



The 1995 data showed sulphate concentrations at least doubling at all depths, and as previously noted, the sulphate concentrations were higher at the two shallower depths than at 9 m. This is likely due to surface water discharges from the newly constructed mine workings associated with the No. 2 south mine that were not yet adequately controlled and diverted to the collection sumps. The sulphate concentration at 9 m, at this time, was still relatively low as sulphate contaminated groundwater from the underground workings may have not yet begun discharging to the lake.

Figure 4 shows that in 1996, the average sulphate concentration decreased slightly coinciding with the closure of No. 2 south underground and the opening of No. 4 south underground. In 1997, average sulphate concentrations began increasing, even though No. 3 south underground closed, and continued through 1999 when No. 4 south underground closed. The major influence in this increase was the rise in sulphate concentrations at the 9m depth, which peaked at 200 mg/L in June 1999. Concentrations at the two shallower depths were more variable and were consistently lower than at 9 m. The disparity in concentrations at various depths is due to specific characteristics of the lake, as discussed below.

Since May 1997, concentrations at 9 m have continually remained above the ambient water quality guideline of 100 mg/L during the summer months (Figure 2). It should be noted that concentrations at the 1 m and 4 m depths also exceeded the guideline in September 1997 and remained above 100 mg/L for successive Septembers thereafter although they stayed at or below the guideline during the months of May and July during this period.

The increasing sulphate concentrations in Long Lake could not be attributed to the settling pond discharge alone, and it was long suspected that there was another source. As noted in section five, this was confirmed by the discovery of a groundwater discharge to the lake that is hydraulically connected to the working of the No. 2 south underground operation. Sampling results indicated sulphate concentrations in this discharge were consistently in the 600 mg/L range. This discharge, and possibly others similar to it, appear to be responsible for the elevated sulphate concentrations in Long Lake. It is logical to assume that groundwater is continuously discharging sulphate to Long Lake. WLAP staff and mine personnel were able to sample the discharge during the months of April through August, the latter months typically being the driest of the year when groundwater flows would be expected to be the lowest.

It is likely, although not confirmed, that sulphate concentrations will be similar at all three depths during the winter months, especially after lake turnover in the late fall. MacIsaac and Stockner's findings of isothermal conditions in the lake developing in late fall support this hypothesis (*MacIsaac and Stockner 1985*). This is, however, not the case during the spring, summer and fall periods when sulphate concentrations are much higher at the 9 m depth. The main reason for this difference is thermal stratification and the lack of hypolimnion flushing. A thermocline typically sets up in April and lasts until late September or early October, resulting in minimal vertical lake circulation. The data also shows that the 9 m depth is below the thermocline, while the 1 m and 4 m depths are above it.

Continuous flushing of the epilimnion occurs during the period the thermocline is in place, especially during the spring and early summer when melt water from the winter snow pack is still in the system. This flushing decreases the rate at which the sulphate concentrations increase at the 1 m and 4 m depths although they are still significantly elevated by the fall. MacIsaac and Stockner confirm this trend in their 1985 report. A third factor that influences increased sulphate concentrations at depth is that sulphate laden water from groundwater sources is colder and denser than lake water thus sinks to deeper water where it stays because of the thermocline and poor flushing of the bottom waters.

There is one other factor influencing the sulphate concentrations below the thermocline. Sulphur, in the form of both mineral and organic sulphates, is an integral part of the biology of freshwater lakes as it is an essential plant nutrient. The lake has a sulphur cycle that uses a small percentage of the available sulphur. During the fall period, after the vast majority of the phytoplankton and zooplankton die and sink to the lake bottom, bacteria break down the biomass releasing sulphur in the form of hydrogen sulphide. The hydrogen sulphide is oxidized to sulphur which in turn is oxidized to sulphate. During the following spring and summer, sulphate is used in protein synthesis in photosynthetic and animal metabolism in which it is reduced to sulfhydryl (-SH) form (*Wetzel 1975*). Because a majority of phytoplankton live above the thermocline in the epilimnion, most of the sulphate in Long Lake is not available for metabolism and the potential for a reduction in sulphate levels at depth is not realized.

As noted in Section five, the pH in Long Lake had risen substantially from a baseline level of 7.0 to a peak of 7.8 at all three depth in July 1998. Data indicates that the pH at 9 m tended to be lower than at the 1 m and 4 m depths throughout the study period, including the period when baseline data was generated. The trend in pH levels closely parallels that of the sulphate concentrations. As Figure 4 indicates, average pH levels rose slightly in 1995 following the 1994 opening of Nos. 2 and 3 south undergrounds before decreasing to baseline levels in 1996. In 1996, No. 2 south underground closed, but No. 4 south underground opened. The development of No. 4 south underground appeared to have influence pH levels in Long Lake as the average increased significantly from 7.3 to 7.7 in 1997 before decreasing somewhat to 7.5 in 1999 when No. 4 south underground closed. The rise in pH is likely the result of soil and overburden disturbances during the various times that both surface and underground mining was developed. Soils in this area contain significant amounts of gypsum and both precipitation and groundwater are likely leaching out readily dissolvable calcareous substances associated with gypsum.

Samples taken in May and June of 2000 from the seep discharging to Long Lake confirm this as the pH averaged 8.25.

Baseline data indicate that nutrient concentrations and associated phytoplankton and zooplankton biomass in Middle Quinsam Lake and Long Lake are characteristic of oligotrophic lakes on Vancouver Island (typically total phosphorus 2 – 5 μg/L, and nitrite/nitrate <100 μg/L (*Deniseger, Erickson 1991*) (*MacIsaac and Stockner 1985*). The latter report also stated that, prior to the mine beginning operations, data indicated moderate nitrogen limitation in Long Lake. It goes on to state that there could be a potential impact on the fisheries and aquatic resources of the lake if nitrogen based explosives were used. Records indicated that nitrogen based explosives were not used at the mine site after the early 1990s.

Nutrient data from 1984 – 1999 does not show any substantive changes to the nitrate, total nitrogen or total phosphorus concentrations in the lake, other than increased nitrate levels at the 9 m depth. This suggests that the lake may still be nitrogen limiting, at least in the epilimnion when the lake is thermally stratified. It should be noted that only nitrate concentrations were used to calculate total inorganic nitrogen concentrations as nitrite and ammonia data were not included in the data set. Typically, however, they make up only a small component of total inorganic nitrogen.

Nitrate, the principle form of nitrogen found in natural waters and an integral part of the nitrogen cycle supporting aquatic life, was consistently found in the greatest concentrations during the month of May at 9 m. Nitrate concentrations at 1 m and 4 m for May, July and September were consistently at or near the detection limit of 5 μg/L. The lower concentrations are expected as light penetration at the shallower depths result in a greater degree of photosynthetic activity by the phytoplankton community. Conversely, the higher nitrate concentrations at the 9 m depth can be attributed to the fact that less photosynthetic activity is occurring as this is below both the thermocline and light extinction depth. In addition, nutrients released by the die-off of phytoplankton at the bottom of the lake are trapped below the thermocline. For example, a significant algal bloom dominated by *Ochromonas/Chromulina* spp in May 1994 and resulting die-off may have caused the elevated nitrate levels present at that time.

Typically, phosphorus concentrations were also higher at depth although the concentration/depth correlation was not as clearly defined as that for nitrates (see Figures 6 and 7). As discussed in the preceding paragraph, higher concentrations of phosphorus at depth can be attributed to a reduced level of photosynthetic activity.

Seasonal succession governs the phytoplankton abundance and species diversity in almost all lakes. A variety of natural factors influence this abundance and diversity resulting in a phytoplankton community that varies from one year to the next while typically remaining within a certain range over a much longer period of time.

Phytoplankton abundance in 1984 was much lower than in succeeding years. This is likely explained by the fact that, as previously mentioned, only the 4 m and 9 m depths were sampled in 1984. Typically, the near surface waters have the highest phytoplankton abundance, and the overall abundance would have likely been in line with succeeding years.

The only obvious change in phytoplankton abundance is a substantial increase, at all depths, between September 1998 and September 1999 (See Figure 8). The largest increases occurred in the month of September although substantial increases were also noted at 9 m in the April/May data. This increase may be the beginning of a trend although more data is required for this to be confirmed. The diversity index also increased in 1999 with the index returning to baseline levels. In the preceding years, the index had decreased from baseline levels. This decrease may be, in part, attributable to blooms that several species underwent between 1994 and 1999.

A likely explanation for the increase in overall phytoplankton abundance in the fall is the recent increase in abundance of Cryptophyte species noted in the spring at 9 m. The water at this depth is relatively nutrient-rich as the phytoplankton community had not occupied these deeper waters in any great numbers until several Cryptophyte species appeared in significant numbers in 1997. The result has been an overall increase in algal biomass and nutrient uptake. Upon dying during the late spring/summer period, the phytoplankton sink to the lake bottom where they undergo decomposition. The subsequent release of nutrients is trapped in the hypolimnion layer until lake turnover at which time the nutrients are recirculated through the relatively homogeneous waters of the lake. The now available nutrients then support an increased phytoplankton population during the fall period. A more specific discussion of the Cryptophyte family is found below.

Particular species that have increased in abundance during the fall of 1998 and/or 1999 include the Chlorophyceae species, *S. setigera* and *B. braunii*, the Cryptophyceae species *C. marssonii*, and the Chrysophyceae species *Ochromonas/Chromulina* spp, *D. cylindricum* and *C. parva*. The latter two species underwent the greatest increase.

The most obvious change to the abundance and diversity of the phytoplankton is associated with the Cryptophyte, *C. obovoidea* (Figure 20). It was noted during the baseline years, but its numbers were minimal. It next appeared in high numbers in the spring of 1997 and was even more abundant during April, May and June of 1998. Although samples were not taken in May and June of 1999, samples taken at the end of April of that year show *C. obovoidea* counts near those noted in April 1998 leading to the probability that the May and June concentrations were similar to those noted in 1998.

Cryptophytes are associated with deeper, darker water due to their photosynthetic pigments, which improve light absorption at depth (*Munro December 1999*). Munro also notes that cryptophyte species, such as *C. obovoidea,* are known to prefer waters that have high sulphur levels and are known to inhabit the nutrient rich water at the fringe of the metalimnion – hypolimnion area of lakes.

Sulphate concentrations at 9 m rose significantly in May 1997 to 125 mg/L from an average of 85 mg/L during 1996. The pH level also increased around this time as did the appearance of *C. obovoidea* in great abundance (see Figure 4). Sulphate concentrations, the pH level and the abundance of *C. obovoidea* all increased in the subsequent two years giving rise to the conclusion that there may be a biological connection between the three. There were no obvious changes in the other three Cryptophyte species assessed that could be considered a trend although the counts for *C. cryptomonas marssonii* were somewhat similar to those of *C. obovoidea*.

As noted in section 5.0, a number of other phytoplankton species changed in abundance over the study period. The Chrysophyte *C. parva* increased significantly in abundance in July 1994 and since then, has continued to be present in numbers well above background levels. Its numbers have spiked four times since, twice in the spring and twice in the fall. The increase parallels the rise in sulphate and pH leading to the possibility that the three are related.

The two representative blue-green algae genera *Aphanothece* sp and *Aphanocapsa* sp were prominent in the background years but were noted in any abundance since September 1996. A review of other Cyanophyceae species indicate that they were still present in the 1997 to 1999 period but not in the abundance or diversity they were previously. It may be that sulphate concentrations and/or the rise in pH is negatively impacting some species of this family. Also, there is an inverse relationship between cyanobacteria and nitrates. The slight increase in nitrates once the mine became operational may have moved the competitive advantage away from the cyanobacteria influencing these counts (*Carmichael, 2002 per com*).

Three of the four representative species of the Bacillariophyceae have changed in abundance. Both *M. italica* and *C. bodanica* have decreased in abundance since September 1996 with the latter being almost non-existent in samples taken since May 1997. Again, sulphate concentrations and/or pH levels are likely factors influencing these changes. *S. radians* was not counted in baseline studies but was counted in great abundance in May 1994 and has been present since then. There is the possibility it was included in the count of other Synedra species in 1983 and 1984, but their abundance was also very low compared to the 1994 – 1999 period. As was the case with *C. parva,* the obvious changes in Long Lake coinciding with the increase in abundance of *S. radians* was the rise in the sulphate concentrations and pH levels.

An overview of the phytoplankton community of Middle Quinsam Lake was also undertaken and the results compared with the data from Long Lake, as their characteristics are similar. As previously mentioned, both are oligotrophic lakes and are part of the same river system. It should be noted that the phytoplankton standing stock in Middle Quinsam Lake was reported to be 15 to 35 percent lower than that in Long Lake when baseline studies were done (*MacIsaac and Stockner, 1985*). This comparison indicates that most of the representative species followed the same trends during the 1983 – 1984 baseline years and the 1994 – 1999 period. The only species that showed major changes was the Cryptophyte, *C. obovoidea*. This species was counted in Middle Quinsam Lake only once during the 1997 to 1999 period, at a concentration of 1 cell/mL. During the same period, maximum concentrations in Long Lake peaked at 269 cells/mL.

Sulphate concentrations and the pH level were also compared. Sulphate levels in Middle Quinsam Lake have risen slowly since the mine began operations with the maximum concentration peaking at approximately 80 mg/L in the 1998 – 1999 period while the pH has ranged from 6.9 to 7.8 in the past four years. This appears to indicate that the Long Lake surface discharge to the eastern end of Middle Quinsam Lake, as described on Section 3, has had little influence on the sulphate levels of Middle Quinsam Lake. The sulphate concentration appears to be the most likely factor influencing the significant disparity in *C. obovoidea* between the two lakes.

It can be concluded that Long Lake remains oligotophic and nitrogen limited, at least in the epilimnion, as nutrient levels appear to have remained relatively stable. Results show that the parameters of concern prior to the mine becoming operational, namely TSS and nutrients, have not changed substantially from pre-mine concentrations and are not impacting the lake. However, sulphate concentrations in Long Lake have increased from baseline levels of 1 - 2 mg/L to concentrations approaching 200 mg/L. Accompanying the increase in sulphate is a significant rise in pH, particularly since 1997. The changes in sulphate and pH have closely paralleled the development and subsequent closures of the various surface and underground workings of the mine along the shore of Long Lake. There have also been some changes in the phytoplankton community, most notably a major increase in *C. obovoidea*. It is of further significance that levels of *C. obovoidea* did not change in Middle Quinsam Lake, where sulphate levels have not increased to the same extent as in Long Lake. Monitoring of water quality and the phytoplankton community will continue to track potential changes in both Long Lake and Middle Quinsam Lake.

REFERENCES

Carmichael, B. November 2002. Personal Communication

Clark, M.(Dr). March 2000. Personal Communication.

Deniseger, J. and L. Erickson. 1991. Trends in Water Quality of Buttle Lake and the Campbell River – Continuing Decreases in Metal Concentrations from 1987 through 1990. Ministry of Environment, Lands and Parks.

Environment Canada. 1983. A Data Report on Water Quality of the Receiving Waters of the Area Around the Proposed Quinsam Coal Development.

Environment Canada. 1984. Quinsam Coal Development - A Data Report on Receiving Water Quality.

Environment Canada. 1985. Quinsam Coal Development - A Data Report on Receiving Water Quality.

International Environmental Consultants Ltd. 1982. Aquatic Environmental Sampling Program.

Kangasniemi, B. J. 1989. Campbell River Area Middle Quinsam Lake Sub-Basin Water Quality Assessment and Objectives. Ministry of Environment, Lands and Parks.

MacIsaac, E. A. and J. G. Stockner. 1985. Current Trophic State and Potential Impacts of Coal Mine Development on Production of Middle Quinsam and Long Lakes.

Ministry of Environment, Lands and Parks. 1999. Draft Ambient Water Quality Guidelines for Sulphate.

Munro, K. 1998 and 1999 data reports

Munro, K. December 1999. Personal Communication

Sturm Environmental Services, Inc. 1996. Status Report of Acid Generating Potential, Quinsam Coal Project.

Wetzel, R. G. 1975. Limnology

ACKNOWLEDGEMENTS

Karen Munro, Munro Environmental Consulting

Lloyd Erickson, Water, Land and Air Protection, Nanaimo

Ian Sharpe, Water, Land and Air Protection, Smithers

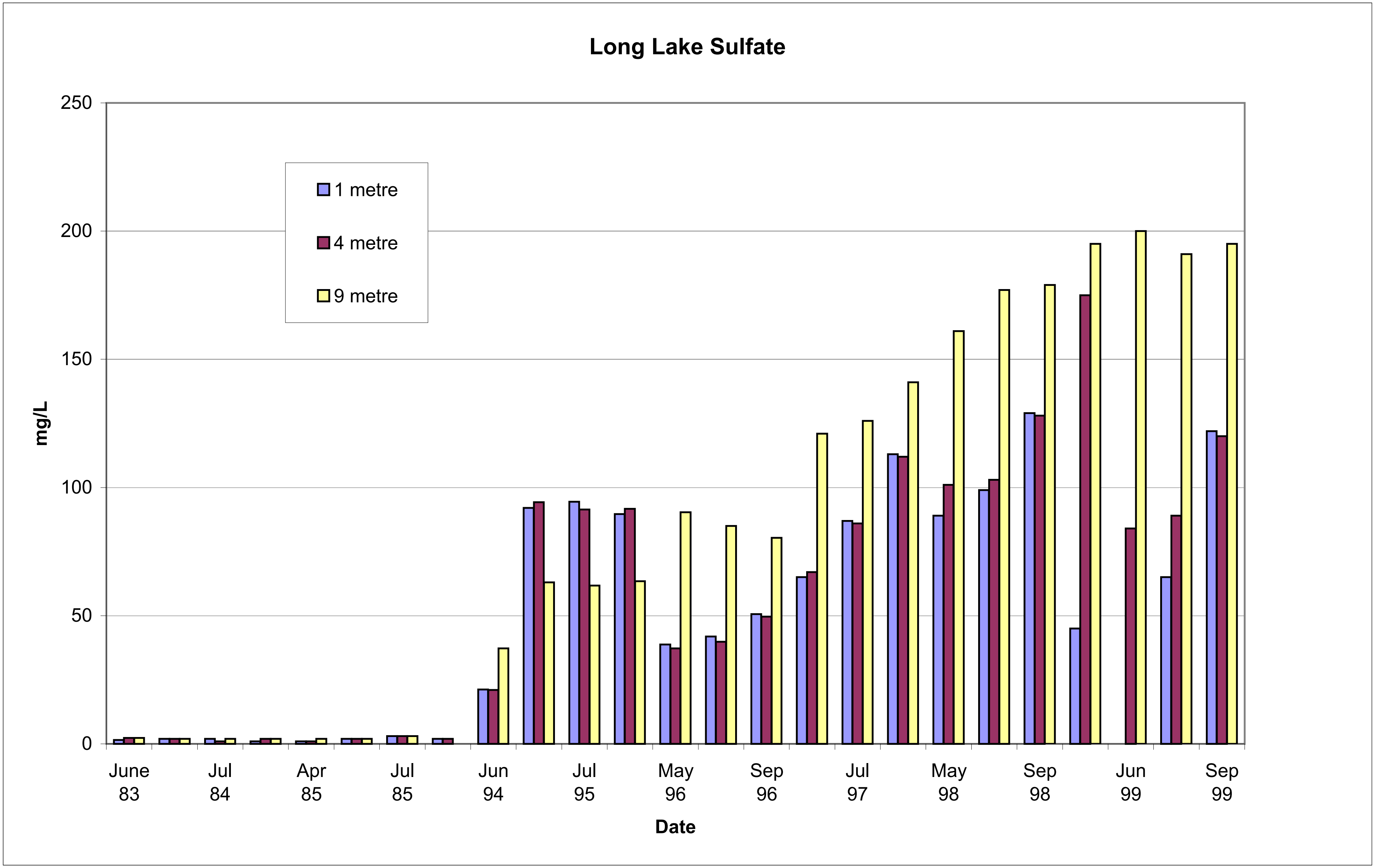
Bruce Carmichael, Water, Land and Air Protection, Prince George

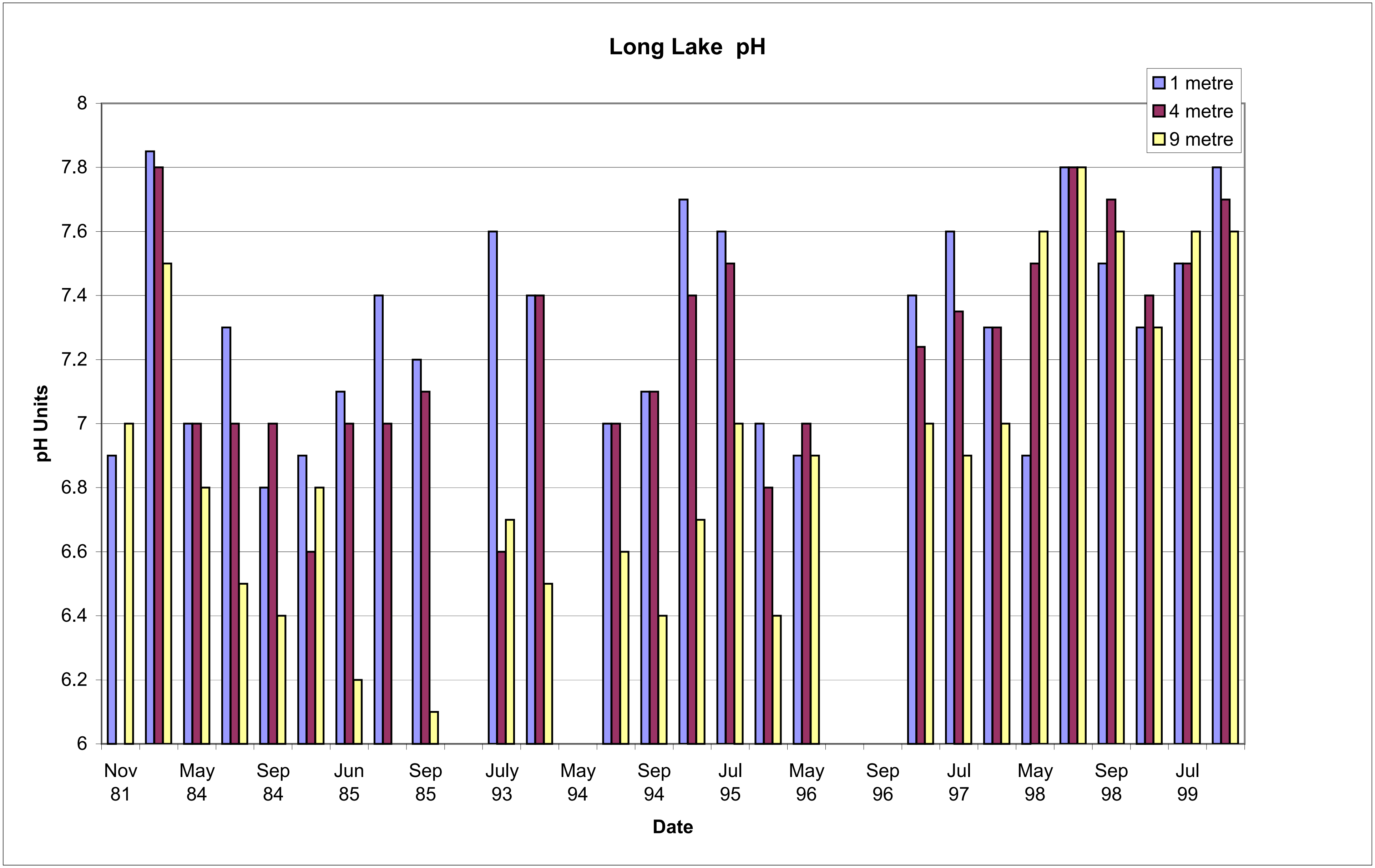
Malcolm Clark, Commonwealth Scientific Limited

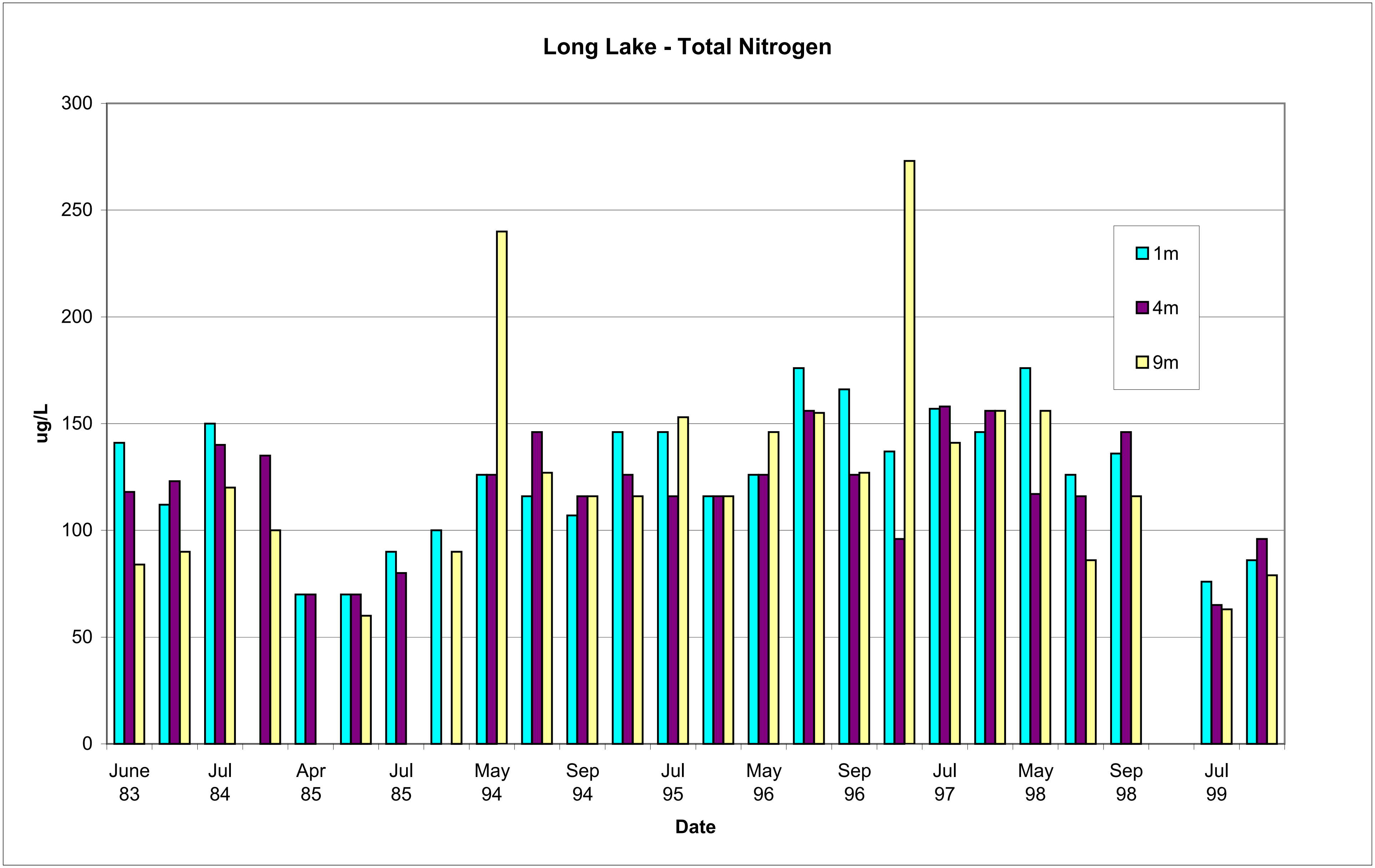
John Deniseger, Water, Land and Air Protection, Nanaimo

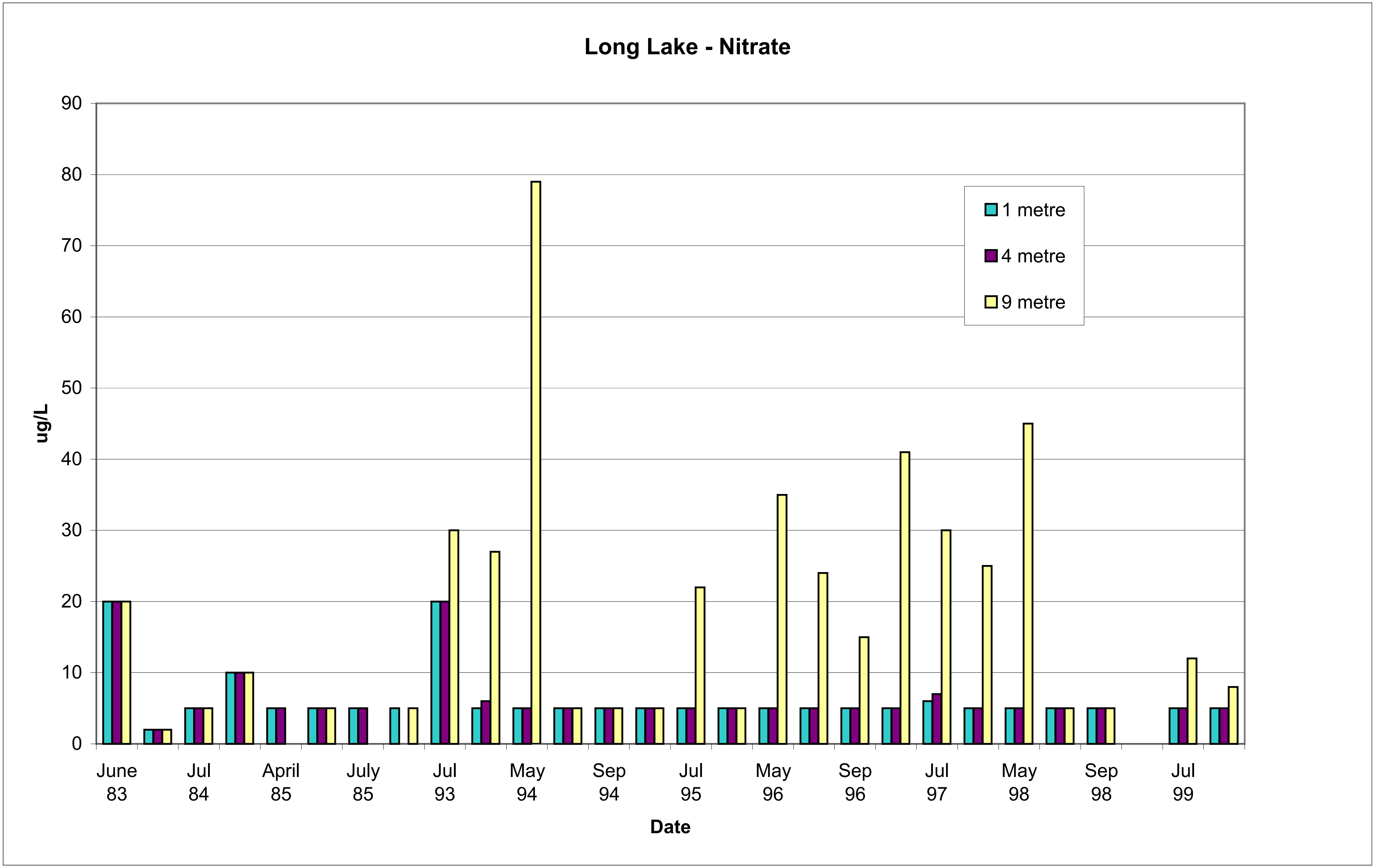
Hubert Bunce, Water, Land and Air Protection, Nanaimo

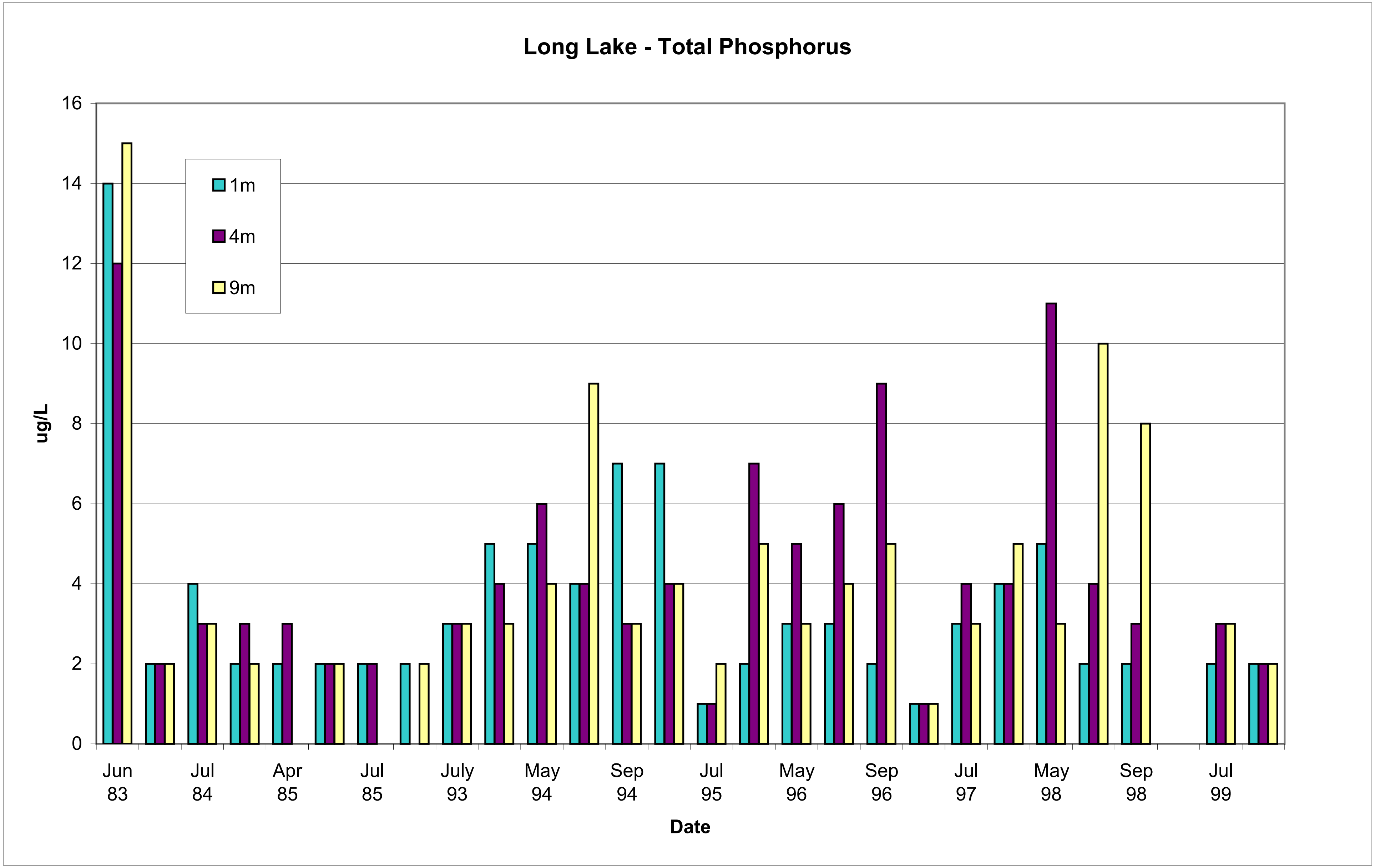
Dawn Sparks, Water, Land and Air Protection, Nanaimo

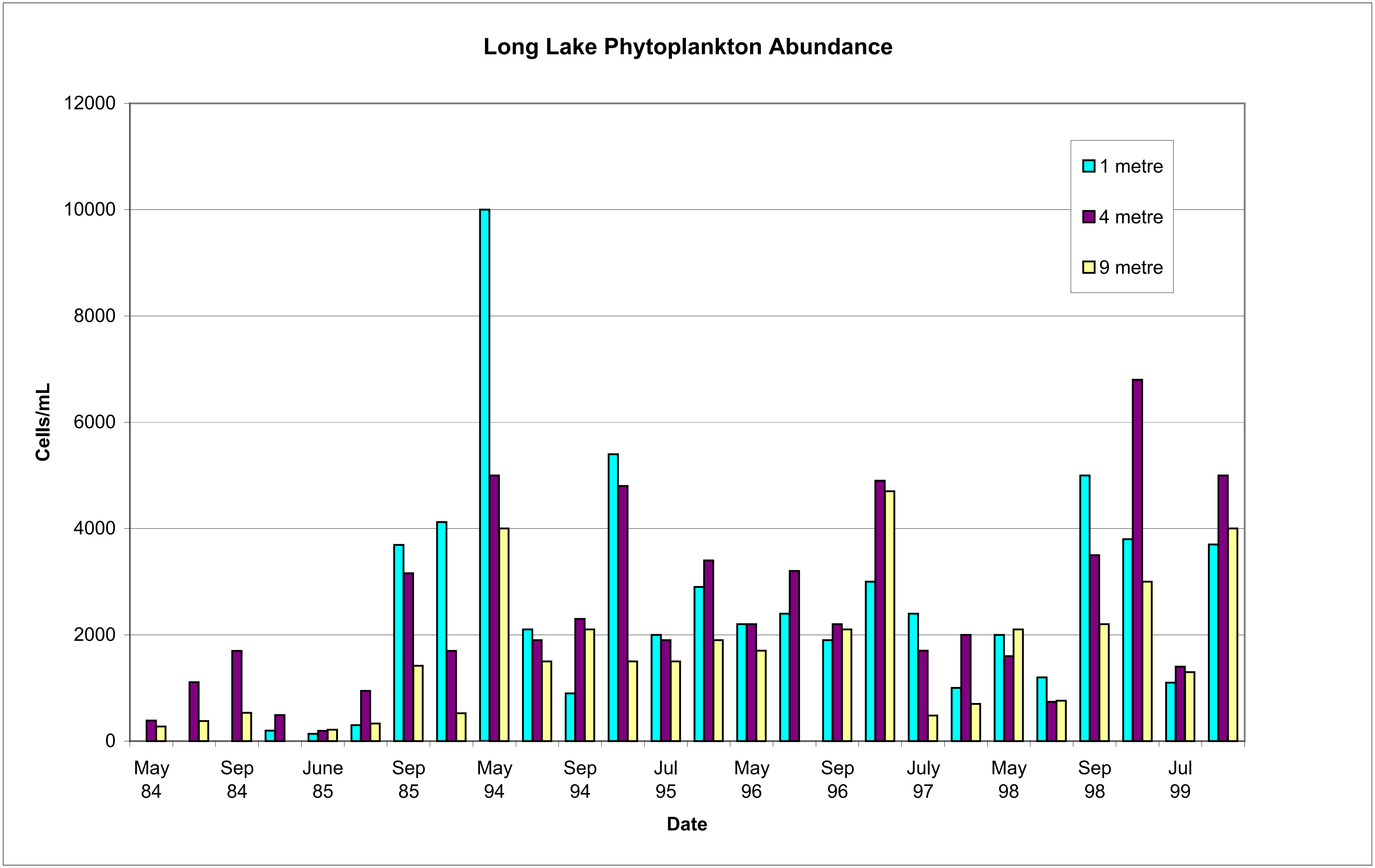

Long Lake Sulfate
1 metre
4 metre
9 metre
mg/L
250
200
150
100
50
0
June 83
Jul 84
Apr 85
Jul 85
Jun 94
Jul 95
May 96
Sep 96
Jul 97
May 98
Sep 98
Jun 99
Sep 99
Date

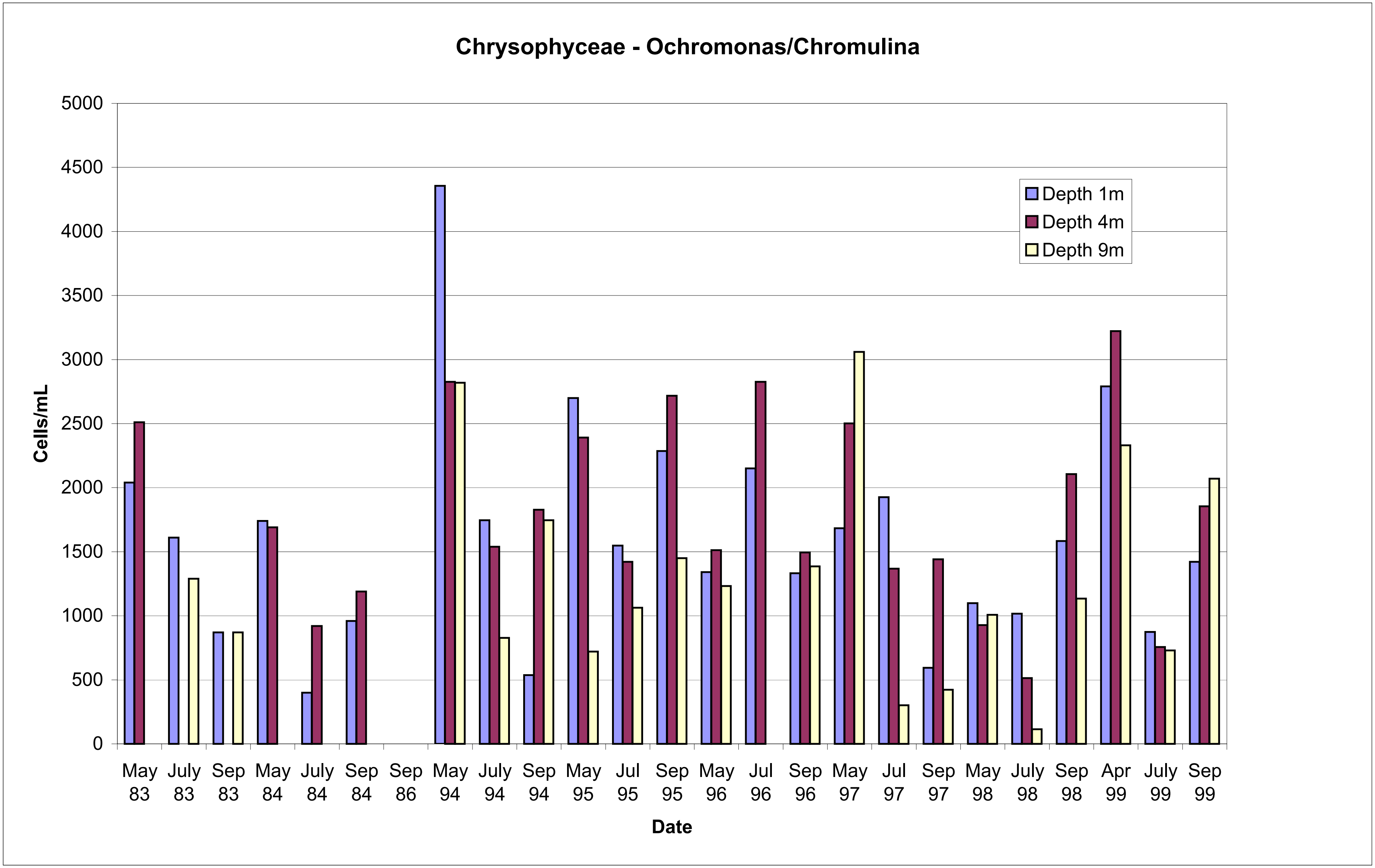

Long Lake pH
1 metre
4 metre
9 metre
pH Units
8
7.8
7.6
7.4
7.2
7
6.8
6.6
6.4
6.2
6
Nov 81
May 84
Sep 84
Jun 85
Sep 85
July 93
May 94
Sep 94
Jul 95
May 96
Sep 96
Jul 97
May 98
Sep 98
Jul 99
Date

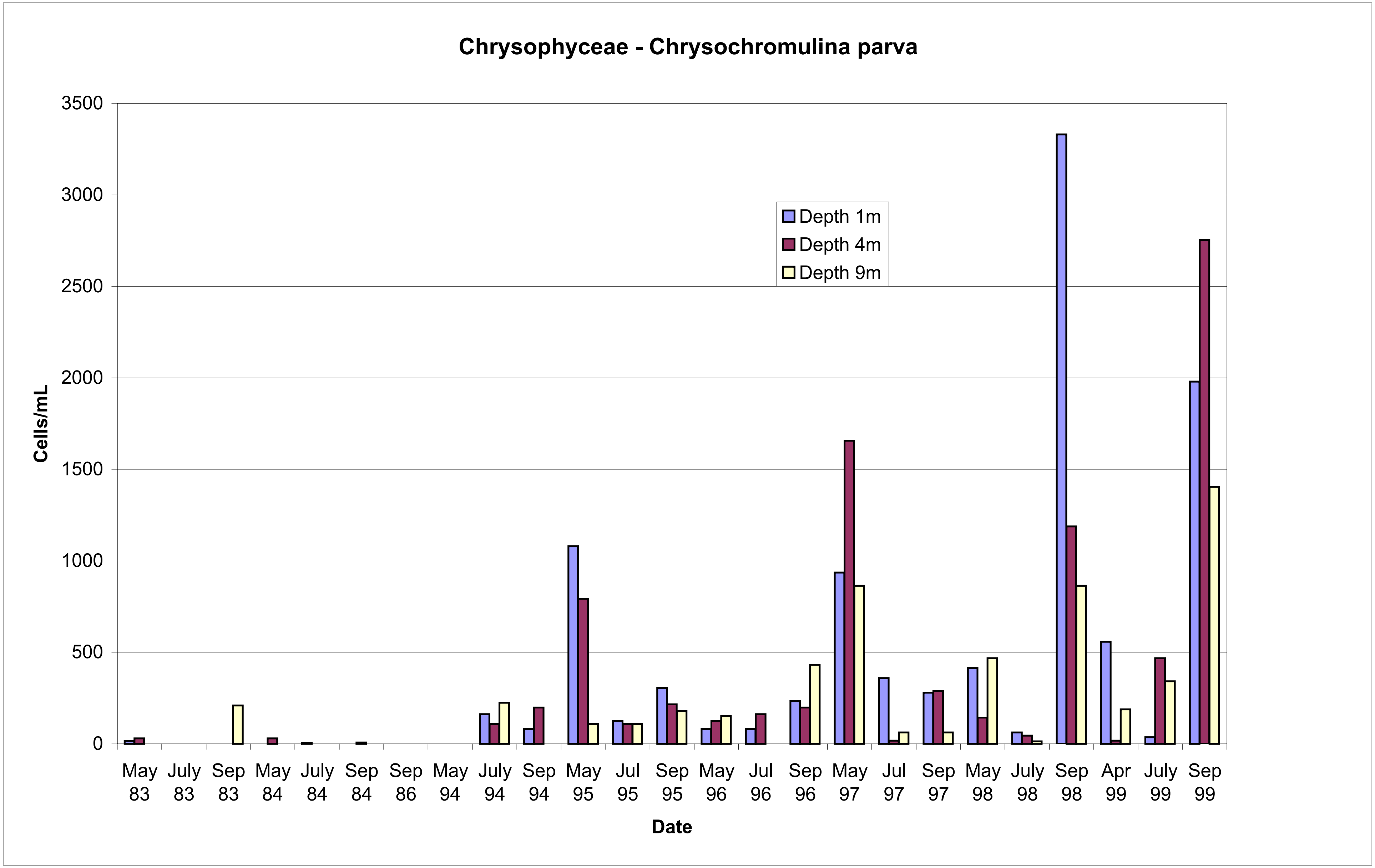

Long Lake - Total Nitrogen
300
250
200
150
100
50
0
ug/L
1m
4m
9m
June 83
Jul 84
Apr 85
Jul 85
May 94
Sep 94
Jul 95
May 96
Sep 96
Jul 97
May 98
Sep 98
Jul 99
Date

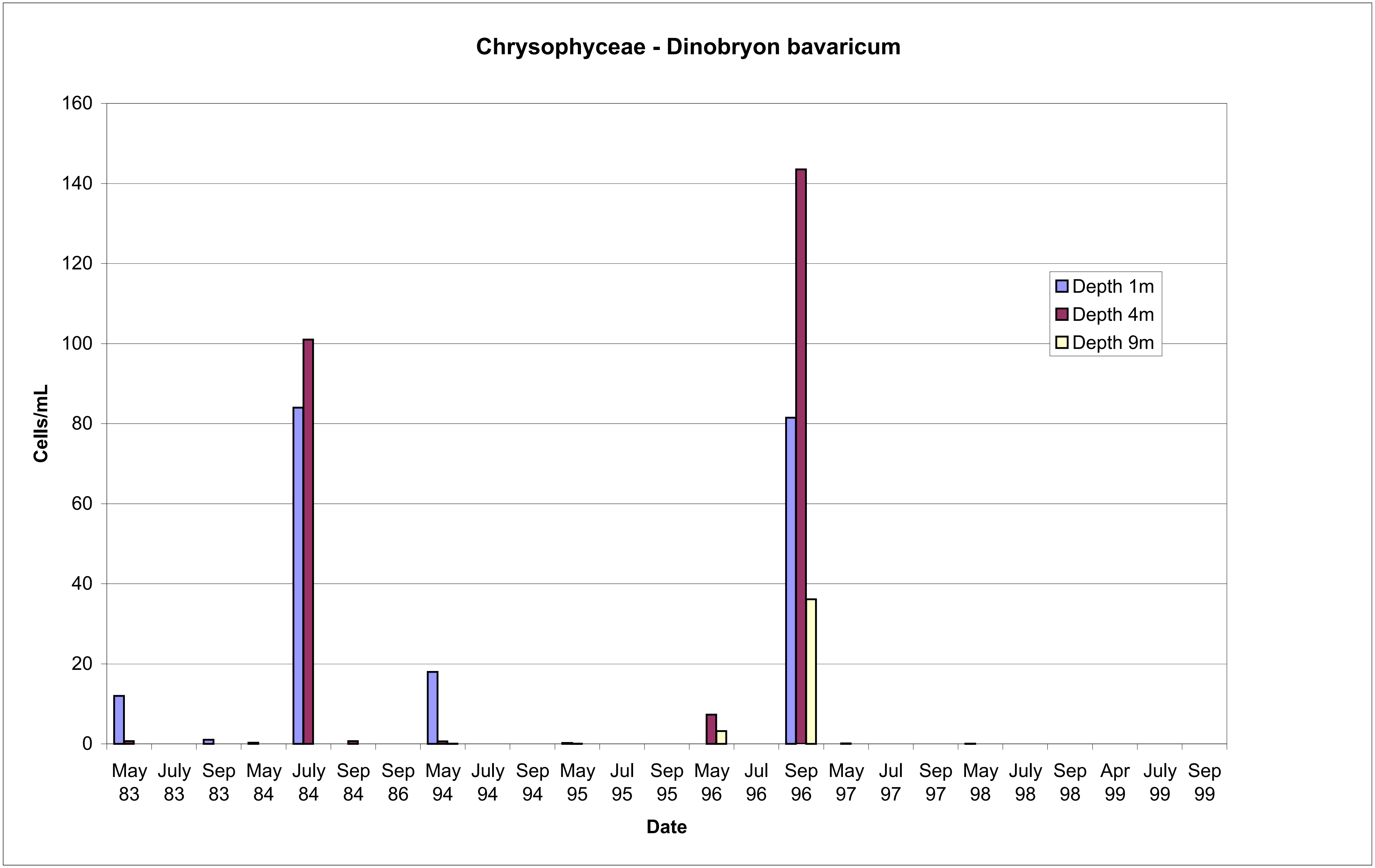

Long Lake - Nitrate
ug/L
90
80
70
60
50
40
30
20
10
0
1 metre
4 metre
9 metre
June 83
Jul 84
April 85
July 85
Jul 93
May 94
Sep 94
Jul 95
May 96
Sep 96
Jul 97
May 98
Sep 98
Jul 99
Date

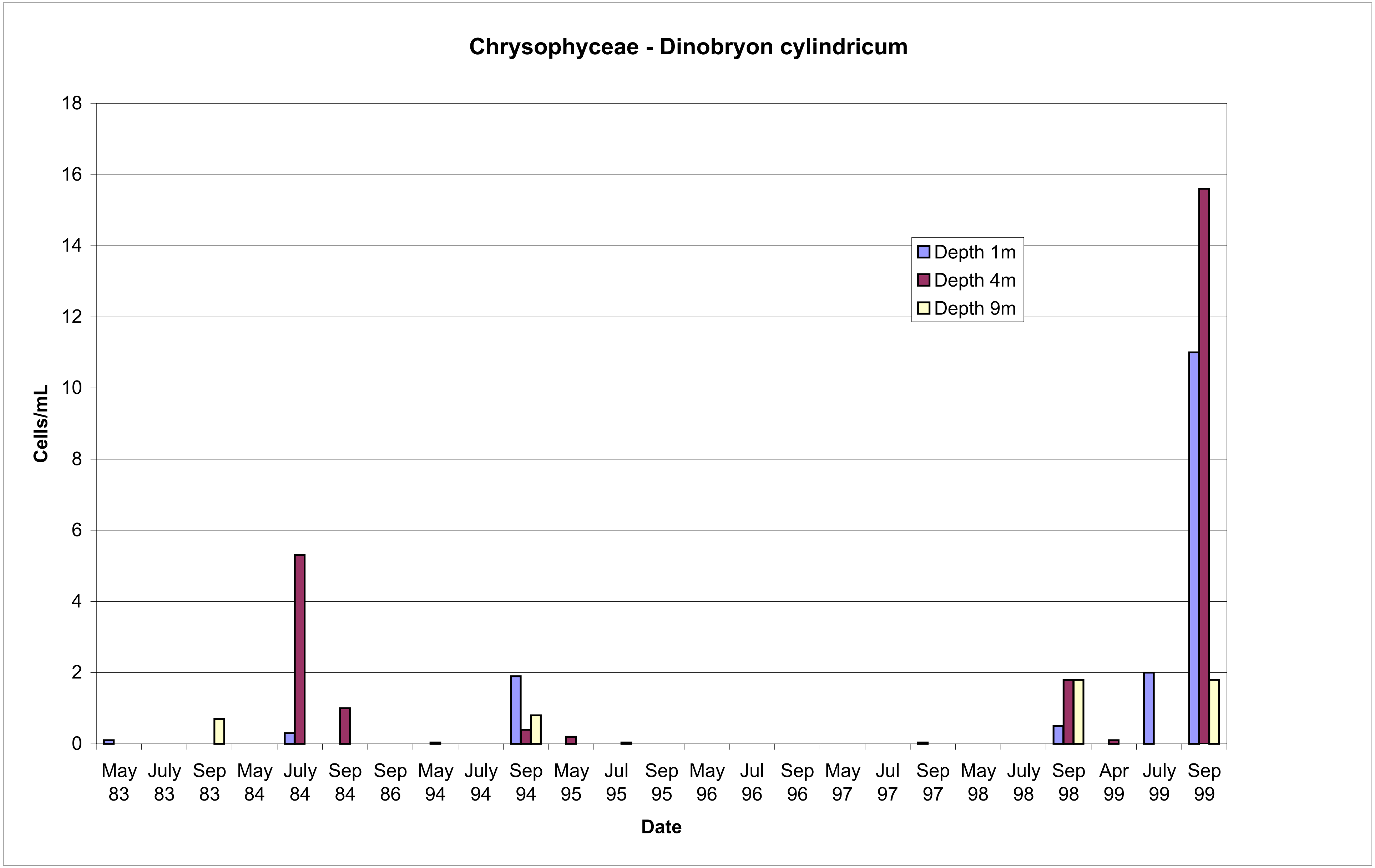

Long Lake - Total Phosphorus
1m
4m
9m
ug/L
0
2
4
6
8
10
12
14
16
Jun 83
Jul 84
Apr 85
Jul 85
July 93
May 94
Sep 94
Jul 95
May 96
Sep 96
Jul 97
May 98
Sep 98
Jul 99
Date

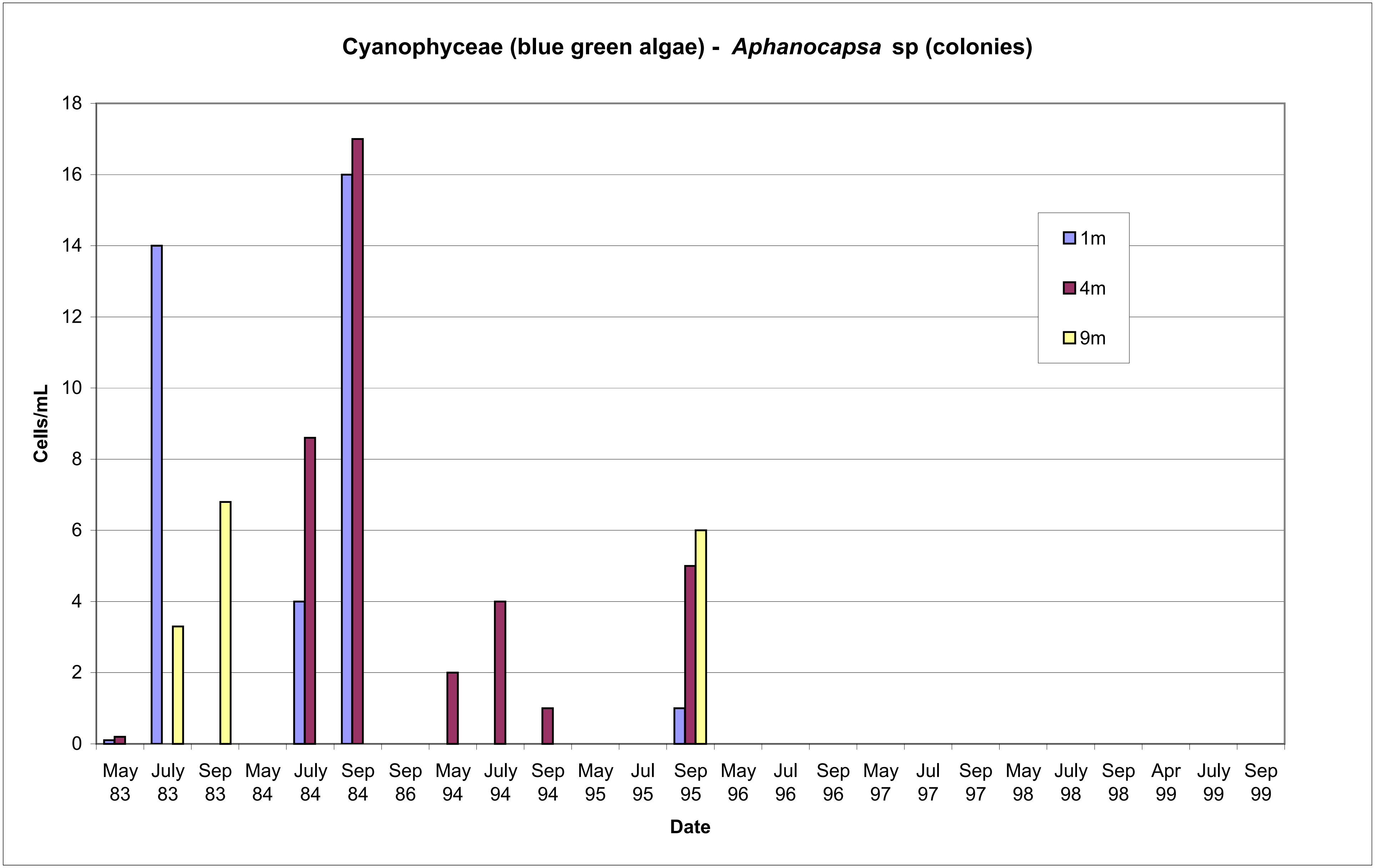

Long Lake Phytoplankton Abundance
Cells/mL
12000
10000
8000
6000
4000
2000
0
1 metre
4 metre
9 metre
May 84
Sep 84
June 85
Sep 85
May 94
Sep 94
Jul 95
May 96
Sep 96
July 97
May 98
Sep 98
Jul 99
Date

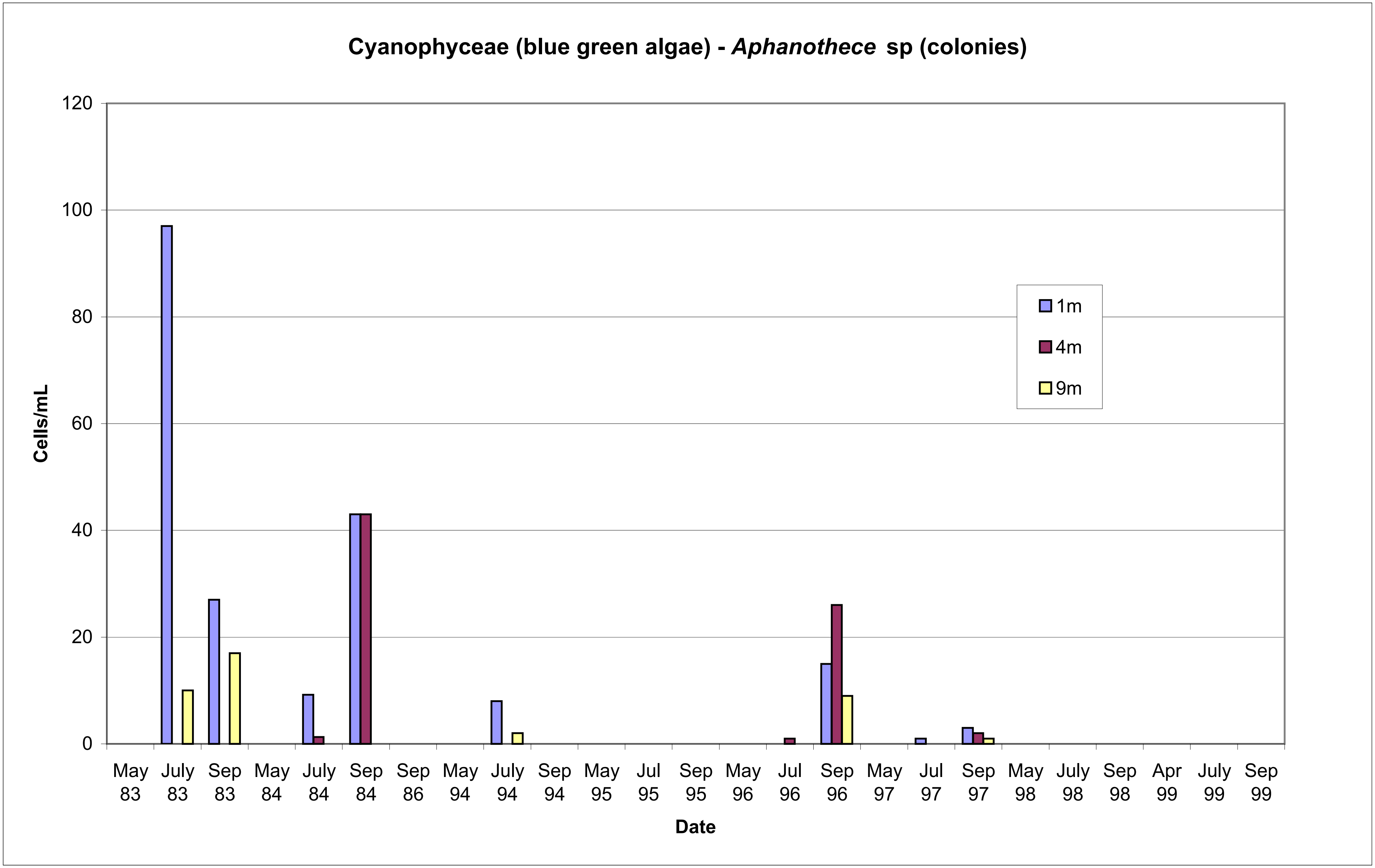

Chrysophyceae - Ochromonas/Chromulina
Depth 1m
Depth 4m
Depth 9m
Cells/mL
5000
4500
4000
3500
3000
2500
2000
1500
1000
500
0
May 83
July 83
Sep 83
May 84
July 84
Sep 84
Sep 86
May 94
July 94
Sep 94
May 95
Jul 95
Sep 95
May 96
Jul 96
Sep 96
May 97
Jul 97
Sep 97
May 98
July 98
Sep 98
Apr 99
July 99
Sep 99
Date

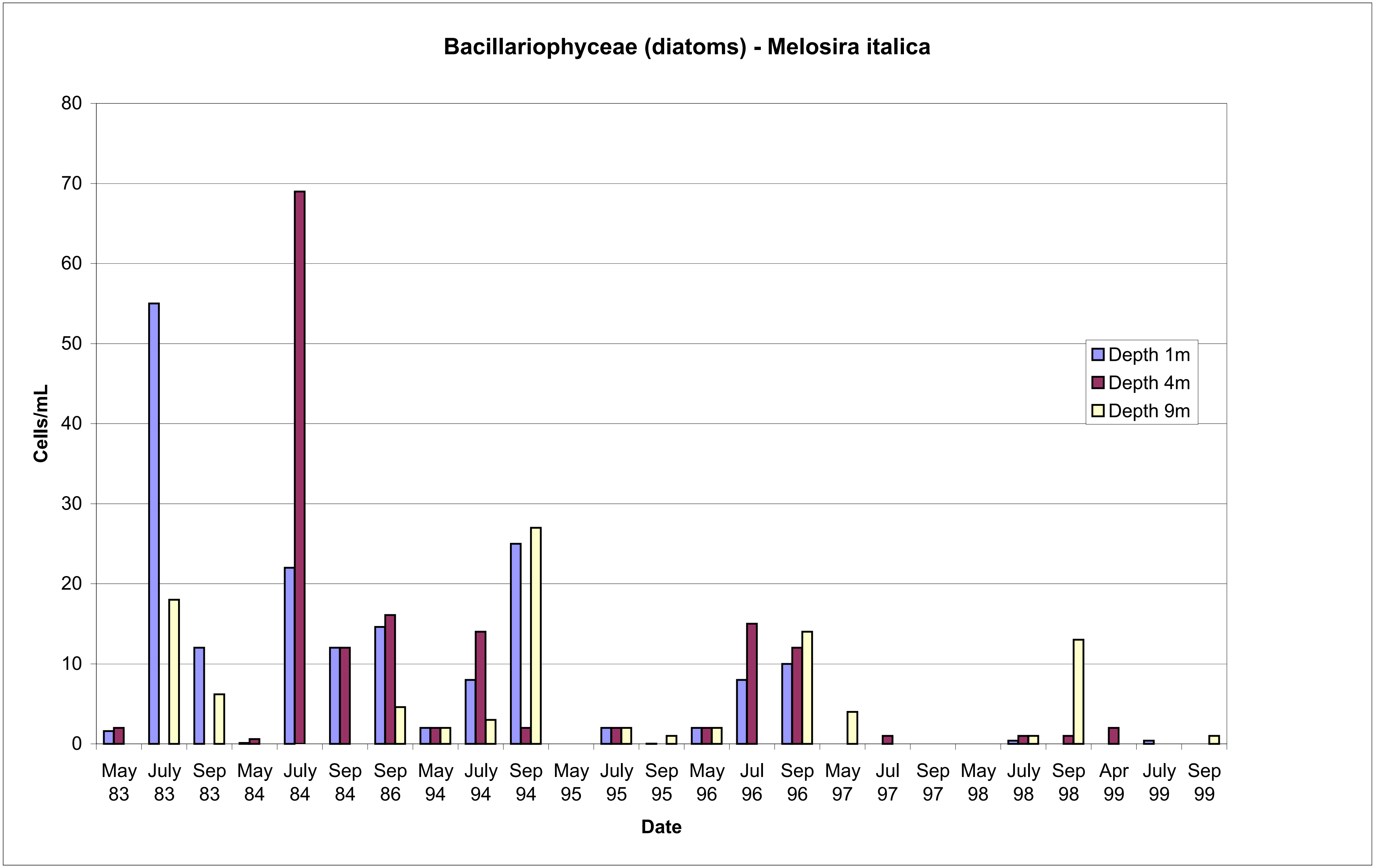

Chrysophyceae - Chrysochromulina parva
Cells/mL
3500
3000
2500
2000
1500
1000
500
0
Depth 1m
Depth 4m
Depth 9m
May 83
July 83
Sep 83
May 84
July 84
Sep 84
Sep 86
May 94
July 94
Sep 94
May 95
Jul 95
Sep 95
May 96
Jul 96
Sep 96
May 97
Jul 97
Sep 97
May 98
July 98
Sep 98
Apr 99
July 99
Sep 99
Date

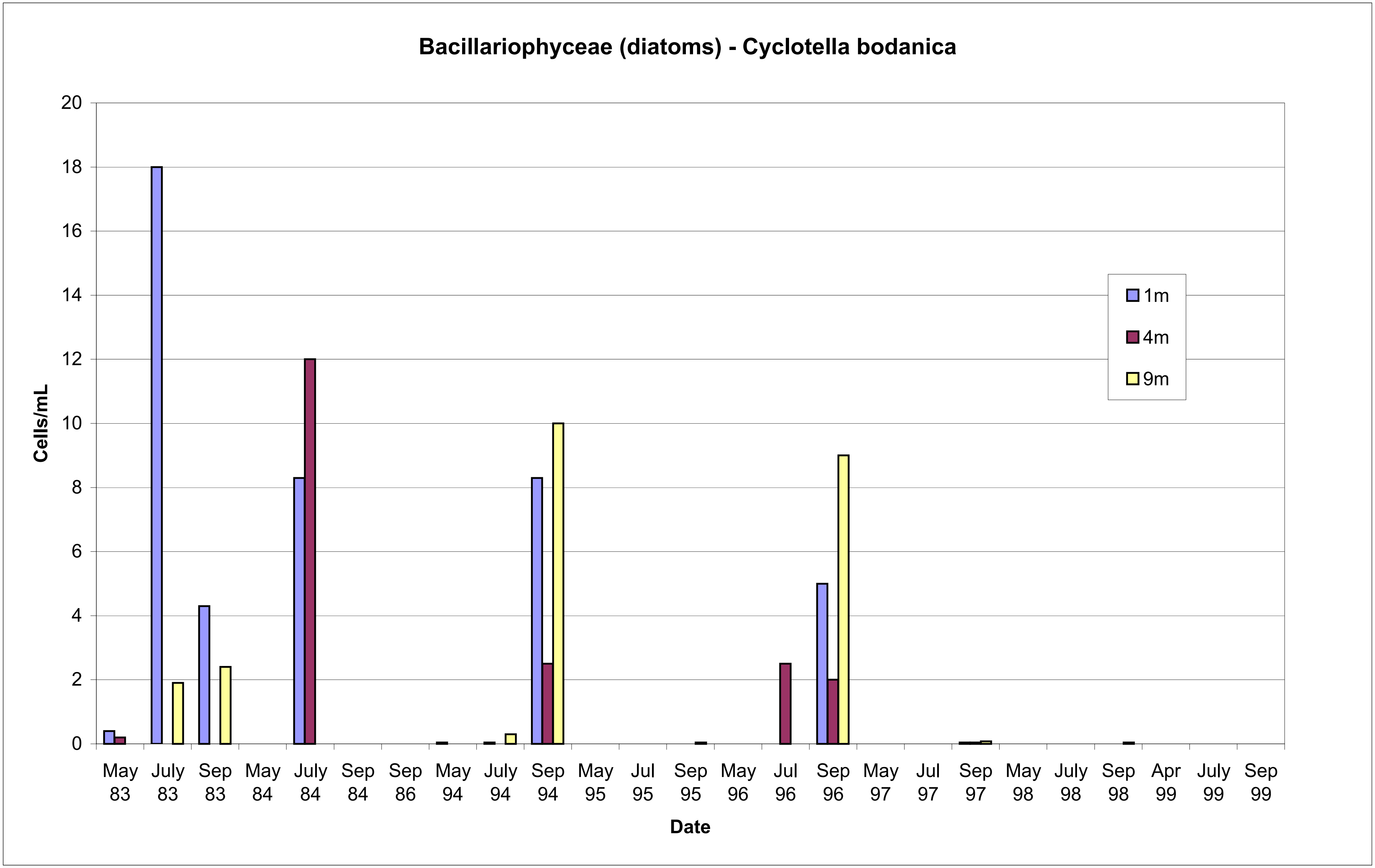

Chrysophyceae - Dinobryon bavaricum
Cells/mL
160
140
120
100
80
60
40
20
0
Depth 1m
Depth 4m
Depth 9m
May 83
July 83
Sep 83
May 84
July 84
Sep 84
Sep 86
May 94
July 94
Sep 94
May 95
Jul 95
Sep 95
May 96
Jul 96
Sep 96
May 97
Jul 97
Sep 97
May 98
July 98
Sep 98
Apr 99
July 99
Sep 99
Date

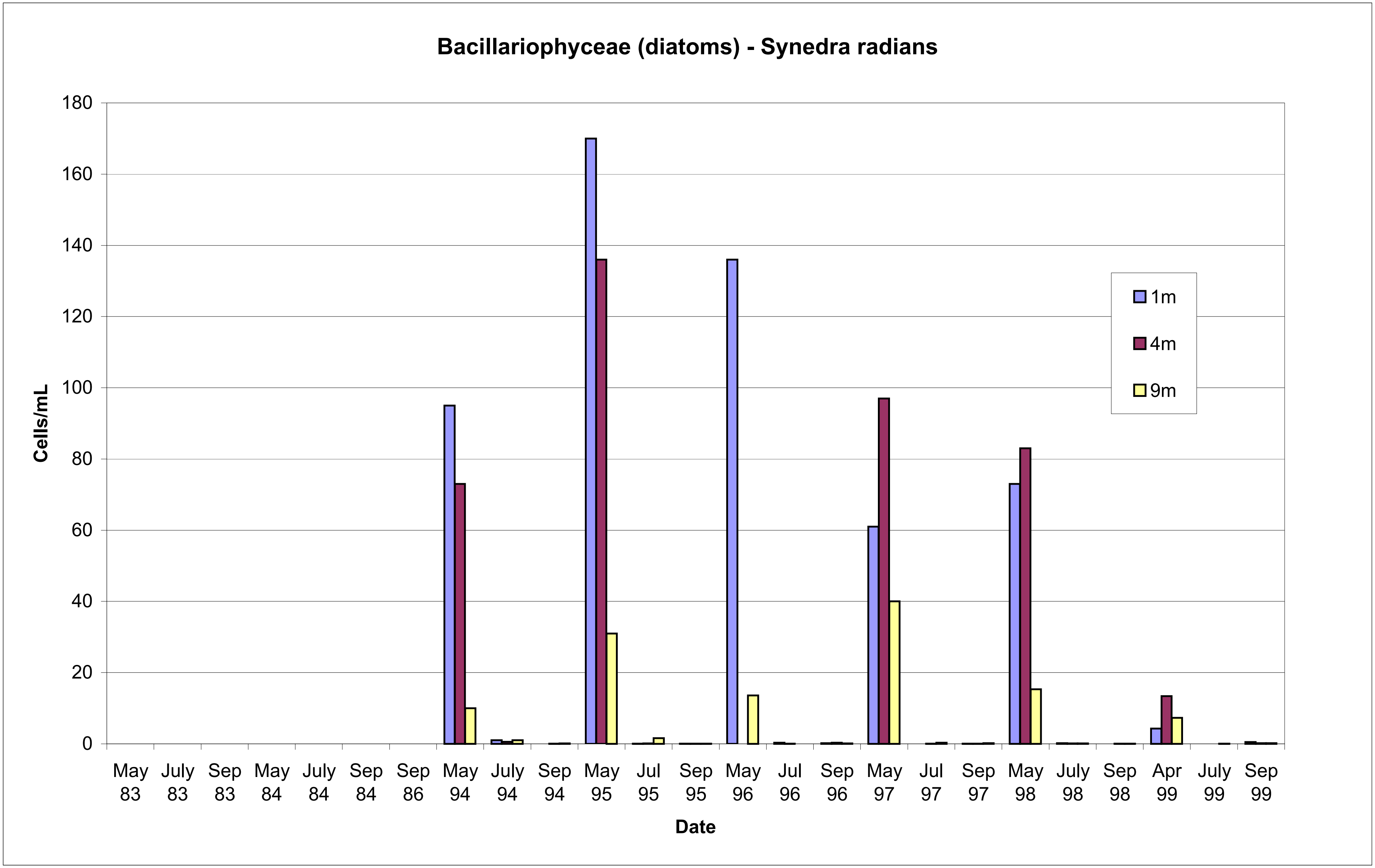

Chrysophyceae - Dinobryon cylindricum
Cells/mL
0
2
4
6
8
10
12
14
16
18
Depth 1m
Depth 4m
Depth 9m
May 83
July 83
Sep 83
May 84
July 84
Sep 84
Sep 86
May 94
July 94
Sep 94
May 95
Jul 95
Sep 95
May 96
Jul 96
Sep 96
May 97
Jul 97
Sep 97
May 98
July 98
Sep 98
Apr 99
July 99
Sep 99
Date

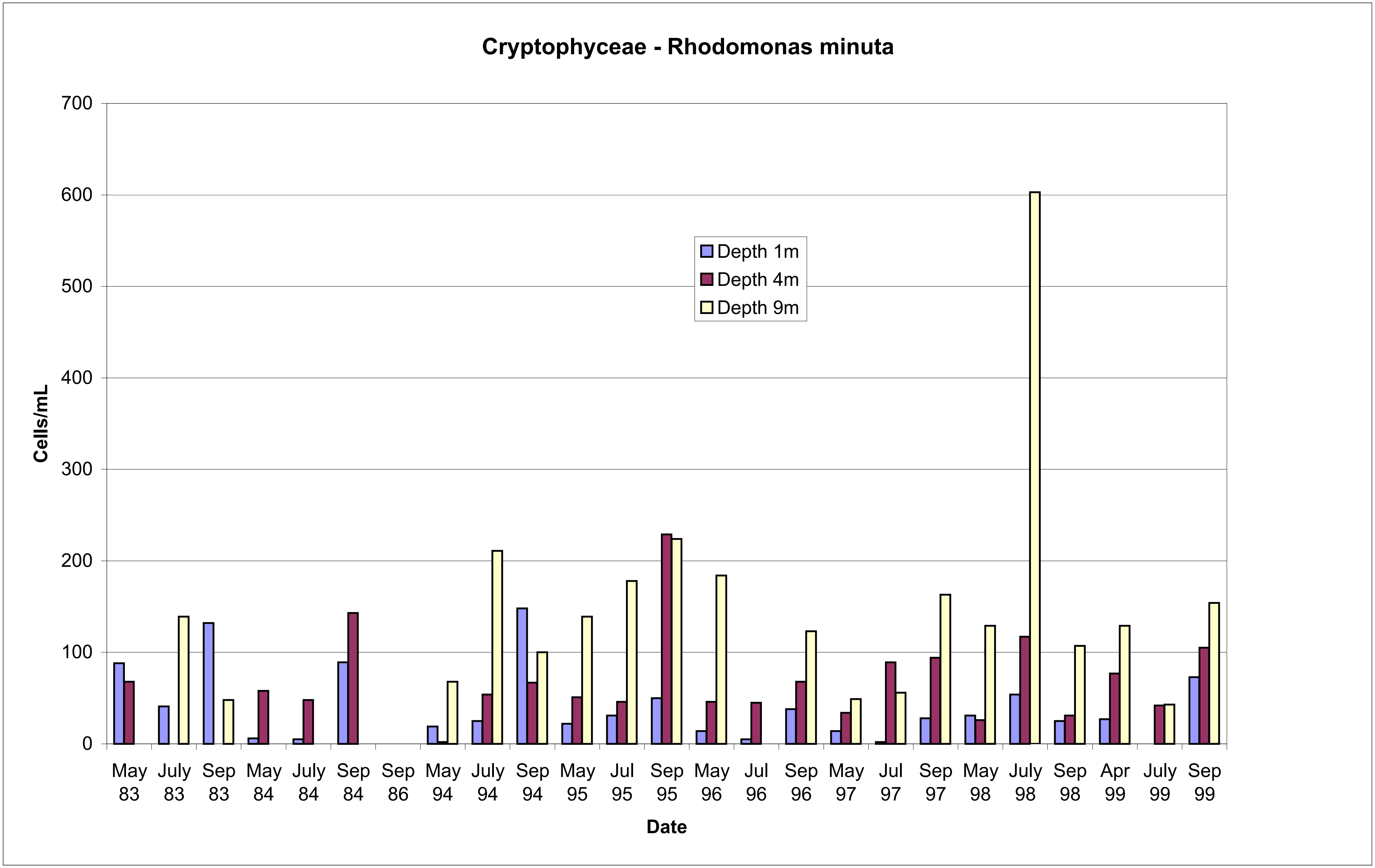

Cyanophyceae (blue green algae) - *Aphanocapsa* sp (colonies)
Cells/mL
0
2
4
6
8
10
12
14
16
18
1m
4m
9m
May 83
July 83
Sep 83
May 84
July 84
Sep 84
Sep 86
May 94
July 94
Sep 94
May 95
Jul 95
Sep 95
May 96
Jul 96
Sep 96
May 97
Jul 97
Sep 97
May 98
July 98
Sep 98
Apr 99
July 99
Sep 99
Date

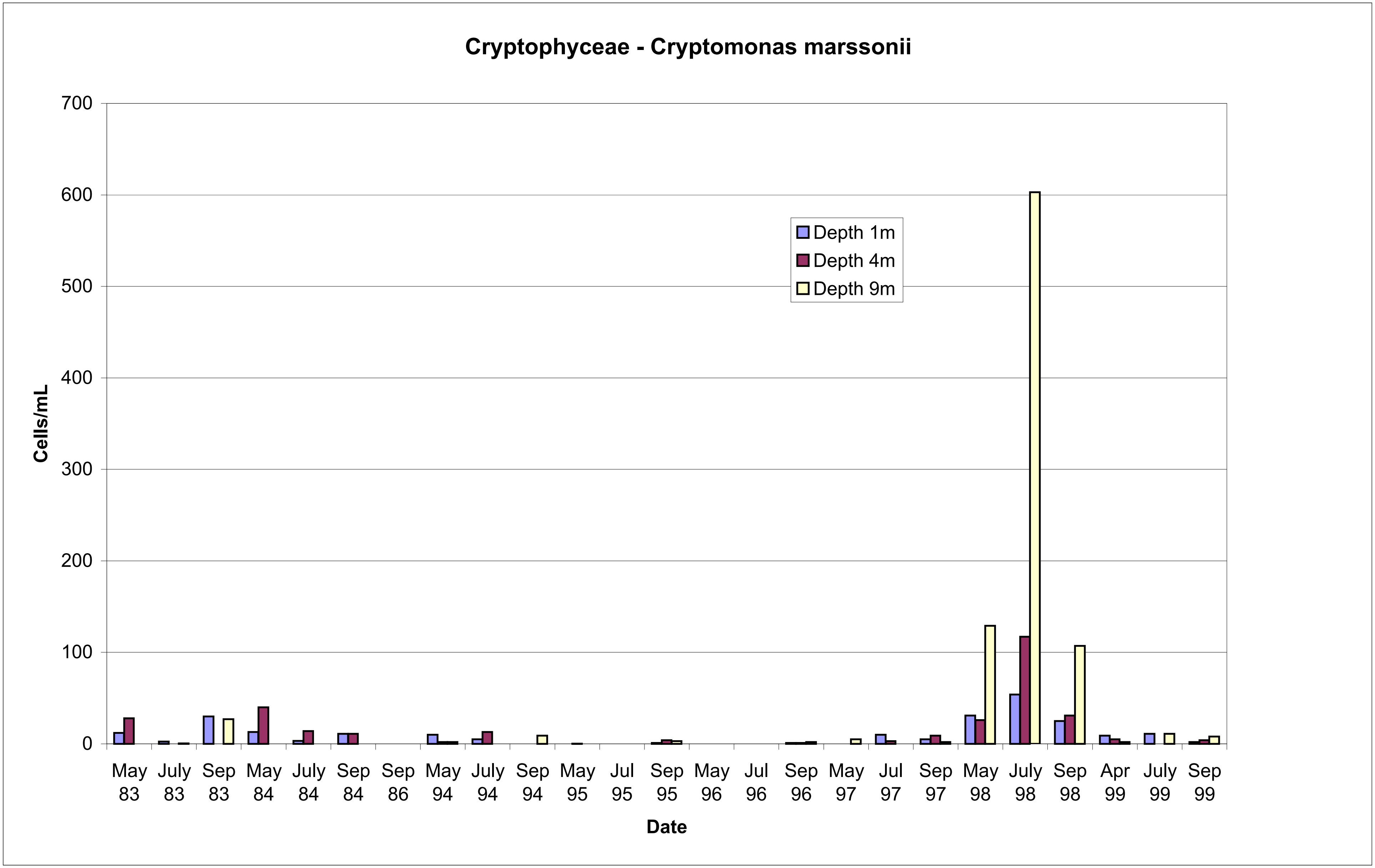

Cyanophyceae (blue green algae) - *Aphanothece* sp (colonies)
Cells/mL
120
100
80
60
40
20
0
1m
4m
9m
May 83
July 83
Sep 83
May 84
July 84
Sep 84
Sep 86
May 94
July 94
Sep 94
May 95
Jul 95
Sep 95
May 96
Jul 96
Sep 96
May 97
Jul 97
Sep 97
May 98
July 98
Sep 98
Apr 99
July 99
Sep 99
Date


Bacillariophyceae (diatoms) - Melosira italica
Cells/mL
0
10
20
30
40
50
60
70
80
Depth 1m
Depth 4m
Depth 9m
May 83
July 83
Sep 83
May 84
July 84
Sep 84
Sep 86
May 94
July 94
Sep 94
May 95
July 95
Sep 95
May 96
Jul 96
Sep 96
May 97
Jul 97
Sep 97
May 98
July 98
Sep 98
Apr 99
July 99
Sep 99
Date

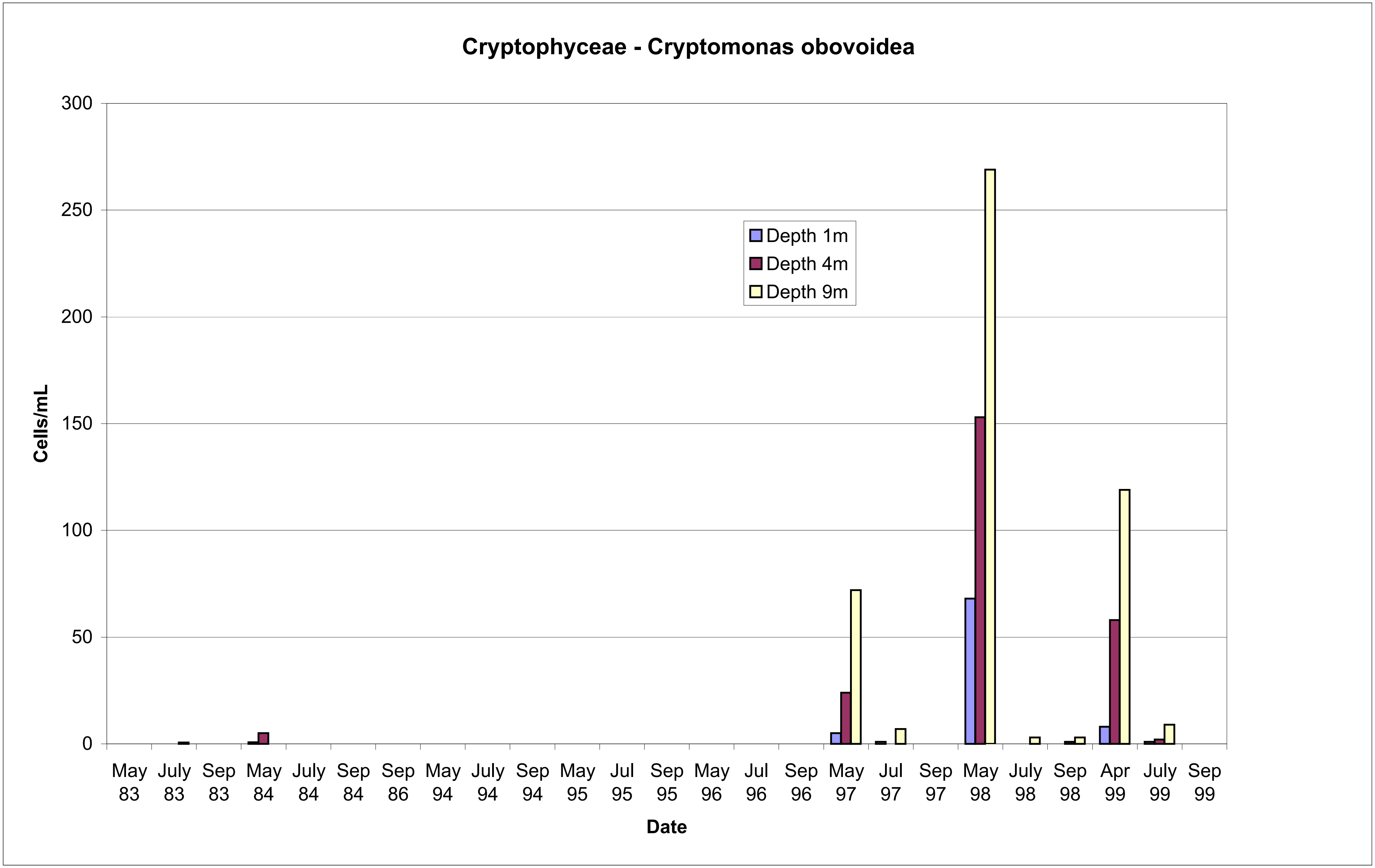

Bacillariophyceae (diatoms) - Cyclotella bodanica
Cells/mL
0
2
4
6
8
10
12
14
16
18
20
1m
4m
9m
May 83
July 83
Sep 83
May 84
July 84
Sep 84
Sep 86
May 94
July 94
Sep 94
May 95
Jul 95
Sep 95
May 96
Jul 96
Sep 96
May 97
Jul 97
Sep 97
May 98
July 98
Sep 98
Apr 99
July 99
Sep 99
Date

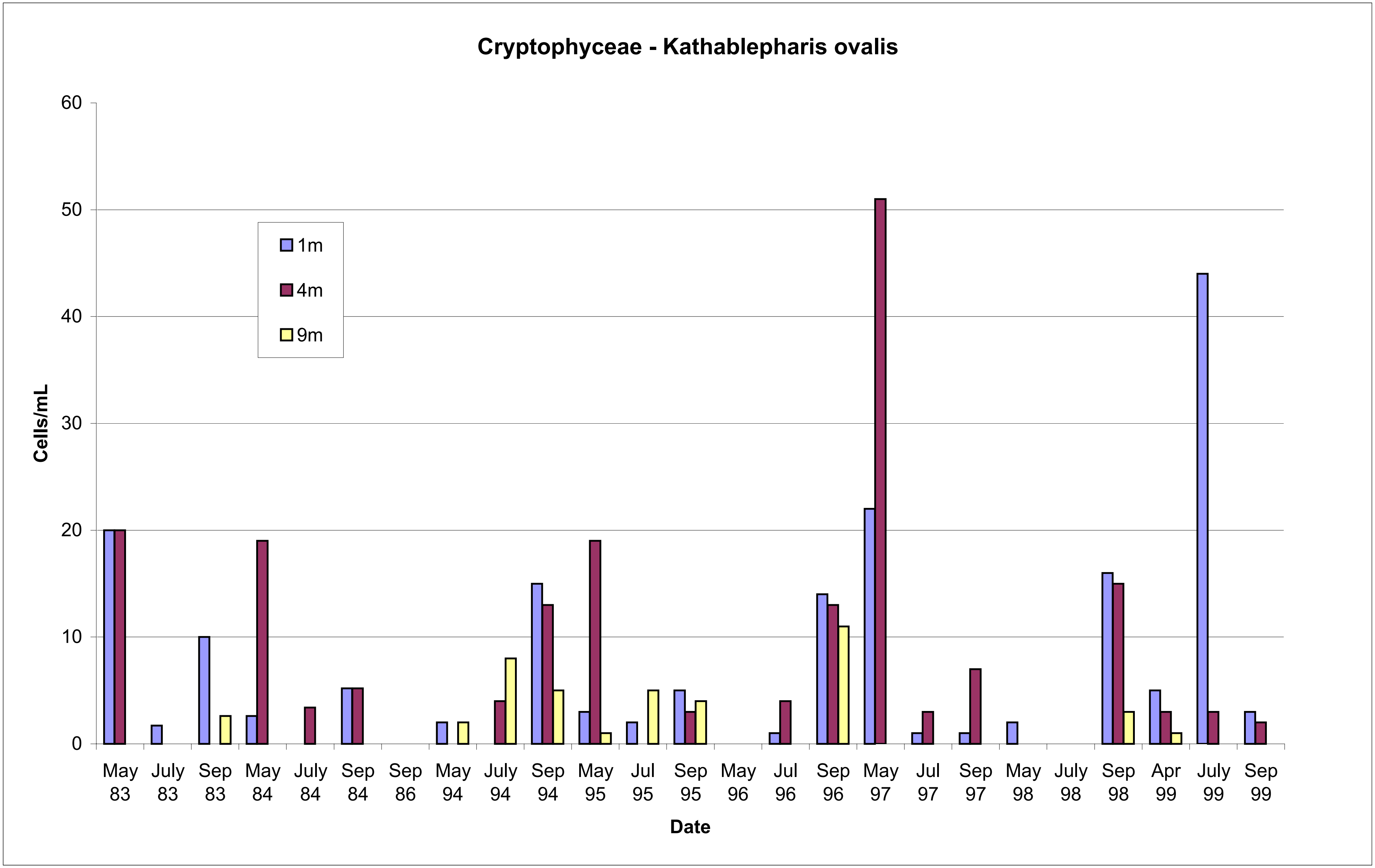

Bacillariophyceae (diatoms) - Synedra radians
Cells/mL
180
160
140
120
100
80
60
40
20
0
1m
4m
9m
May 83
July 83
Sep 83
May 84
July 84
Sep 84
Sep 86
May 94
July 94
Sep 94
May 95
Jul 95
Sep 95
May 96
Jul 96
Sep 96
May 97
Jul 97
Sep 97
May 98
July 98
Sep 98
Apr 99
July 99
Sep 99
Date

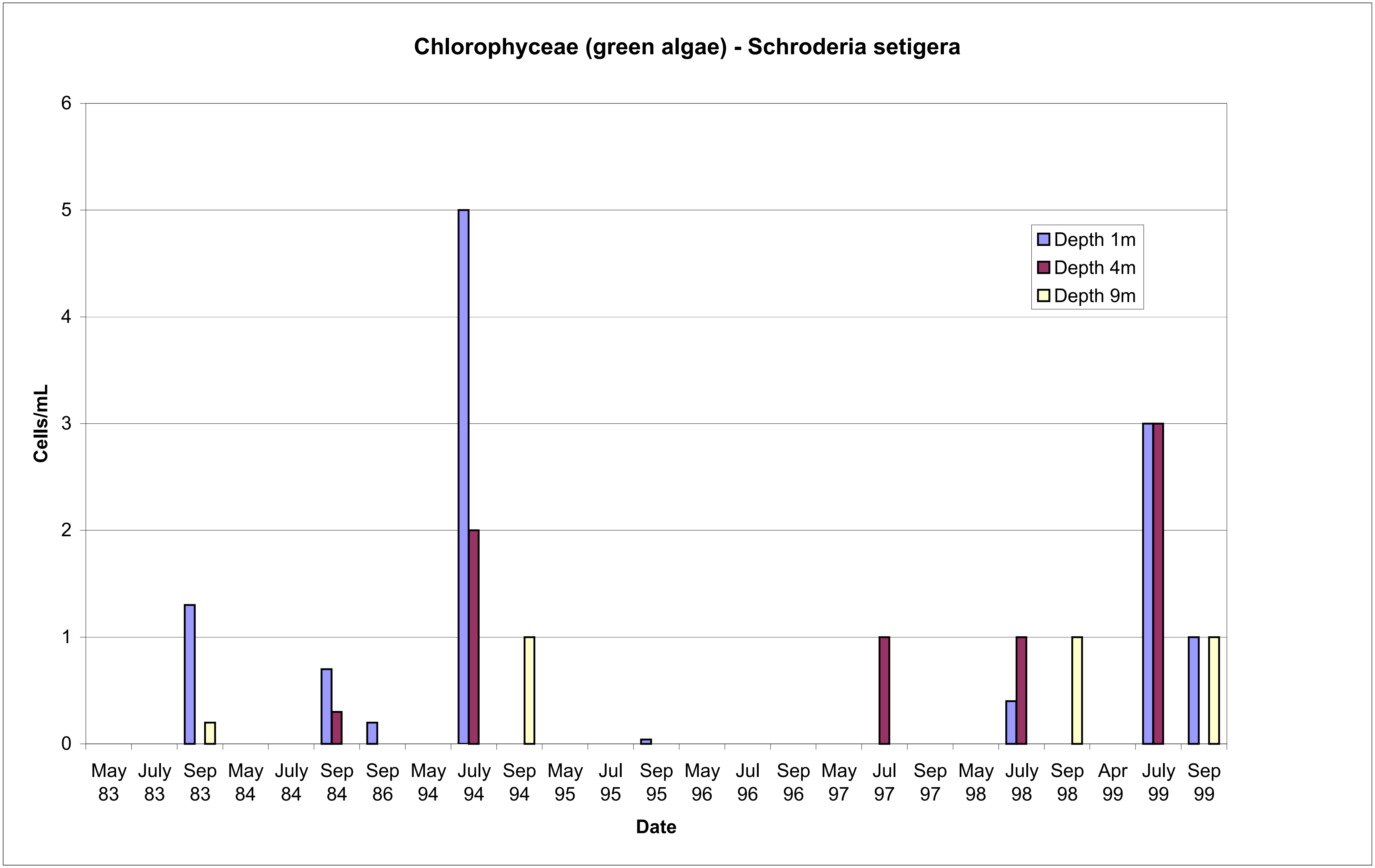

Cryptophyceae - Rhodomonas minuta
Depth 1m
Depth 4m
Depth 9m
Cells/mL
700
600
500
400
300
200
100
0
May 83
July 83
Sep 83
May 84
July 84
Sep 84
Sep 86
May 94
July 94
Sep 94
May 95
Jul 95
Sep 95
May 96
Jul 96
Sep 96
May 97
Jul 97
Sep 97
May 98
July 98
Sep 98
Apr 99
July 99
Sep 99
Date

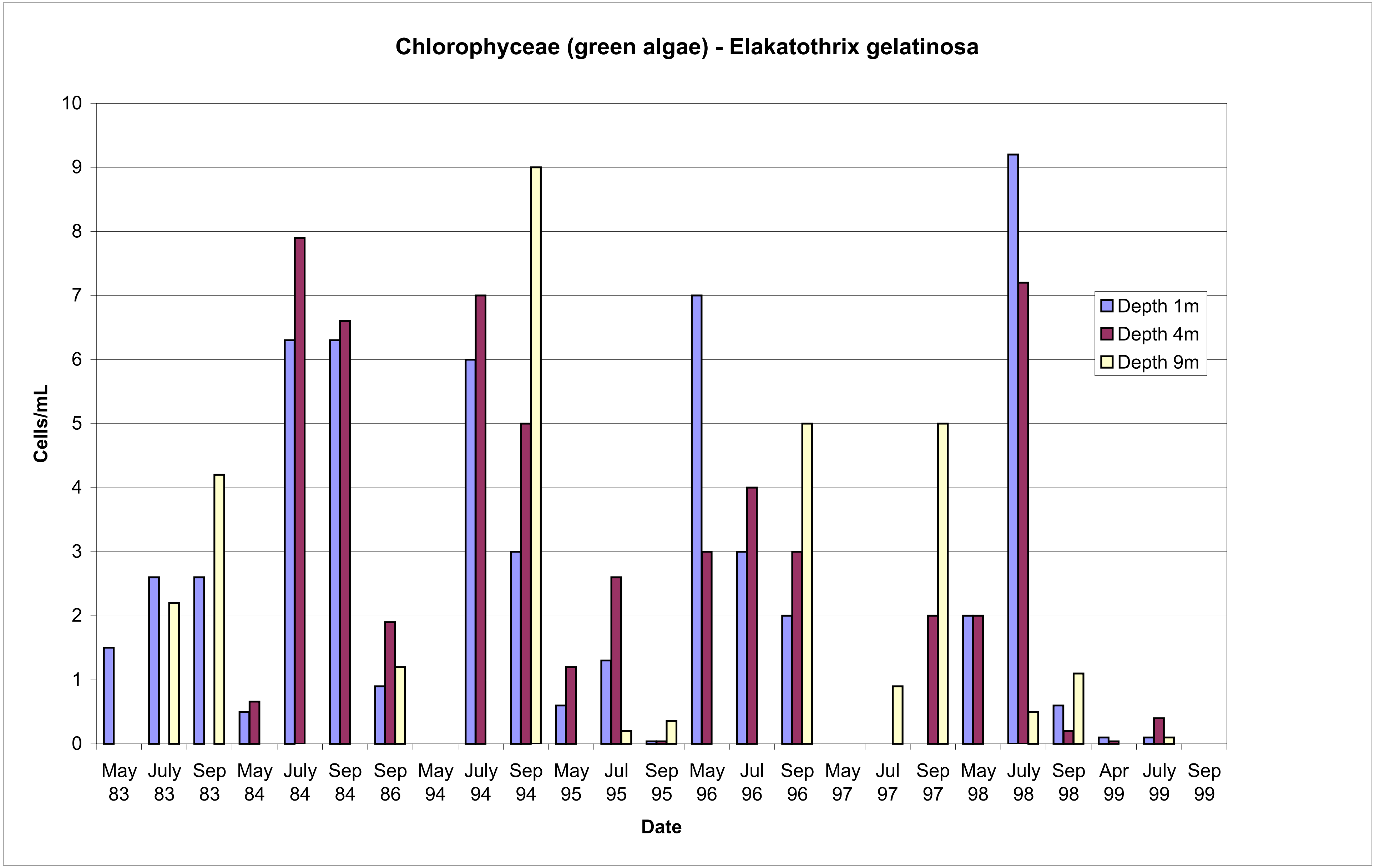


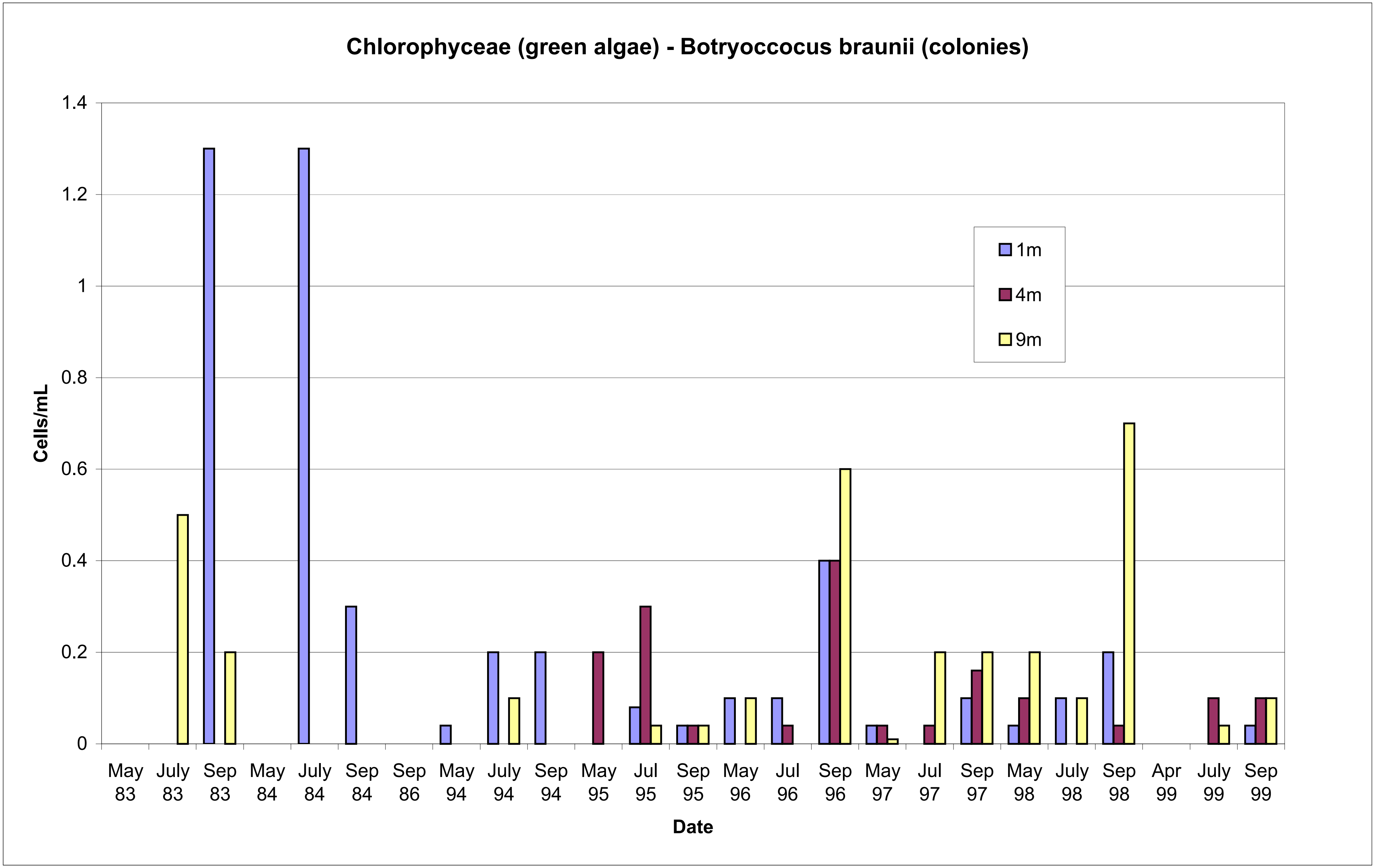
Cryptophyceae - Cryptomonas marssonii
Cells/mL
700
600
500
400
300
200
100
0
Depth 1m
Depth 4m
Depth 9m
May 83
July 83
Sep 83
May 84
July 84
Sep 84
Sep 86
May 94
July 94
Sep 94
May 95
Jul 95
Sep 95
May 96
Jul 96
Sep 96
May 97
Jul 97
Sep 97
May 98
July 98
Sep 98
Apr 99
July 99
Sep 99
Date

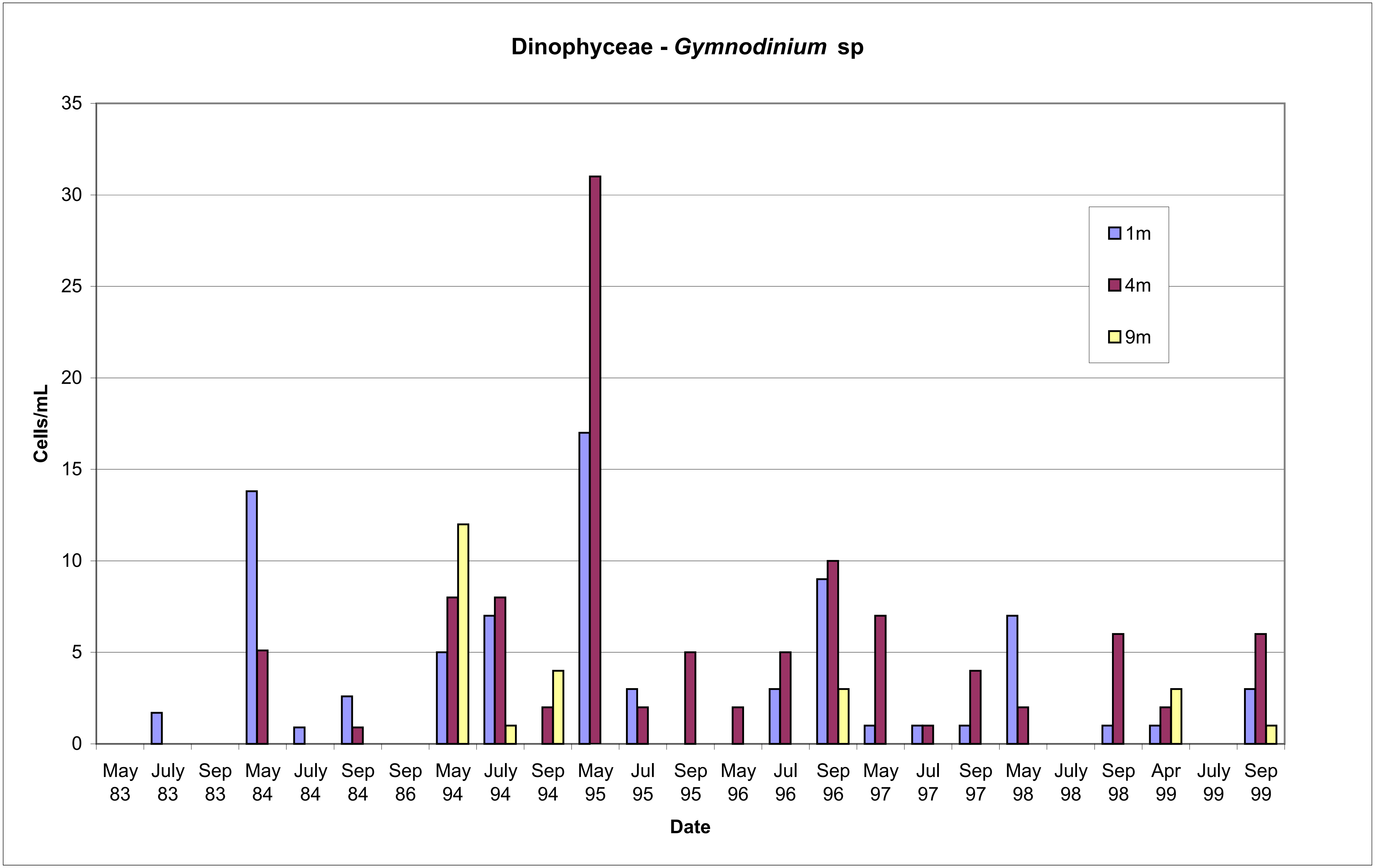

Cryptophyceae - Cryptomonas obovoidea
Depth 1m
Depth 4m
Depth 9m
Cells/mL
300
250
200
150
100
50
0
May 83
July 83
Sep 83
May 84
July 84
Sep 84
Sep 86
May 94
July 94
Sep 94
May 95
Jul 95
Sep 95
May 96
Jul 96
Sep 96
May 97
Jul 97
Sep 97
May 98
July 98
Sep 98
Apr 99
July 99
Sep 99
Date

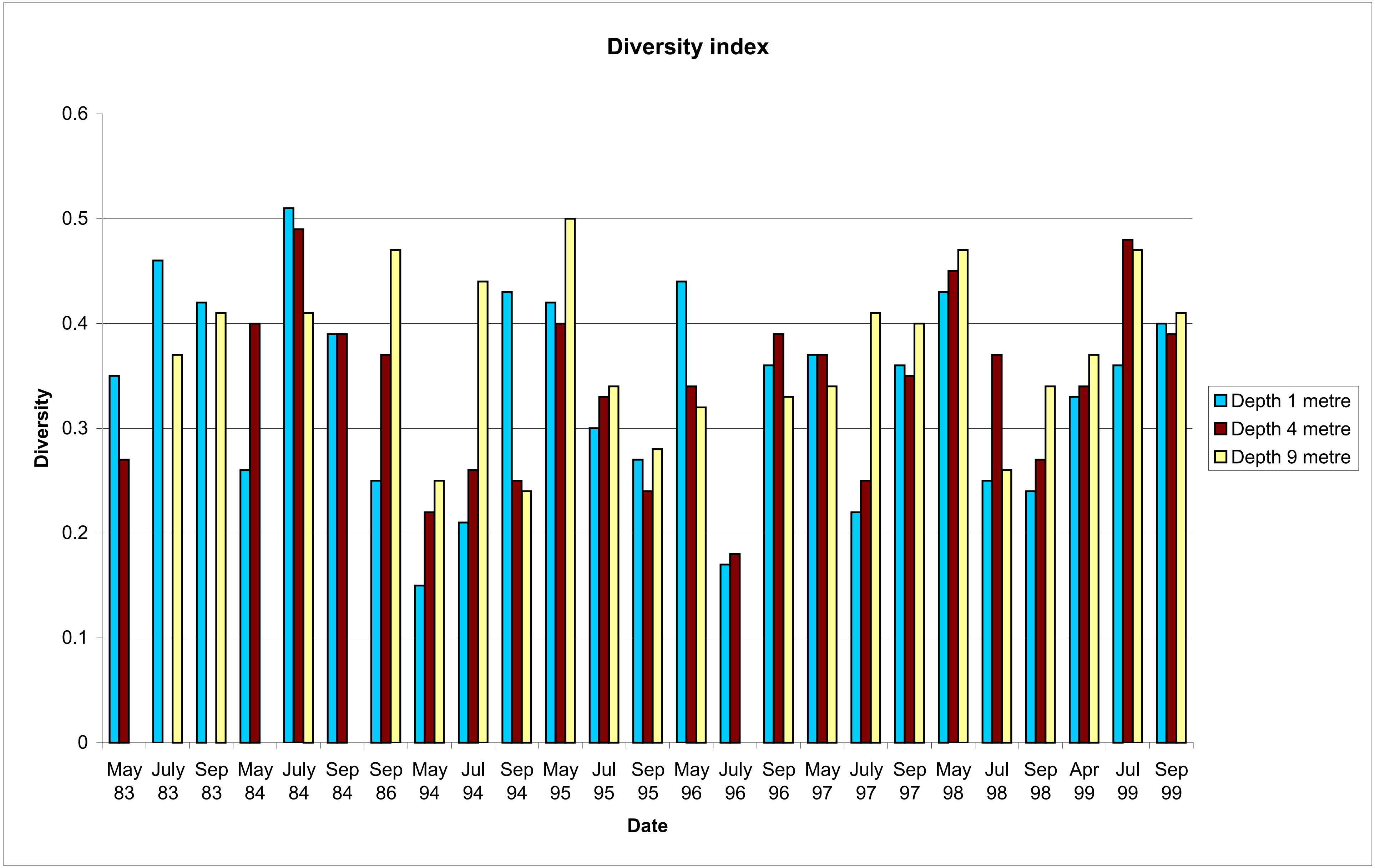

Cryptophyceae - Kathablepharis ovalis
1m
4m
9m
Cells/mL
0
10
20
30
40
50
60
May 83
July 83
Sep 83
May 84
July 84
Sep 84
Sep 86
May 94
July 94
Sep 94
May 95
Jul 95
Sep 95
May 96
Jul 96
Sep 96
May 97
Jul 97
Sep 97
May 98
July 98
Sep 98
Apr 99
July 99
Sep 99
Date

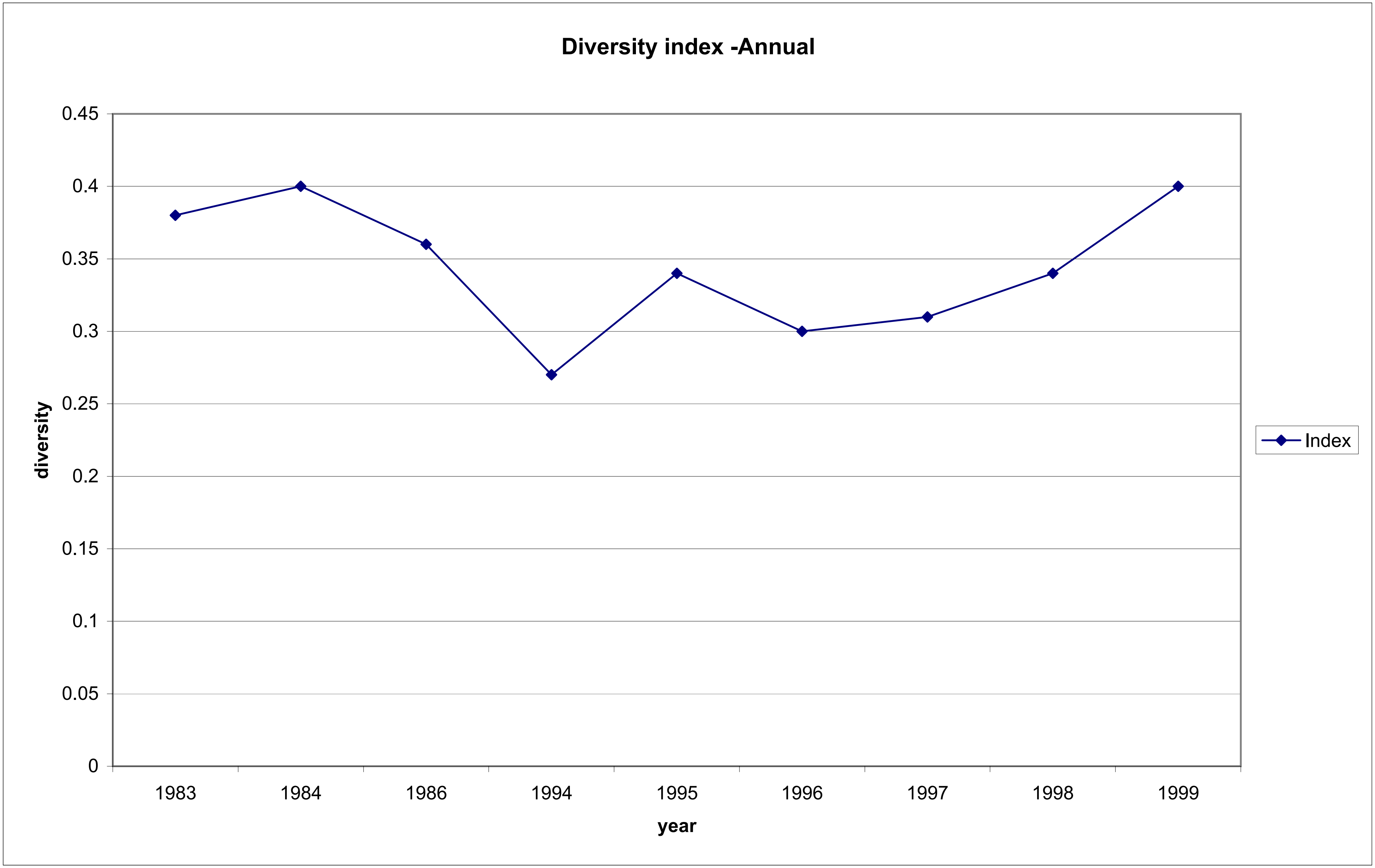

Chlorophyceae (green algae) - Schroderia setigera
Cells/mL
0
1
2
3
4
5
6
Depth 1m
Depth 4m
Depth 9m
May 83
July 83
Sep 83
May 84
July 84
Sep 84
Sep 86
May 94
July 94
Sep 94
May 95
Jul 95
Sep 95
May 96
Jul 96
Sep 96
May 97
Jul 97
Sep 97
May 98
July 98
Sep 98
Apr 99
July 99
Sep 99
Date


Chlorophyceae (green algae) - Elakatothrix gelatinosa
Cells/mL
0
1
2
3
4
5
6
7
8
9
10
Depth 1m
Depth 4m
Depth 9m
May 83
July 83
Sep 83
May 84
July 84
Sep 84
Sep 86
May 94
July 94
Sep 94
May 95
Jul 95
Sep 95
May 96
Jul 96
Sep 96
May 97
Jul 97
Sep 97
May 98
July 98
Sep 98
Apr 99
July 99
Sep 99
Date


Chlorophyceae (green algae) - Botryoccocus braunii (colonies)
Cells/mL
1.4
1.2
1
0.8
0.6
0.4
0.2
0
1m
4m
9m
May 83
July 83
Sep 83
May 84
July 84
Sep 84
Sep 86
May 94
July 94
Sep 94
May 95
Jul 95
Sep 95
May 96
Jul 96
Sep 96
May 97
Jul 97
Sep 97
May 98
July 98
Sep 98
Apr 99
July 99
Sep 99
Date


Dinophyceae - *Gymnodinium* sp
Cells/mL
0
5
10
15
20
25
30
35
1m
4m
9m
May 83
July 83
Sep 83
May 84
July 84
Sep 84
Sep 86
May 94
July 94
Sep 94
May 95
Jul 95
Sep 95
May 96
Jul 96
Sep 96
May 97
Jul 97
Sep 97
May 98
July 98
Sep 98
Apr 99
July 99
Sep 99
Date


Diversity index
Diversity
0.6
0.5
0.4
0.3
0.2
0.1
0
May 83
July 83
Sep 83
May 84
July 84
Sep 84
Sep 86
May 94
Jul 94
Sep 94
May 95
Jul 95
Sep 95
May 96
July 96
Sep 96
May 97
July 97
Sep 97
May 98
Jul 98
Sep 98
Apr 99
Jul 99
Sep 99
Date
Depth 1 metre
Depth 4 metre
Depth 9 metre


Diversity index -Annual
0.45
0.4
0.35
0.3
0.25
0.2
0.15
0.1
0.05
0
diversity
1983
1984
1986
1994
1995
1996
1997
1998
1999
year
Index