MINISTRY OF ENVIRONMENT AND PARKS
PROVINCE OF BRITISH COLUMBIA

Destratification-Aeration of Langford Lake: Physical, Chemical and Biological Responses

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SUMMARY

Langford Lake is a small (60 ha), shallow (15m), eutrophic, monomictic lake situated 12 km west of Victoria, British Columbia. The lake is an important recreational lake for swimming, sailing, and fishing (rainbow trout and small mouth bass). The eutrophic conditions are caused by excessive phosphorus loading from flooded agricultural land and internal phosphorus loading. A destratification aerator was installed in the spring of 1984 to eliminate internal phosphorus loading, and increase fisheries habitat by preventing the formation of an anoxic hypolimnion.

The aerator was unable to meet the oxygen consumption of the sediments in 1984 apparently because of inefficiency of the diffuser and because of inadequate air volume. The result was an anoxic zone at the bottom of the lake, phosphorus and hydrogen sulphide release from the sediments, and a dense algal bloom that was worse than the pre-treatment conditions. A more efficient diffuser, installed in the spring of 1985, oxygenated the entire lake and increased fisheries habitat by 35%. Internal phosphorus loading was eliminated during the summer months when the aerator was operating. Internal loading occurred in September and October after the aerator had been turned off. As a result, the 1986 spring over-turn phosphorus concentrations did not decline below pre-aeration levels. Water clarity was perceived by the local residents to have improved due to a post-aeration shift from filamentous cyanobacteria to diatoms; however, Secchi disc depths remain unchanged. Summer maximum temperatures were not detrimental to the cold water fishery despite record sunshine in July and August 1985.

An examination of phytoplankton, crustacean zooplankton, and macrophyte communities, before and after aeration, showed significant changes in each group. The major response in the phytoplankton was a shift during mid-summer from an assemblage either dominated by, or with a major component of,

cyanobacteria; to an assemblage dominated by greens, diatoms, or chrysophytes. Standing crop did not appear to change as a consequence of aeration. Crustacean zooplankton showed changes both in species composition and in numbers. Both large body (*Daphnia pulex*) and small body (*Ceriodaphnia* spp.) cladocera increased in number and became the dominant group in the zooplankton. Overall numbers of zooplankton also increased. An effect on the non-rooted macrophyte plants was also noted, with *Ceratophyllum* and *Elodea* being significantly reduced in coverage.

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1. INTRODUCTION

Artificial aeration systems have been used for many years to improve water quality or fish habitat (Smith et al. 1975, Pastorok et al. 1980, Taggart and McQueen 1981, Cooke et al. 1986). Lake aeration systems are basically either hypolimnetic or destratification, and both types have been used in British Columbia over the past twenty years (Halsey 1968, Ashley 1983). This paper describes the effects of a destratification system on Langford Lake, a small (60 ha) warm monomictic lake in a suburban area near Victoria on southern Vancouver Island, British Columbia.

Langford Lake has two major recreation uses - swimming/boating and recreational fishing - which were negatively affected by the eutrophic nature of the lake. Phosphorus input from both external and internal loading resulted in high algal standing crops (affecting swimming and aesthetics) and hypolimnetic oxygen depletion (affecting useable fish habitat and causing additional internal loading). A destratification system was chosen for its lower cost relative to hypolimnetic aeration, the generally good success of destratification in aeration of fish habitat and previous reports of positive benefits to water quality as a consequence of destratification aeration (Strecker et al. 1977, Malueg et al. 1973, Barnett 1971, Lorenzen 1977, Lorenzen and Michell 1975, Brynildson and Serns 1977). On the basis of a previous study of the lake, McKean and Munteanu (1981) felt aeration would improve water quality because of the suitable morphometry and lake conditions.

Langford Lake (Figure 1) is located in a suburban area 12 km west of the centre of Victoria, British Columbia, Canada. Langford is a 60 ha kettle lake, formed 10 500 years bp following the Vashion glaciation. The lake is shallow with a maximum depth of 15 m and a mean depth of 6.4 m (Figure 2). The winter climate of the Pacific Northwest is typically mild preventing ice formation and winter stratification. The mild climate rarely allows snow to stay for more than a few days. The result is direct surface runoff during the high precipitation months of December to March. The small watershed (3.3 km^2) in relation to lake volume produces an average flushing time of 3.3 years.

Because of the lake's proximity to residential areas, the lake has several important recreational values:

1) aesthetics for residents around the lake,
2) swimming - mostly families and young children,
3) fishing - stocked annually with rainbow trout, plus smallmouth bass,
4) sailing and canoeing - no motor boats allowed.

The lake's eutrophic conditions are caused by extensive phosphorus loading from agricultural land (flooded in the winter) and internal phosphorus loading (McKean and Munteanu 1981). Because the poor water quality associated with the eutrophic conditions affected recreation, lake restoration techniques were considered so as to reduce the phosphorus loading. Two restoration techniques were recommended:

1) diversion of the agricultural runoff - not completed, and
2) aeration to prevent internal phosphorus loading - installed in 1984.

Destratification aeration was initially installed due to limited capital funds. Capital and maintenance money came from the Habitat Conservation Fund administered by the Provincial Fisheries Branch. Operational Costs ($3000 annually for electricity) were paid by the local Langford Lake Improvement District through taxation of lakeshore residents. The aerator was sized and designed by K. Ashley of the Fisheries Branch, B.C. Ministry of Environment and Parks. The system was designed to be retrofitted, when additional capital funds were available, with a hypolimnetic aerator to enhance the cold water fishery if summer water temperatures exceeded recommended upper limits for rainbow trout.

The objectives of the aeration project were as follows:

1) to maintain dissolved oxygen levels ≥6 mg/L throughout the lake[1],
2) to eliminate the reducing conditions in the summer hypolimnion that cause internal phosphorus loading,
3) to eliminate cyanobacteria blooms through lower phosphorus and pH levels, and
4) to improve water clarity through reduced chlorophyll a concentrations and elimination of cyanobacteria.

The aerator, installed in 1984, consisted of two shore-based compressors (which can be operated alternately or together), a buried supply hose, and a diffusor suspended 2 m above the sediment-water interface at the deepest point in the lake (Figure 2). The system is powered by two 5.6 KW (7.5 hp) rotary vane compressors (0.79 m^3 min^{-1} each FA.D* at 7.0 kg cm^{-2}) operating on 220V single phase electricity (Ashley 1987). The diffusor was constructed of 3.7 cm. I.D. galvanized steel pipe arranged in a cross with 40, 1.5 mm (1/16 inch) orifices. An Atara diffusor (Model 18, ATARA INCORPORATED, 299 Forest Avenue, Paramus, N.J. 07652) was installed in 1985 to provide more efficient oxygen transfer and water circulation.

* free air delivered

[1] Davis (1975) noted a minimum level of 6 mg/L represented the point where oxygen stress will affect the average member of a cold water species.

2. METHODS

Temperature and dissolved oxygen profiles were measured at 1 m intervals with a YSI model 57 meter (YELLOW SPRINGS INSTRUMENT CO., Yellow Springs, Ohio). The meter's thermistor was checked semi-annually in the laboratory, with a precision mercury thermometer. The meter was calibrated with water-saturated air for dissolved oxygen on site prior to each profile.

Water samples were collected with a Van Dorn bottle, preserved (if required) and shipped to the British Columbia Ministry of Environment and Parks Laboratory in Vancouver. Water analyses were completed using the procedures outlined by McQuaker (1976).

Physical, chemical and biological data were collected at intervals of three to four weeks. Unconcentrated phytoplankton samples were collected 1 m below the surface with a Van Dorn bottle, and preserved with Lugol's solution. The organisms were counted with an inverted microscope. Samples for chlorophyll *a* analysis were collected with a Van Dorn bottle at 2 m intervals from the surface to 10 m. Zooplankton were collected with a bottom to surface tow with a 20 mesh (64 μ) net. The macrophytes *Elodea* and *Ceratophyllum* were surveyed before and after aeration by evaluating coverage and density in late August or September.

3. RESULTS

3.1 TEMPERATURE

Figure 3 summarizes the pre- and post-destratification temperature profiles. Maximum summer temperatures were 21-23°C in a normal pre-aeration year (e.g., 1980). Destratification aeration did not change surface water temperatures above or below pre-aeration levels; however, it did raise the temperature of the bottom waters. The bottom of lakes is considered a refuge for cold water fishes such as rainbow trout. In the case of Langford Lake, the cold water refuge was anoxic prior to aeration and not available for trout habitat. The upper lethal temperature for rainbow trout is 25°C (Brett, 1956), consequently destratification was not considered to be hazardous to the rainbow trout.

Destratification was incomplete in July 1984, with a 9°C difference between the surface and bottom waters (Figure 3). The temperature gradient in 1985 was reduced to 5°C due to the more effective diffusor.

Maximum sediment temperatures increased as a result of destratification from 6°C in 1981 to 15°C in 1984 and 18°C in 1985.

3.2 DISSOLVED OXYGEN

The pre-aeration dissolved oxygen profiles (Figure 4) were typical for a monomictic, eutrophic lake. The entire water column was well oxygenated during the unstratified period, December to late April. With the onset of thermal stratification, anoxia quickly developed throughout the entire hypolimnion. Through the months of July and August the anoxic zone represented 35 percent of the lake volume.

During the first year of destratification (1984), the anoxic area was reduced; however, the area greater than 6 mg/L dissolved oxygen was restricted to the top 1 m during late July and August. At this time 85 percent of the lake volume was below 6.0 mg/L.

The low dissolved oxygen concentrations were caused by the transportation, by the aerator, of anoxic water and hydrogen sulphide from the lake bottom to the surface. The anoxic conditions at the bottom of the lake were caused by inadequate circulation and oxygenation, and by the increased oxygen demand of the sediments due to higher sediment temperatures. In retrospect, when the presence of hydrogen sulphide was detected in July 1984, aeration should have been stopped to prevent the surface depletion of oxygen.

In order to improve the oxygen transfer and water circulation an Atara model 18 diffusor was installed in March 1985. An increase in dissolved oxygen concentrations was observed in 1985 (Figure 4). The anoxic zone was restricted to the sediment-water interface and no hydrogen sulphide was detected. Over 95 percent of the lake volume had dissolved oxygen concentrations greater than 6 mg/L through the critical period of late July and August. The result was a 35 percent increase, based on lake volume, in fisheries habitat.

3.3 ANION-CATION WATER CHEMISTRY

As expected, there were no major changes ($p \geq 0.05$) in the anions and cations following aeration (Table 1). Sulphate concentrations, however, increased ($p << 0.001$) but remained inconsequential to the overall water quality.

TABLE 1

MAJOR ANIONS AND CATION IN LANGFORD LAKE

	Pre-aeration	Post-aeration
Ca^{++}	19.8±1.2 (n=31)	19.8±2.1 (n=38)
Mg^{++}	3.9±0.2 (n=29)	4.4±0.4 (n=30)
Na^{+}	6.5±0.1 (n=2)	6.9 (n=1)
HCO_3^-	77.6±5.2 (n=13)	84.3±7.6 (n=15)
SO_4^-	6.0±0.5 (n=11)	8.2±0.3 (n=7)
Cl^-	10.3±0.5 (n=3)	11.4 (n=1)

Aeration, however, did reduce the maximum summer pH from 9.3 to 8.1 (August 1981 and 1985, respectively). This change was not unexpected as destratification aerators introduce CO_2 (both evolved from the lake sediments and incorporated in the compressed air) into the surface waters, which reduces the effects of algal photosynthesis on pH. The physical disruption of dense surface blooms of cyanobacteria can also result in a lowering of pH since dispersion would cause a reduction in pH due to a lessening of intense photosynthetic activity which would otherwise occur.

3.4 WATER CLARITY

Summer water clarity, measured with a Secchi disc, decreased in the first two years of aeration (Figure 5). Depth measurements (m) averaged 3.8±1.2 (n=6), 2.4±1.0 (n=11), and 3.3±1.2 (n=8) in 1980, 1984, and 1985, respectively. Decreased water clarity in 1984 (Figure 5), was caused by an extensive algal bloom during July and August. The Secchi measurement of 0.9 m recorded on July 10, 1984, was the lowest ever recorded for Langford Lake. The Secchi values increased to an average of 3.3 m in 1985.

Although the Secchi disc results in 1985 were not better than the pre-aeration period, there was a perception amongst the lakeshore residents that water clarity had improved. This was due to the absence of floating filamentous cyanobacteria and the dominance of diatoms in July and August 1985.

3.5 PHOSPHORUS

Based on an average N:P ratio of 26:1 (atomic weight ratio in surface water), phosphorus was determined to be limiting algal growth in Langford Lake. A phosphorus budget for 1980 was completed by McKean and Munteanu (1981). The principle sources of phosphorus were agricultural drainage (Figure 2) and internal loading (approximately 50 and 30 percent respectively). Septic tanks, minor inflows and aerial input accounted for the remaining 20 percent.

Spring overturn phosphorus concentrations averaged 25 µg/L (total phosphorus) and ranged between 15 and 30 µg/L (influenced directly by hydraulic loading). Spring overturn was defined in this monomictic lake as the period prior to summer stratification (mid to late April), although algal blooms (principally diatoms) begin as much as two months earlier. The pre-aeration spring over-turn phosphorus concentrations were typically less than the February values (Table 2) because of the utilization of phosphorus and the subsequent sedimentation of the spring diatom bloom. With the initiation of destratification aeration the difference between February and April phosphorus concentrations was reduced, due to the reduced sedimentation of diatoms.

Prior to destratification, the summer surface phosphorus concentrations were low (≤20 µg/L, Table 2), which resulted in higher Secchi disc values (Figure 5), and low chlorophyll a concentrations (Nordin and McKean 1987). In the first year of destratification summer phosphorus concentrations increased to 69 µg/L, which resulted in the largest algal blooms ever observed. Internal loading was responsible for the observed increase in phosphorus and algae. Following the shutdown of the aerator in early September of 1984 and 1985, internal loading was responsible for the rise in the lake's phosphorus concentrations (Table 2).

Internal phosphorus loading in Langford Lake was caused by reducing conditions present at the sediment-water interface (McKean, 1986), and the fact that the sediments are acidic (pH 5 to 6). Under acidic pH conditions phosphorus is bonded to oxidized iron and released when the iron becomes reduced (Mortimer 1941 and 1971). Maintenance of an oxidized sediment-water interface is critical in preventing internal loading. Adequate oxygenation of the sediments in the summer of 1985 reduced internal loading, and phosphorus concentrations remained marginally lower than the 1985 spring overturn results (Table 2).

Aeration through October would eliminate the internal loading observed in 1984 and 1985; however, because of adequate oxygen concentrations for fish and added operational costs, aeration was not continued. To date, destratification has not been successful in reducing the spring overturn phosphorus concentrations because internal loading has not been eliminated.

TABLE 2

TOTAL PHOSPHORUS CONCENTRATIONS FOR THE EPILIMNION OF LANGFORD LAKE

	Winter (Jan → Feb)	Spring Overturn (April)	Summer (July/August)	Fall Overturn (Nov/Dec)
1980	37	25	20	76
1983	28	16	17	57
1984	-	* 22	69 *	75
1985	39	* 35	32 *	51
1986	20	* 25	- *	

Period of Aeration

3.6 PHYTOPLANKTON

The annual phytoplankton succession displays some variation year to year but the general pattern can be characterized as follows. In the normal (pre-aeration) winter and early spring period (November through April), diatoms are the dominant forms (*Asterionella formosa*, *Fragilaria crotonensis*, *Synedra* spp.). Winter standing crop is generally low (<2 μg/L chlorophyll *a* or <1000 cells/mL). Numbers increase with higher light levels in February and March. Spring phytoplankton increases with chlorophyll *a* values up to 5 μg/L. In June, a distinct change takes place with cyanobacteria such as *Aphanizomenon*, *Anabaena*, *Microcystis* and *Coeleosphaerium* gaining dominance and standing crop increasing to greater than 10 μg/L chlorophyll-*a* (>10 000 cells per mL, often to 100 000 cells/mL). Phytoplankton biomass usually decreases during August and September then increases in late autumn as an increased supply of nutrients from the eroding thermocline and hypolimnion is made available to the photic zone. The late autumn blooms are often as large or larger than the vernal peaks with chlorophyll *a* concentrations up to 35 μg/L.

There appear to be two major factors controlling the species composition and amount of phytoplankton. The diatoms appear to dominate in the thermally unstratified period (December to April) when vertical water movements assist them to remain in suspension. When the lake becomes well stratified the diatoms appear to lose their competitive advantage to cyanobacteria which can maintain their position in the water column and consequently have lower sinking rates than diatoms. Depletion of silica in summer also plays a part in summer cyanobacterial dominance. Phytoplankton biomass appears to be correlated with inputs of nutrients to the lake in spring (from watershed runoff) and in fall (from hypolimnetic flux).

The intent of the destratification with regard to phytoplankton was to maintain the physical environment in which the diatoms appear to flourish. In the first year of aeration (1984), it became apparent that the aeration system did not operate efficiently enough to break up the summer stratification, and the phytoplankton successional pattern was very similar to pre-aeration years. However, biomass was high due to high nutrients. In 1985, a more efficient diffuser system achieved better circulation and better air dissolution and a different situation occurred with diatoms retaining dominance through the summer. The pattern of standing crop also changed with high biomass being maintained through the summer, presumably because of a constant supply of nutrients from the sediments and deep waters being conveyed to the photic zone by the aerator.

The species composition also changed with aeration in 1985. During that summer, the dominant forms were greens and diatoms, in direct contrast to earlier years when cyanobacteria dominated. In 1985, the aerator was shut off since it was anticipated that fall cooling would maintain the unstratified water column. In late September, the cyanobacteria had returned to dominate the phytoplankton with a major bloom of *Microcystis* (Figure 2). Cyanobacteria continued to form the majority of standing crop through the fall of 1985.

3.7 ZOOPLANKTON

The aeration of Langford Lake appears to have also had a significant effect on crustacean zooplankton. Prior to aeration, the zooplankton was composed of five major species: three cladocera (*Ceriodaphnia reticulata*, *Daphnia pulex*, *Bosmina coregoni*) and two copepods (*Diaptomus oregonensis* and *Cyclops bicuspidatus*). *Chydorus* and *Diaphanasoma* occurred in small numbers. Of the major species, *Daphnia* and *Ceriodaphnia* were the dominants with *Daphnia* most numerous in spring and early summer and *Ceriodaphnia* in the late summer and fall. Standing crop prior to aeration was moderately high (summer average 380 mg/m^2 dry weight). In 1985 during aeration, *Daphnia* standing crop increased significantly both in absolute biomass as well as in relation to other species. Comparing pre-aeration and aeration summer standing crop data, *Daphnia* accounted for more than 80% of the June and July biomass and *Ceriodaphnia* more than 80% of the August biomass during aeration. The standing crops of both species were much greater than during the non-aerated period (generally by 50%). The total crustacean zooplankton biomass for the summer months (May, June, July, August) increased to 1067 mg/m^2 dry weight (1985). Bosmina and Cyclops both showed declines in numbers during aeration.

3.8 VASCULAR PLANTS

We were interested in the possible response of the macrophytes of Langford Lake. Two species are plentiful in Langford Lake: Ceratophyllum demersum and Elodea canadensis. Neither species has well developed root systems and both appear to depend on aqueous rather than sediment nutrient supply. It had been determined previously (Warrington and McKean 1984), that if a phosphorus reduction to less than 20 μg/L occurred, Cerotophyllum should decrease (Figure 3). The lake was surveyed annually from 1983 to 1987 in August and September to determine coverage in littoral areas of these macrophytes. The abundance of the plants has been identified by recreational users of the lake as a problem and the plants are usually cut and removed by a mechanical harvester on an annual basis. The 1983 survey was used as baseline for later surveys. In 1984 little, if any difference, was noted; however, as noted earlier, the aeration was not effective in 1984 so little change was expected. In 1985, a noticeable change occurred with reduced numbers and coverage of both Ceratophyllum and Elodea (Figure 4). Fewer weeds reached the surface of the lake, many less weed fragments were generated, and no harvesting was required. Additional details of the surveys can be found in Nijman (1985). In 1987 the situation appeared to continue to improve and local residents attribute the improvement to the aeration (Nijman 1987).

3.9 BENTHOS

No quantitative sampling was carried out to document changes in benthic organisms; however, it was noted during sediment sampling after aeration that chironomids were very common. Pre-aeration sediments, in contrast, rarely contained any animals.

3.10 FISH RESPONSE

Creel surveys indicated that the aerator had improved fishing success for rainbow trout on Langford Lake (Clough 1986). The possible factors are increased zooplankton production, increased feeding efficiency, and expanded habitat by the aeration project.

4. DISCUSSION

The purpose of the destratification aeration project was to improve the recreational suitability of Langford Lake. The success of destratification was evaluated on changes in the physical, chemical and biological environments.

Beneficial physical and chemical changes were the increased fisheries habitat through higher dissolved oxygen concentrations without elevating water temperatures to levels considered lethal to cold water species; the lower maximum pH values which decreased by 1.2 units; and the perceived improvements in water clarity. The project failed to eliminate the reducing conditions at the sediment-water interface causing internal phosphorus loading. This in turn prevented reductions in spring overturn phosphorus concentrations and prevented measurable improvements in water clarity.

Internal loading in 1984 appeared to be caused by an inadequate amount of aeration (volume of air or inadequate diffusion (design problem) while in 1985 it was likely caused by the failure to operate the aerator through the fall (operational problem).

In general, however, the Langford Lake aeration project has been considered by the public to be a success, on the basis of subjective observations of changed algal composition, reduced aquatic plants and better fishing success. The technical evaluation appears to lend some support to this evaluation, although, in an objective sense, there are some qualifications to the apparent success. A number of aeration projects described in the literature have reported similar positive results in biological response to destratification aeration; however, other reports exist of insignificant change or negative consequences.

The shift in phytoplankton species composition from cyanobacteria to diatoms or greens has been reported as a typical response to aeration (e.g., Pastorak _et al_. 1980). This shift can be attributed to the changes in pH,

CO_2 (Shapiro, 1984) and turbulence (Knoechel and Kalff 1975) caused by aeration and enhanced zooplankton grazing pressure which may result from aeration. Little or no change in phytoplankton standing crop or numbers appears to be the most common response to destratification aeration (Pastorak et al. 1982) and this appears to be the case for Langford Lake.

The response of zooplankton, notably the increase in Daphnia, has also been cited by a number of other authors as a response to aeration (Brynildson and Serns 1977, Shapiro et al. 1975). Zooplankton response has been attributed to a more suitable food supply as a consequence of the shift from cyanobacteria to more "digestible" phytoplankton types (Porter 1973) and possibly to an expansion of vertical habitat.

Few studies are available in the literature to which one can compare the results of the unquantified responses of aquatic macrophytes, benthos and fish of Langford Lake.

In the Pacific Northwest of North America there are many lakes which appear to be amenable to destratification aeration as a means of improving water quality and fisheries. The largest risk with destratification systems is the possibility of exceeding lethal temperatures for fish by increasing heat input into the lake. In cool coastal areas, if Langford Lake is used as an example, this risk is minimal. One precaution which must be taken, however, is to ensure correct sizing of the equipment in order to mix the water column completely and maintain destratification. This was not done for Langford Lake in 1984 nor for several other cases with poor results reported in the literature.

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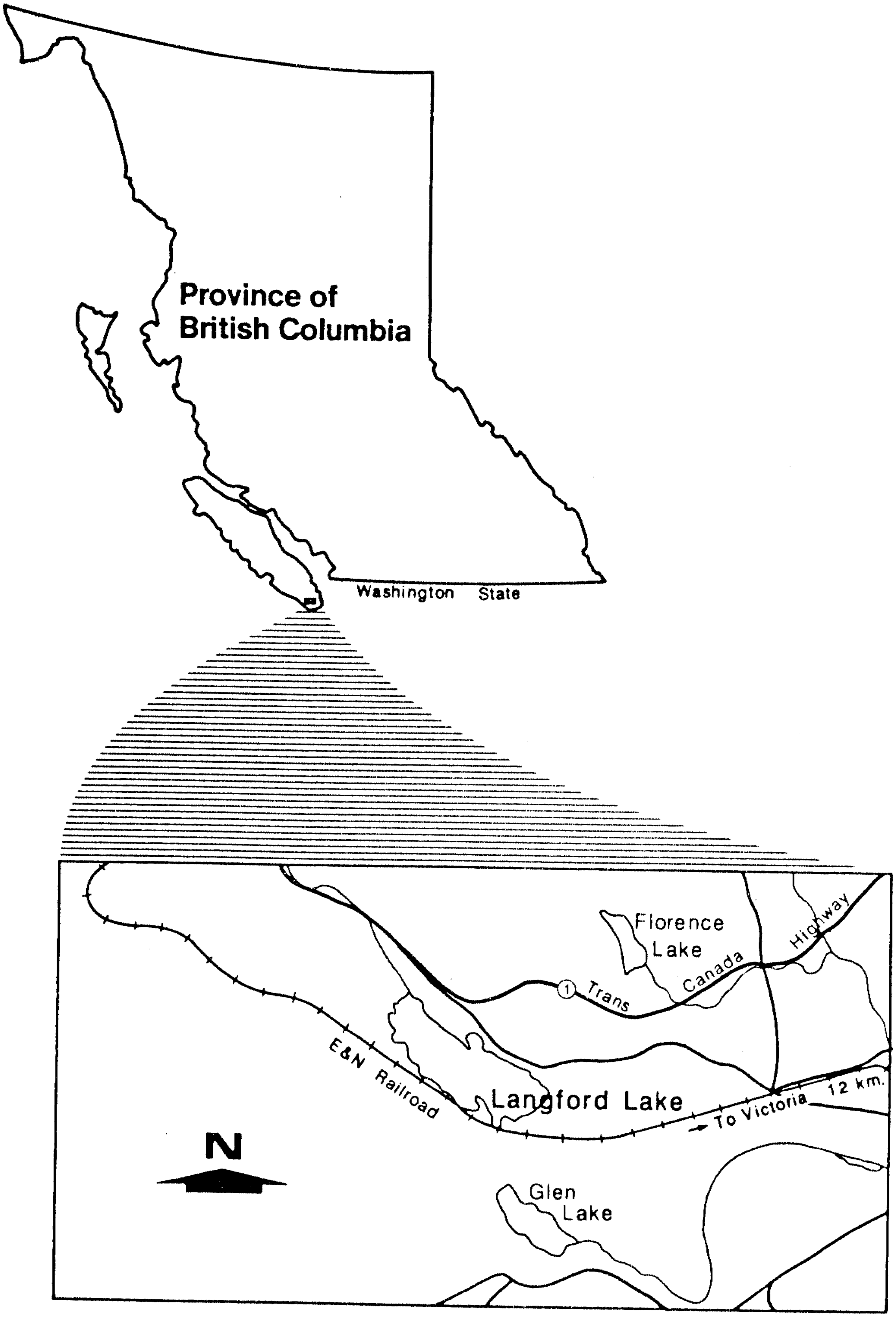

FIGURE 1: Location of Langford Lake.

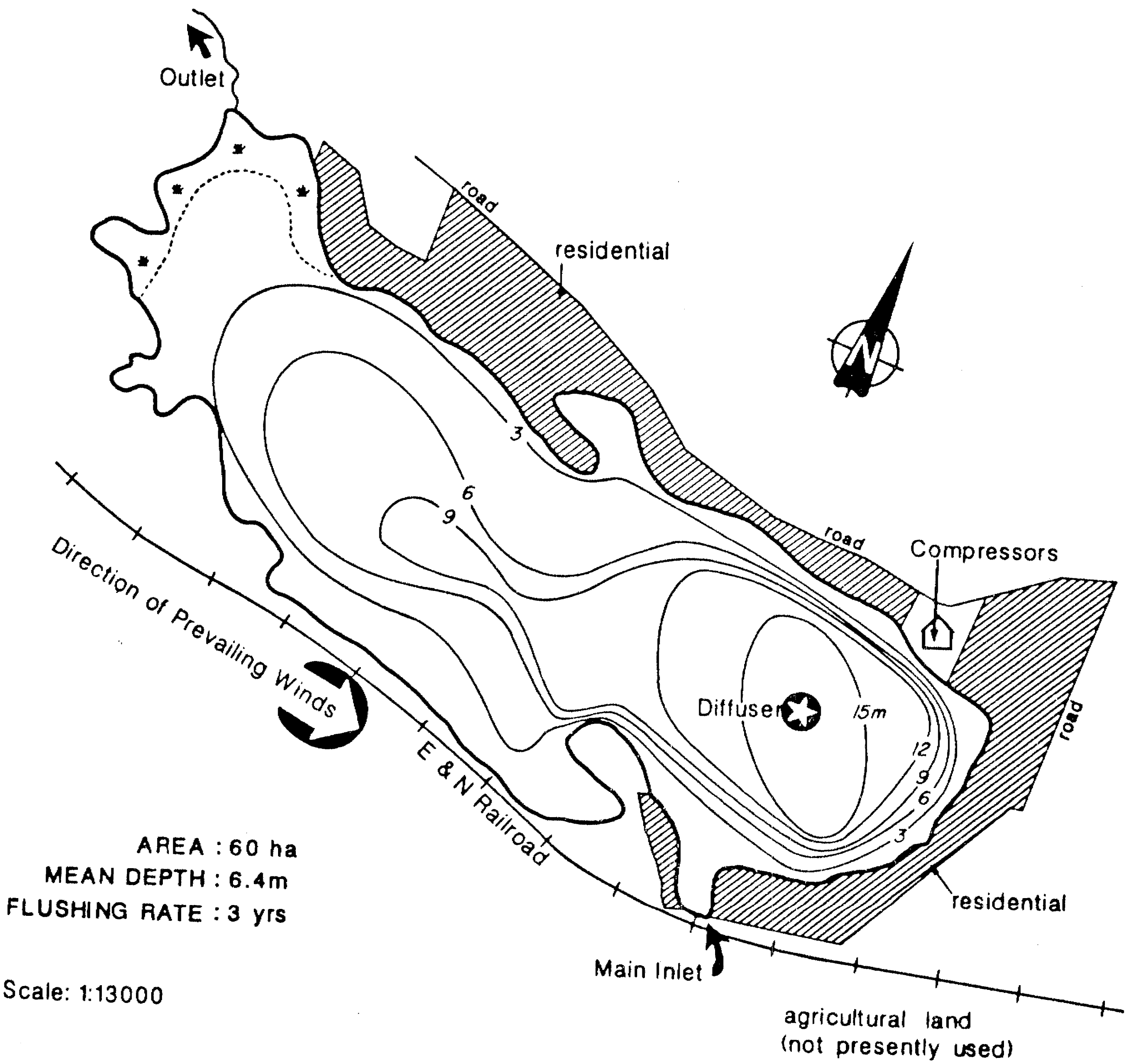

FIGURE 2: Bathymetry and Foreshore Development around Langford Lake.

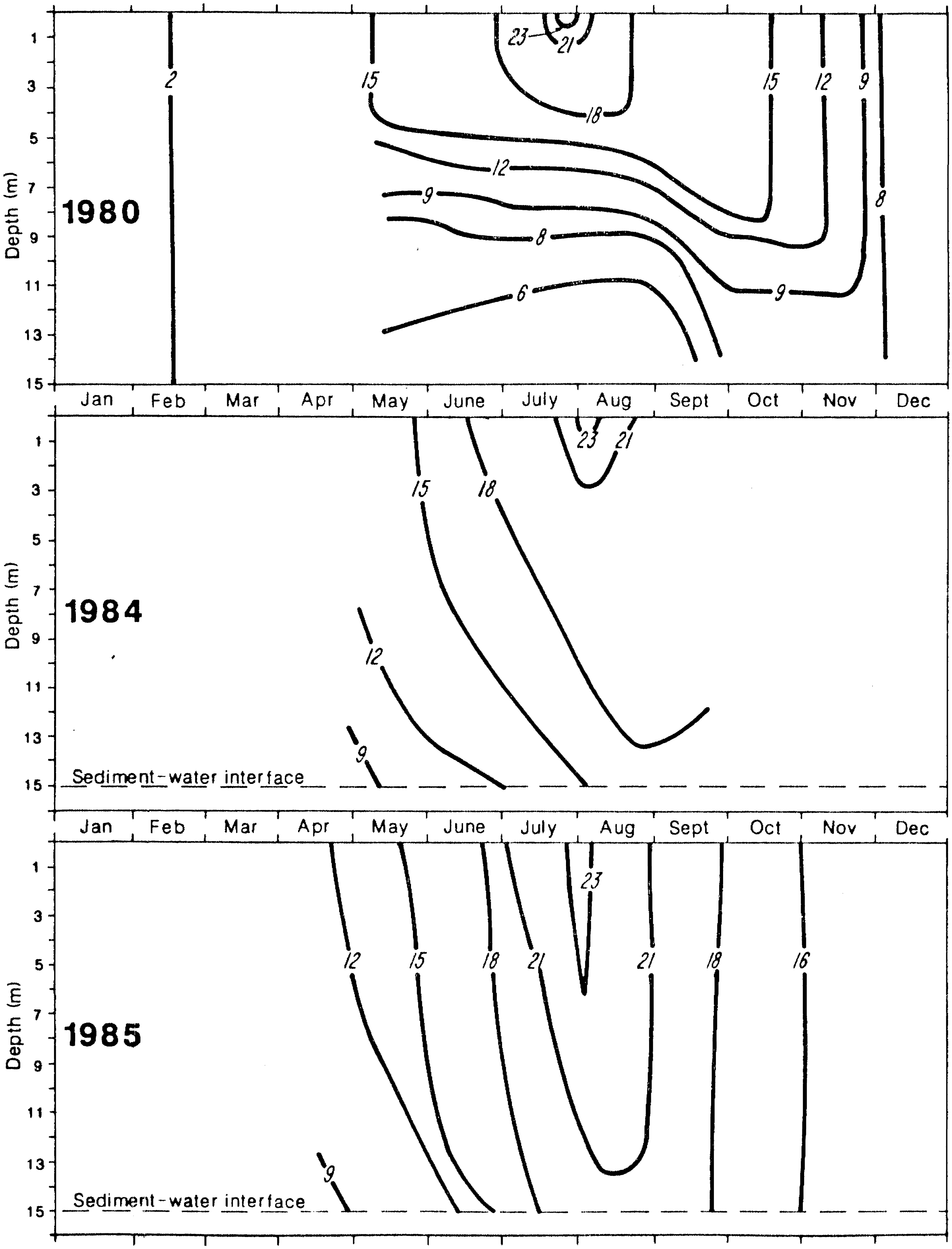

FIGURE 3: Isotherms (^{o}C) for Langford Lake: 1980, 1984, and 1985.

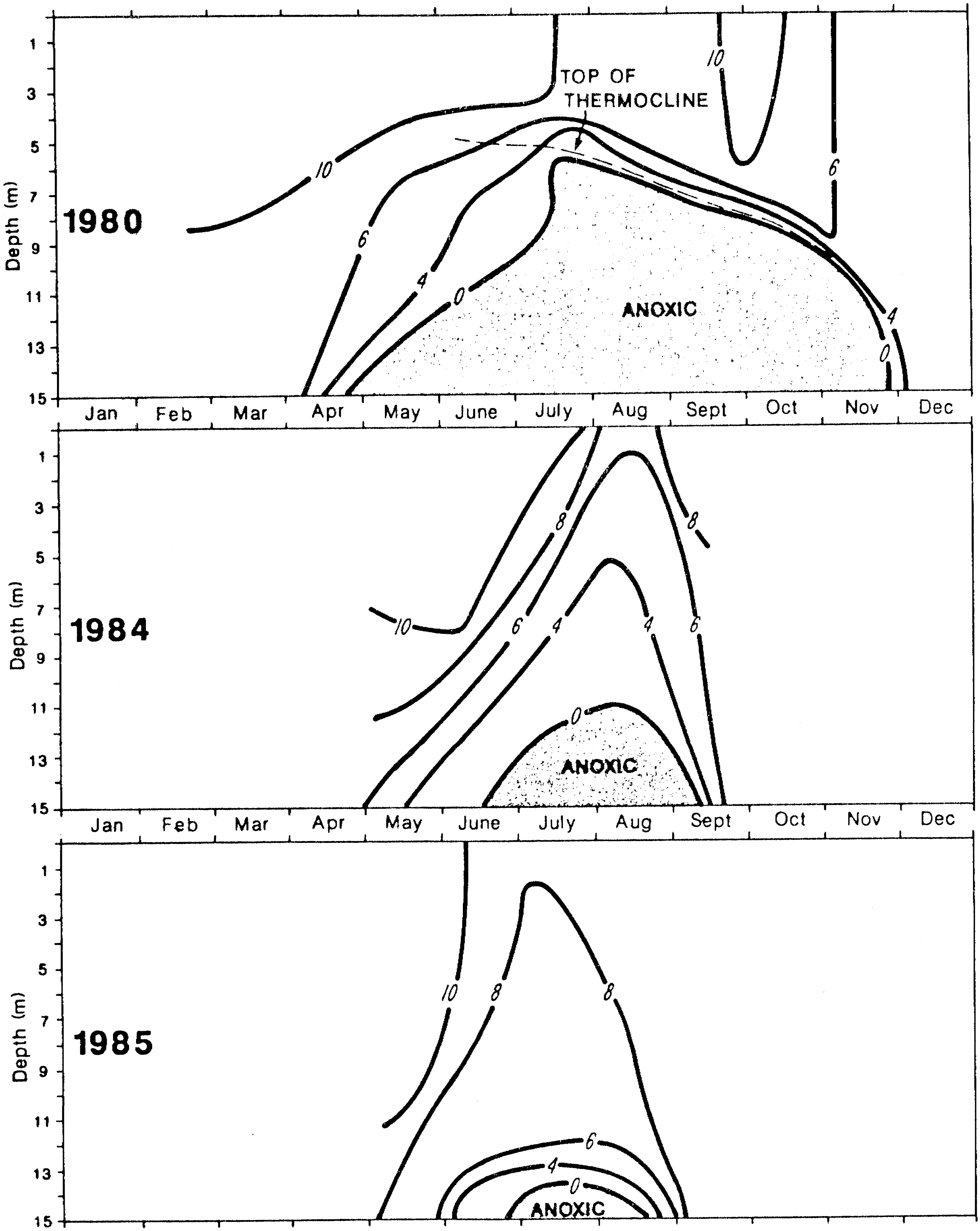

FIGURE 4: Isopleths of Dissolved Oxygen (mg/L) for Langford Lake, 1980, 1984, and 1985.

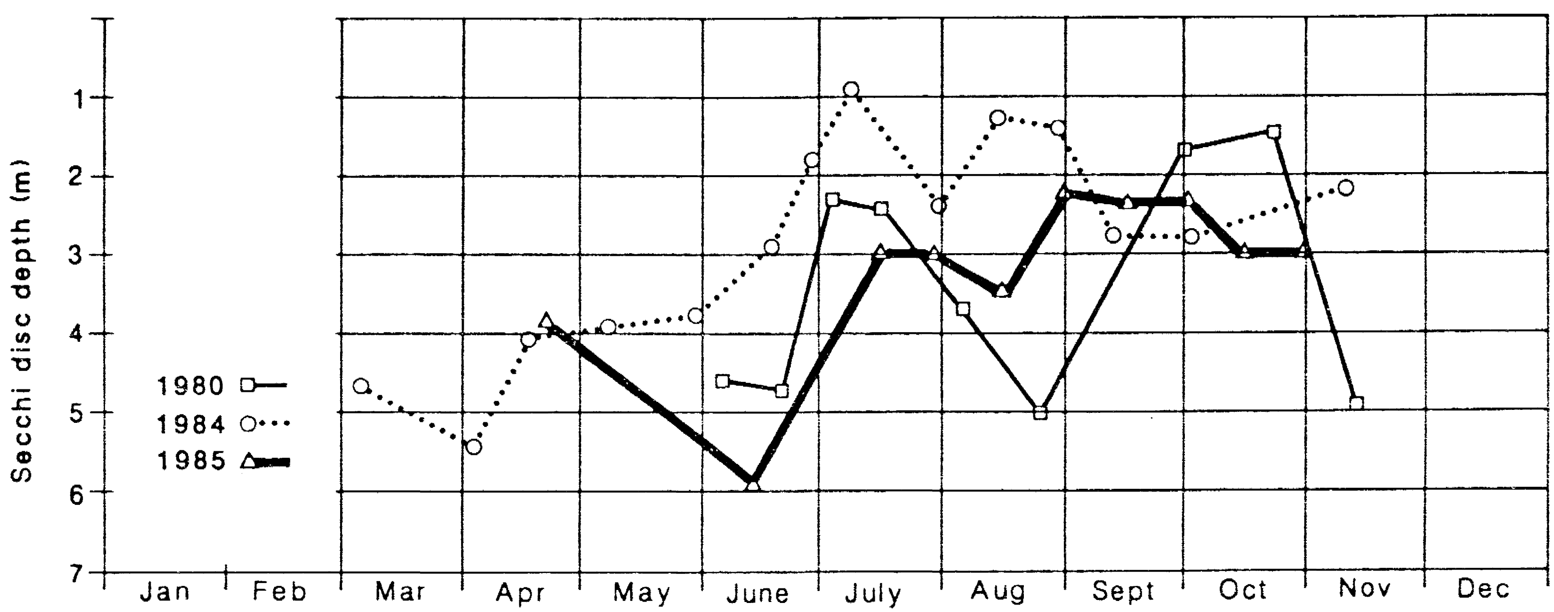

FIGURE 5: Secchi disc depths for Langford Lake: 1980, 1984, and 1985.

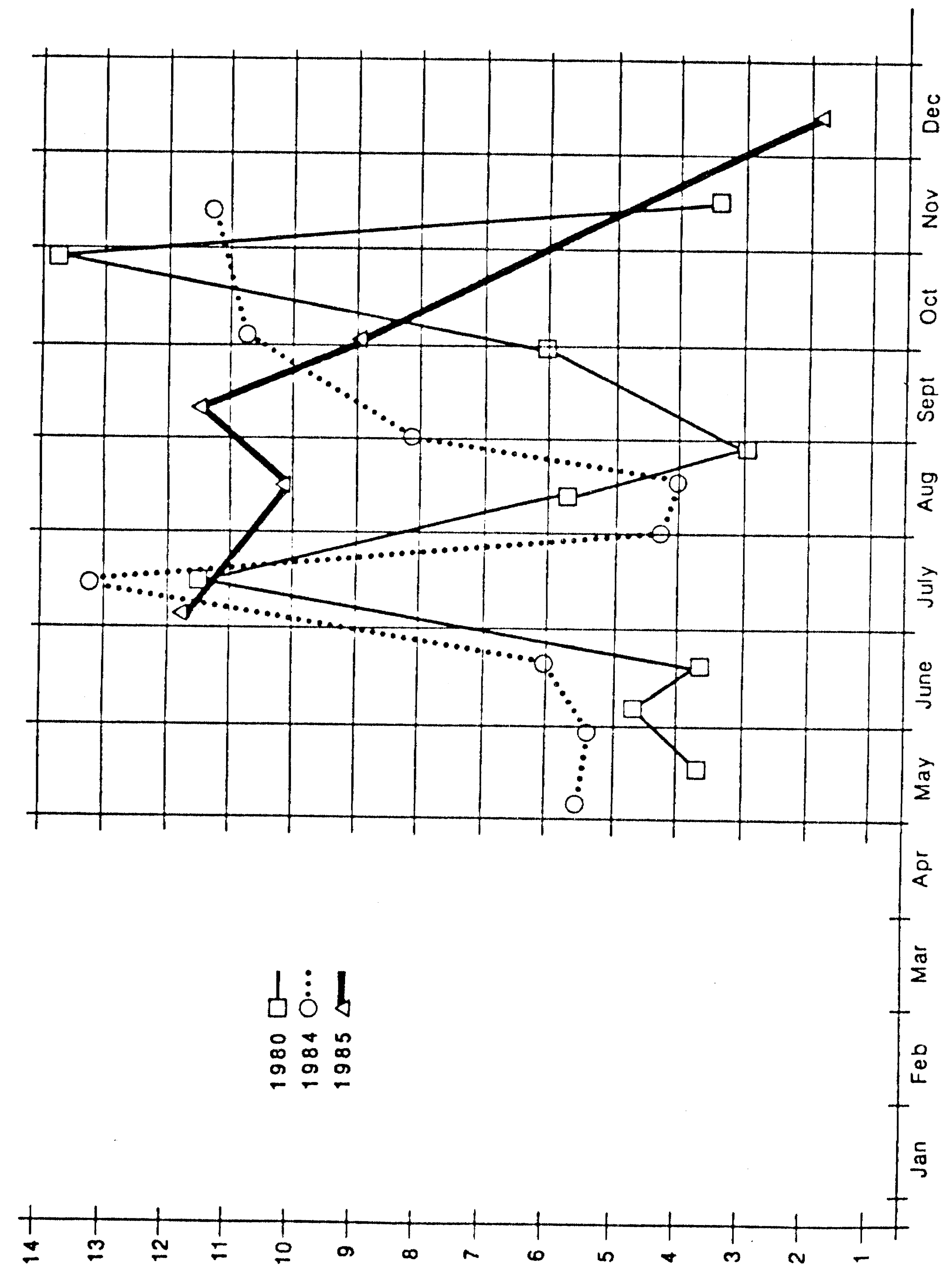

FIGURE 6: Phytoplankton standing crop.

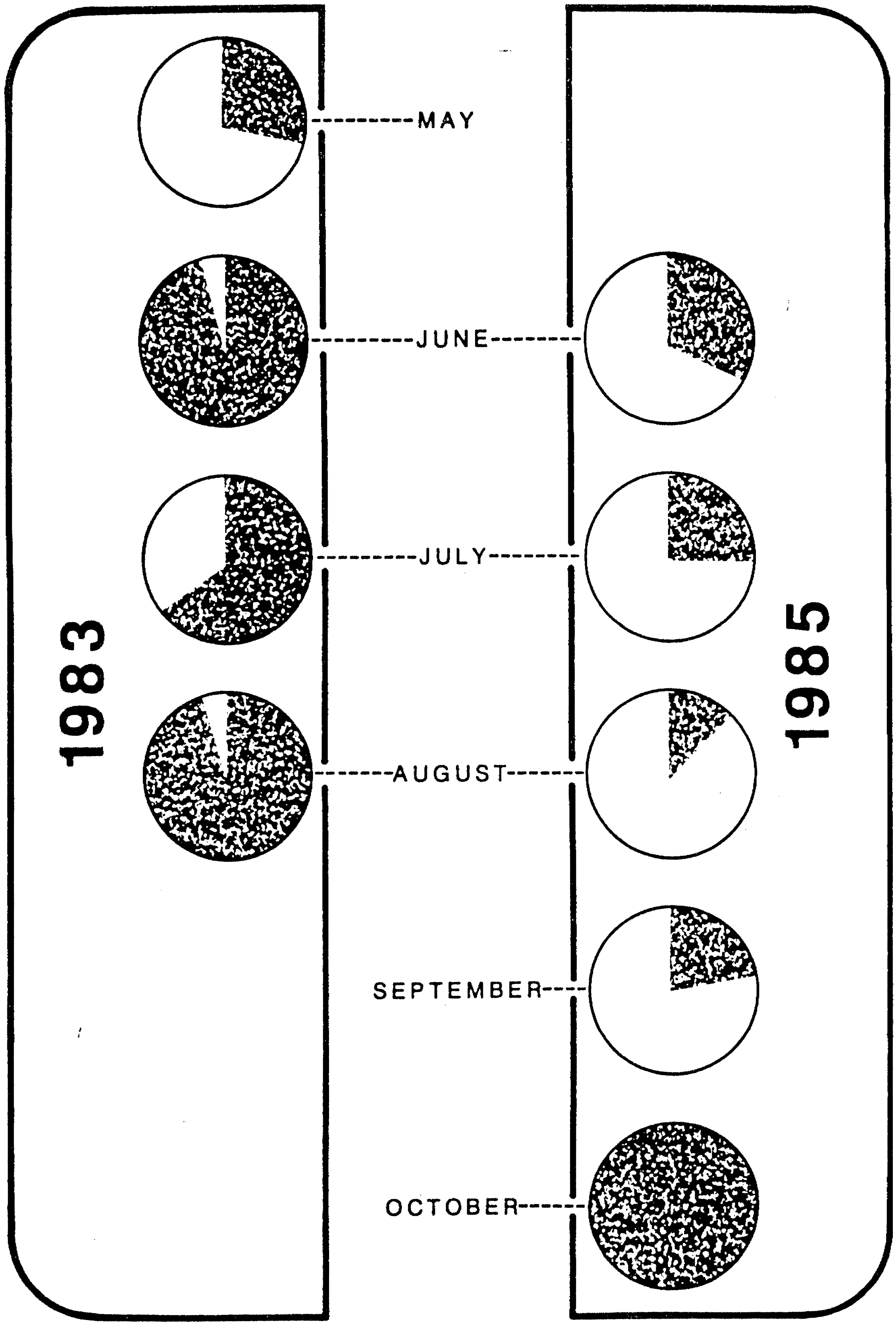

FIGURE 7: Cyanobacteria as a percentage of phytoplankton numbers.

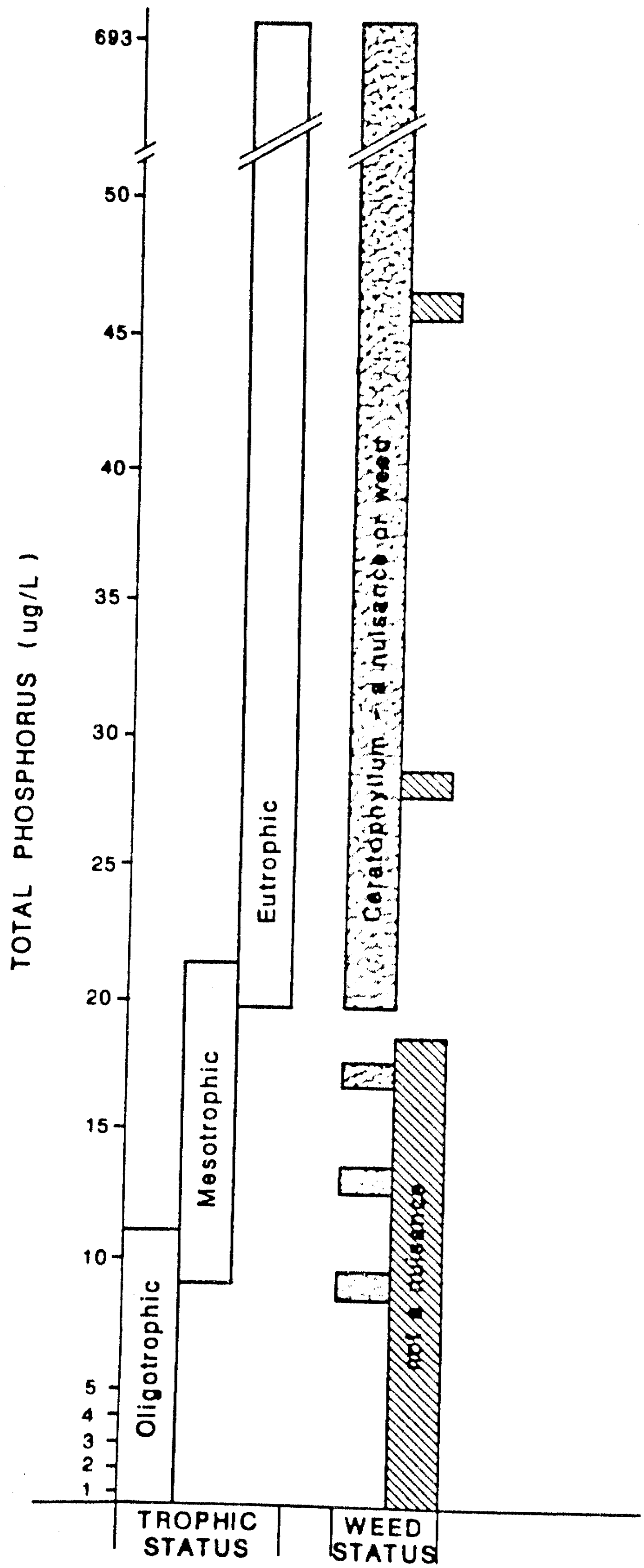

FIGURE 8: Lake Phosphorus concentration and the nuisance growth of Ceratophyllum.

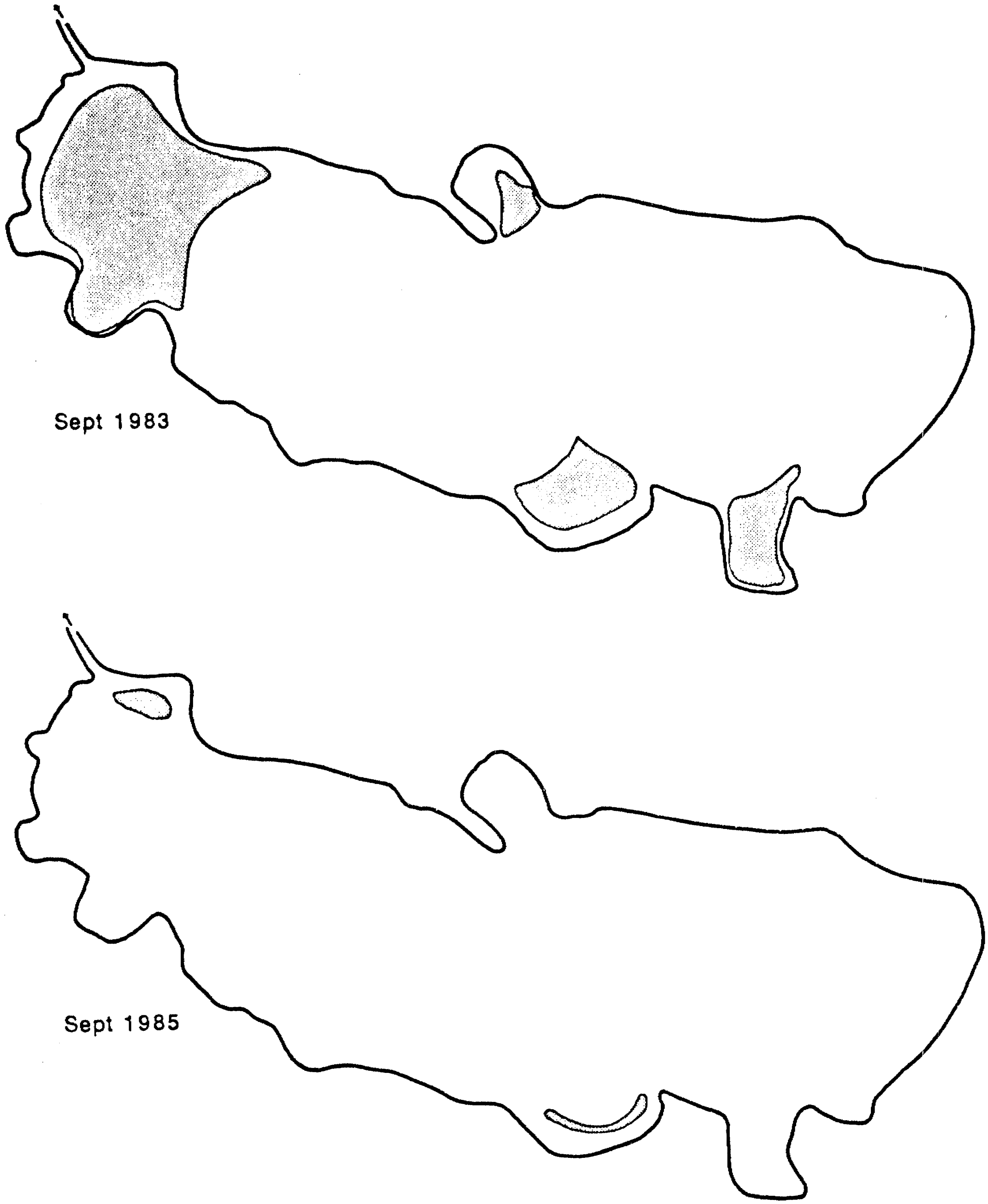

FIGURE 9: Areas of Langford Lake with heavy coverage of *Ceratophyllum demersum*.