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THE EFFECT OF CONTINUOUS ADVANCED WASTEWATER TREATMENT BY THE CITY OF SPOKANE ON THE TROPHIC STATUS OF LONG LAKE, WA. DURING 1979

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THE EFFECT OF CONTINUOUS ADVANCED WASTEWATER TREATMENT BY THE CITY OF SPOKANE ON THE TROPHIC STATUS OF LONG LAKE, WA. DURING 1979

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ABSTRACT

Long Lake, WA. is an impoundment of the Spokane River approximately 24 km northwest of Spokane. Prior to 1978, the reservoir was characterized as eutrophic. The major source of algal nutrient was determined to be the City of Spokane's primary sewage treatment plant effluent.

To address the problem of excessive nutrient loading to the reservoir, an advanced wastewater treatment (AWT) facility was built by the City of Spokane to remove at least 85 percent of the phosphorus from its municipal wastewater. The facility began operation on 22 August 1977 and phosphorus removal was initiated 15 December 1977.

Investigation during 1978 showed that with reduced primary productivity, chlorophyll _a_ concentrations, spring phosphorus concentrations and hypolimnetic anoxia, Long Lake had reverted to a more mesotrophic condition. The overall decrease in chlorophyll _a_ standing crop was directly related to decreased phosphorus loading. Also, phytoplankton biovolumes declined in all classes with the exception of the cyanophytes.

The purpose of this study was to document the water quality of Long Lake and its tributaries during the second year of operation of Spokane's AWT facility.

As in 1978, Long Lake continued to exhibit mesotrophy. Phosphorus removal from the sewage effluent resulted in significantly lower phosphate loads to the reservoir than before AWT. Phytoplankton

biovolumes were also reduced. Correlation showed that chlorophyll a concentrations gave a reasonable estimate of phytoplankton standing crop. In addition, areal phosphorus loading gave an excellent estimate of mean chlorophyll a concentration.

The successional patterns for the major zooplankters were similar to those observed in 1978 and prior to AWT. Zooplankton standing crops were less dense than those determined before AWT, possibly the result of decreased food availability.

SUMMARY AND CONCLUSIONS

1. The total monthly discharges for the Spokane River sampling stations (F.W., S.M., N.M. and DAM) during the growing season (June - November) were not significantly different from other sampling years (Soltero _et al._, 1973; 1974; 1976; 1978; 1979).

2. Analysis of variance and least significant difference statistical evaluation showed that mean daily total and orthophosphate loads in 1979 for S.E. were significantly lower than those prior to advanced wastewater treatment (AWT), but not from those determined in 1978. Mean daily total nitrogen loads for S.E. in 1979 were also found to be significantly less than those determined before AWT. Nitrate and ammonia were the predominant forms of inorganic nitrogen in 1979, reflecting the activated sludge process and secondary clarification.

3. The sewage outfall effected an increase of total inorganic nitrogen ($NO_3^-\cdot N + NO_2^-\cdot N + NH_3\cdot N$) and orthophosphate concentrations in the river of 1.6 and 2.5 times, respectively. Similar inorganic nitrogen increases had been observed prior to and following AWT (Soltero _et al._, 1973; 1974; 1975; 1976; 1978; 1979). However, before AWT, orthophosphate concentrations in the river usually increased 13 fold following the introduction of the sewage effluent.

4. The S.E. was the major contributor of phosphate to Long Lake prior to AWT, but the Spokane River became the primary contributor following AWT in 1978 (Soltero _et al._, 1978). During the growing

biovolumes were also reduced. Correlation showed that chlorophyll *a* concentrations gave a reasonable estimate of phytoplankton standing crop. In addition, areal phosphorus loading gave an excellent estimate of mean chlorophyll *a* concentration.

The successional patterns for the major zooplankters were similar to those observed in 1978 and prior to AWT. Zooplankton standing crops were less dense than those determined before AWT, possibly the result of decreased food availability.

season in 1979, the river contributed approximately 43 percent of the phosphate load entering the reservoir, while S.E. made up 40 percent of the influent load. The river above its confluence with H.C. was also determined to be the major source of nitrogen entering the reservoir in 1979.

5. The highest retention time calculated was approximately 75 days for August. No significant differences were evident in monthly mean retention times during 1979 when compared with other sampling years (Soltero et al., 1973; 1974; 1975; 1976; 1978; 1979) for the period of June through November.

6. As in 1978 (Soltero et al., 1979), hypolimnetic anoxia was present for approximately two months below the 21 meter depth at station 2. Anoxia was also apparent at stations 0 and 2 for about one month below 18 meters.

7. Phytoplankton species composition and seasonal succession observed in 1979 was similar to 1978 findings (Soltero et al., 1979). Overall decreased phytoplankton biovolumes occurred with the Bacillariophyceae being the major contributor to standing crops. The Cyanophyceae were observed, but biovolumes were considerably less than those determined in 1978. Microcystis aeruginosa dominated the phytoplankton at stations 0 and 2 in mid-August and early September, respectively. Anabaena circinalis and A. spiroides were the major contributors to the standing copy in the upper end of the reservoir during September. Biovolumes contributed by these two species were substantially less than those determined for A. flos-aquae in 1976 (Soltero, unpublished data) but slightly higher than the 1977 values (Soltero et al., 1978). Anabaena flos-aquae was observed only once

at station 4 during early September. By early October, blue-green standing crop had declined throughout the reservoir with the Bacillariophyceae becoming dominant.

8. The lower chlorophyll a concentrations observed in 1979 (as compared to pre-AWT levels) can be directly related to decreased phosphate loading. However, the mean daily total phosphate load to the reservoir in 1979 was higher than that determined in 1978 and resulted in greater specific surface phosphate loading and elevated chlorophyll a concentrations during the growing season.

9. Zooplankton standing crop in 1979 remained both compositionally and numerically similar to that observed in 1978 (Soltero et al., 1979). The Rotifera dominated the community followed by the Eucopepoda and then the Cladocera. Grazing did not appear to significantly limit phytoplankton production. Also, zooplankton standing crops were less dense than those determined before AWT (Soltero et al., 1973; 1974; 1976; 1978), possibly the result of decreased food availability.

10. Long Lake maintained its mesotrophic status during 1979.

INTRODUCTION

Long Lake is an impoundment of the Spokane River approximately 24 km northwest of Spokane, WA. Previous limnological investigation (Soltero, Gasperino and Graham, 1973; 1974 and Soltero _et_ _al_., 1975; 1976; 1978) has shown that the major source of algal nutrients influent to Long Lake was the City of Spokane's primary sewage treatment effluent via the Spokane River. These nutrients promoted excessive algal growth, extensive hypolimnetic anoxia and increased hypolimnetic concentrations of orthophosphate, which are characteristic of a eutrophic system.

In compliance with Washington State Department of Ecology directives, the City of Spokane built an advanced wastewater treatment (AWT) facility with phosphorus removal which was put into operation on 22 August 1977. (Phosphorus removal began on 15 December 1977.) Advanced treatment was defined as 85 percent or better BOD removal, 90 percent or more suspended solids removal and 85 percent or greater phorphorus removal.

Since the initiation of AWT, phosphorus loading to the reservoir has decreased (Soltero _et_ _al_., 1979). The corresponding decline in chlorophyll _a_ concentrations, primary productivity, orthophosphate concentrations and hypolimnetic anoxia indicated that eutrophic Long Lake reverted to a more mesotrophic state within one year of "on-line" AWT.

The primary objective of this study was to document the water quality of Long Lake and its tributaries during the second year of AWT operation. At various points throughout the report an attempt was made to compare these findings with previous study years.

DESCRIPTION OF STUDY AREA

Sampling stations were established on the Spokane River and its tributaries at the following locations (Fig. 1A):

Hangman Creek (H.C.)--lat. 47°39'10", long. 117°26'55", in NW¼ Sec. 24, T. 25N, R. 42E., Spokane County, at river km 1.3.

Spokane River, Fort Wright Bridge (F.W.)--lat. 47°40'50", long. 117°27'00", in NE¼ Sec. 11, T. 25N., R. 42E., Spokane County, at river km 112.5.

Sewage Effluent, Spokane Advanced Wastewater Treatment Plant (S.E.)--lat. 47°41'40", long. 117°28'30", in NE¼ Sec. 3, T. 25N, R. 42E., Spokane County.

Spokane River, Seven Mile Bridge (S.M.)--lat. 47°44'25", long. 117°31'10", in NE¼ Sec. 20, T. 26N, R. 42E., Spokane County, 11.7 km northwest of Spokane, at river km 99.6.

Spokane River, Nine Mile Dam (N.M.)--lat. 47°46'29", long. 117°32'35", in SE¼ Sec. 6, T. 26N., R. 42E., Spokane County, off left bank below Nine Mile powerhouse and at river km 93.2.

Little Spokane River (L.S.R.)--lat. 47°47'00", long. 117°31' 44", in SE¼, NW¼ Sec. 5, T. 26N., R. 42E., Spokane County, at county road bridge and at river km 1.8.

Spokane River, Long Lake Dam (DAM)--lat. 47°50'12", long. 117°50'25", in NW¼, SW¼ Sec. 13, T. 27N., R. 39E, Lincoln County, off left bank below Long Lake powerhouse and at river km 54.5.

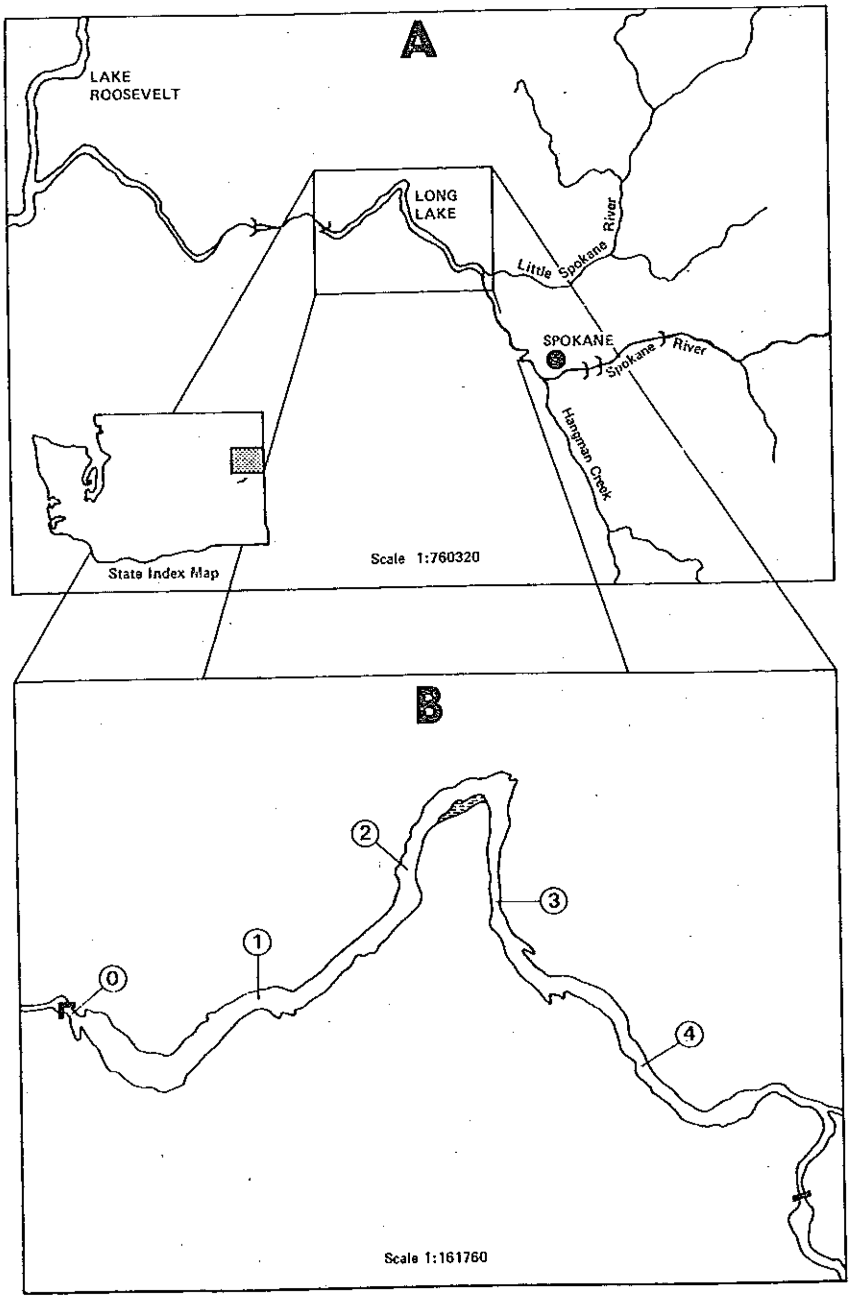

Figure 1. Map of the lower Spokane River system detailing the study area.

Long Lake was formed by a concrete dam (located in Lincoln County) completed in 1913 and raised in 1950. Water can be discharged from the dam through two outlets: the spillways (crestline elevation 460 m) and the power penstocks (centerline elevation 457 m). Unless it is necessary to discharge water over the spillways, all water is discharged through the power penstocks. Morphometric data for the reservoir are given in Table 1. Five sampling stations were established on the reservoir (Fig. 1B). Station 0 was located just behind the dam with the remaining stations located at approximately 8 km intervals up the reservoir for 32 km to station 4.

Table 1. Morphometric data for Long Lake, WA. at maximum capacity (elevation 468.3 m).

Maximum Length	35.4 km
Maximum Effective Length	5.8 km
Maximum Width	1.1 km
Maximum Effective Width	1.1 km
Mean Width	571.8 m
Maximum Depth	54.9 m
Mean Depth	14.6 m
Area	208.4×10^5 m^2
Volume	304.9×10^6 m^3
Shoreline Length	74.3 km
Shoreline Development	4.6
Bottom Grade	0.15%

METHODS AND MATERIALS

Spokane River and Its Tributaries

Sixteen samplings were made from 5 January to 3 December 1979. Sampling was monthly except for the period of June through September when it was biweekly. All samples were collected with an eight-liter rubber bucket from the main current of the stream.

Water Chemistry

Temperature, pH, conductivity (at 25°C) and dissolved oxygen were determined with a Hydrolab® System 8000. Turbidity (nephelometric method), total alkalinity (potentiometric titration), calcium (EDTA titration), magnesium (calculation method), nitrate nitrogen (chromotropic acid method), nitrite nitrogen (diazotization), ammonia nitrogen (Nessler's method), total and total soluble nitrogen (Kjeldahl digestion--electrode method), orthophosphate (stannous chloride method), total and total soluble phosphate (persulfate digestion--stannous chloride method) and silica (molybdosilicate method) determinations were made in the laboratory within 24 hours after collection as described by A.P.H.A. (1976).

The sum of sodium plus potassium ion concentrations (me l^{-1}) was computed by taking 0.01 of the sample conductance and subtracting from the product calcium and magnesium ion concentrations (me l^{-1}).

Hydrology

Discharge measurements for Hangman Creek (gauging station 12424000) Spokane River (gauging station 12422500) and the Little Spokane River (gauging station 12431000) were supplied by the U.S. Geological Survey. Records of total discharge from Nine Mile and Long Lake dams were furnished by the Washington Water Power Company. The sewage discharge measurements were obtained from the daily records kept at Spokane's wastewater treatment facility.

Reservoir

Reservoir collections were made from 6 April to 5 November 1979. Sampling frequency was the same as that for the river and its tributaries Water samples were collected at three-meter intervals with a one-liter Kemmerer sampler. Also, a euphotic zone composite was collected at each station by combining samples (usually taken at two-meter intervals) of equal volume from the surface to the lower limit of the euphotic zone.

Water Chemistry

The methodology for the reservoir chemistry was the same as that previously described for its influent and effluent waters. Temperature, pH, conductivity and dissolved oxygen measurements were made *in situ*, while turbidity, orthophosphate, nitrate nitrogen, nitrite nitrogen and ammonia nitrogen were determined in the laboratory.

Light

Vertical profiles of light attenuation were obtained by measuring *in situ* light intensities with a Kahlsico submarine photometer (Model No. 278WA300) at one meter intervals to a depth of one percent of incident surface radiation. This depth determined the vertical extent of the euphotic zone (Verduin, 1964). Extinction coefficients were computed as described by Hutchinson (1957). Secchi disk visibility was determined as outlined by Welch (1948).

Phytoplankton Standing Crop, Chlorophyll a and Primary Productivity

A 250 ml subsample of each euphotic zone composite was preserved with Lugol's solution (A.P.H.A., 1976) and used to determine phytoplankton cell volume-counts per unit volume of water utilizing the sedimentation method described by Schwoerbel (1970). Phytoplankton were identified to species, whenever possible, using the taxonomic keys of Hustedt (1930), Smith (1950), Prescott (1962), Weber (1971), and Patrick and Reimer (1966; 1975).

Euphotic zone chlorophyll a concentrations were determined by filtering (0.45 micron Millipore® filters) a known volume (usually 500 ml) of the composite followed by extraction. Acetone (90%) was used as the extraction solvent and chlorophyll a concentrations were determined as outlined by A.P.H.A. (1976).

The method of Ryther and Yentsch (1957) as modified by Martin (1967) was used to estimate primary productivity. Solar radiation data required for the calculation of primary productivity were obtained from the National Weather Service at Spokane International Airport. Established monthly mean solar radiation values were used in the above calculations since daily solar data collection by the weather service was terminated in February 1977.

Zooplankton Standing Crop

Bottom to surface oblique tows were made with a Clarke-Bumpus sampler to obtain quantitative zooplankton samples. Collections were preserved with formaldehyde (final solution concentration of 3.7%) and stained with 1.0 ml of saturated Eosin Y-ethanol solution and 0.5 ml of Lugol's solution.

Zooplankton were identified to species, whenever possible, according to Edmondson (1959) and Ruttner-Kolisko (1974), and enumerated as outlined by Edmondson and Winberg (1971). Organisms from subsamples of 2 ml were counted in a modified rotary counting chamber (Ward, 1955). Depending on zooplankton density, successive subsamples were counted until either 100 organisms or ten 2 ml aliquots had been examined. At least three subsamples were counted, and total numbers from each were analyzed using a chi-square test to insure homogeniety.

Hydrology

Stage, inflow, total discharge and storage records for the reservoir were supplied by the Washington Water Power Company. Water retention times for the reservoir were computed by dividing the mean storage for a month by the mean daily outflow for that month.

RESULTS AND DISCUSSION

Spokane River and Its Tributaries

Hydrology

Fluctuations in the discharge of the Spokane River and its tributaries were evident during the various sampling periods (Table 2). The least variation in discharge occurred for S.E. with values ranging from 1.28 to 1.85 m^3 sec^{-1}.

Maximum total monthly discharge for the Spokane River occurred during May (Table 3). Peak flows for H.C. and L.S.R. occurred in February and March, respectively. Minimal discharge occurred for all stations in late summer or early fall. During the study, the S.E. comprised approximately two percent of the flow below its point of outfall.

Water discharged from the reservoir was through the power penstocks except for period of high inflow (during April and May), when water was also released over the spillways. Spokane River flows accounted for about 93 percent of the total inflow to Long Lake during the study.

Analysis of variance (ANOV) and least significant difference (LSD) statistical tests were used to determine if the total monthly discharges for the Spokane River at Spokane during 1979 were significantly different from those of 1972, 1973, 1974, 1975, 1977 and 1978 (U.S.G.S., 1973; 1974; 1975; 1976; 1977; 1978; 1979; Unpublished Provisional Data). Except for the higher discharges in 1974, river flows in 1979 were not

Table 2. Mean discharge (m^3 sec^{-1}) for the established stations on the Spokane River and its tributaries during the various sampling periods (1979).

Date	H.C.	F.W.	S.E.	S.M.	N.M.	L.S.R.	DAM
1/5	0.65	54.4	1.44	55.8	77.3	3.68	125.7
2/2	0.65	52.7	1.47	54.2	66.2	3.40	108.7
3/2	21.90	141.3	1.63	142.9	159.4	11.50	169.8
4/6	5.89	243.0	1.28	244.3	261.5	10.40	275.4
5/4	4.64	615.9	1.39	617.3	614.5	9.48	624.0
6/8	0.82	320.6	1.41	322.0	341.4	4.44	353.5
6/25	0.65	105.6	1.36	107.0	129.2	3.51	136.7
7/9	0.34	68.0	1.54	69.5	81.3	3.28	89.1
7/23	0.24	43.8	1.47	45.7	58.5	2.89	64.8
8/6	0.19	28.0	1.48	29.5	44.0	2.69	65.9
8/20	0.23	23.7	1.50	25.2	36.9	2.97	30.8
9/4	0.25	52.6	1.60	54.2	63.4	3.68	70.5
9/17	0.17	53.4	1.43	54.8	71.7	3.28	82.1
10/1	0.20	44.1	1.36	45.5	69.5	3.23	71.3
11/5	0.42	60.1	1.16	61.3	76.2	4.19	77.0
12/3	0.62	57.0	1.85	58.9	76.9	4.30	75.6

Table 3. Total monthly discharge (m^3 sec^{-1}) for the established stations on the Spokane River and its tributaries (1979).

Month	H.C.	F.W.	S.E.	S.M.	N.M.	L.S.R.	DAM
Jan.	21.1	1694	43.0	1737	2360	109.7	3127
Feb.	1098.0	3212	56.5	3269	3693	230.6	4021
Mar.	797.5	7967	46.6	8014	8211	483.4	7751
Apr.	216.5	9895	38.8	9934	10294	306.5	10604
May	153.5	20507	49.8	20557	20607	285.1	20903
June	22.6	5472	41.6	5514	6137	123.1	7444
July	9.8	1685	44.1	1729	2169	97.4	2293
Aug.	6.5	923	45.6	969	1378	87.7	1443
Sept.	6.1	1469	43.0	1512	1923	100.2	2016
Oct.	9.4	1577	42.2	1619	1996	109.7	2159
Nov.	12.1	1729	35.0	1764	2198	117.2	2351
Dec.	28.8	2062	40.7	2103	2569	150.0	2693

significantly different from the other study years (Soltero et al., 1973; 1974; 1975; 1976; 1978; 1979).

Analysis of variance and LSD were also used to determine if the total monthly discharges in 1979 for H.C., S.E. and L.S.R. significantly varied from those of previous study years. The H.C. flows in 1979 were not significantly different from the other study years (U.S.G.S., 1973; 1974; 1975; 1976; 1977; 1978; 1979; Unpublished Provisional Data). Discharges of S.E. were significantly less in 1979 than (P = 0.05) those observed in 1972, 1973, 1974 and 1975 (Unpublished Treatment Plant Records). With the exception of the higher monthly discharges in 1974, L.S.R. flows were not significantly different from those of previous study years (U.S.G.S., 1973; 1974; 1975; 1976; 1977; 1978; 1979; Unpublished Provisional Data).

In addition, ANOV and LSD analyses showed no significant differences between 1979 and other study years in the total monthly discharges of the mainstem Spokane River sampling stations (F.W., S.M., N.M. and DAM), June through November, a period of time common to all studies (Soltero et al., 1973; 1974; 1975; 1976; 1978; 1979).

Water Chemistry

Table 4 presents the range and mean of the determined water quality parameters for the Spokane River and its tributaries. Calcium and bicarbonate were the principle cation and anion, respectively, for both H.C. and L.S.R. Nitrate was the major form of nitrogen in both tributaries with their mean concentrations exceeding all other river sampling stations. Particulate phosphate was the predominant phosphorus fraction in H.C., while particulate and orthophosphate were the major

Table 4. Range and mean of the determined water quality parameters for each of the established sampling stations on the Spokane River and its tributaries (1/5/79-12/3/79).

	H.C.	F.W.	S.E.	S.M.	N.M.	L.S.R.	DAM
Ca^{+2} (me l^{-1})	0.72-2.35 1.86	0.33-1.20 0.72	1.41-2.73 2.11	0.36-1.40 0.86	0.36-1.36 0.86	1.28-1.88 1.57	0.33-1.22 0.87
Mg^{+2} (me l^{-1})	0.35-1.17 0.82	0.16-0.78 0.38	0.58-1.32 1.07	0.13-0.73 0.41	0.14-0.72 0.40	0.58-1.02 0.90	0.12-0.80 0.49
$Na^{+} + K^{+}$ (me l^{-1})	0.31-1.07 0.46	0.05-0.20 0.13	1.96-5.71 3.30	0.11-0.64 0.22	0.14-1.43 0.35	<0.01-2.12 0.28	0.08-0.60 0.22
HCO_3^- (me l^{-1})	0.73-3.31 2.69	0.43-1.68 0.97	1.15-4.15 2.42	0.42-1.79 1.09	0.42-1.83 1.12	1.84-2.48 2.28	0.54-1.81 1.23
SO_4^{-2} (me l^{-1})	0.13-0.45 0.31	0.09-0.22 0.16	0.66-1.75 1.15	0.11-0.27 0.19	0.11-0.27 0.19	0.15-0.29 0.24	0.09-0.24 0.18
Cl^- (me l^{-1})	0.04-0.59 0.21	0.01-0.14 0.04	0.96-3.17 1.75	0.01-0.87 0.13	0.02-0.36 0.10	0.07-0.19 0.09	0.03-0.17 0.08
$NO_3^- \cdot N$ (mg l^{-1})	0.24-4.29 1.27	0.06-1.16 0.37	0.25-11.4 5.1	0.06-1.23 0.59	0.07-1.23 0.58	0.35-1.93 1.06	0.10-1.34 0.52
$NO_2^- \cdot N$ (mg l^{-1})	<0.001-0.019 0.005	<0.001-0.004 0.001	<0.001-0.163 0.04	<0.001-0.004 0.001	<0.001-0.003 0.001	<0.001-0.004 0.001	<0.001-0.008 0.002
$NH_3 \cdot N$ (mg l^{-1})	0.05-0.32 0.12	0.01-0.14 0.06	0.11-19.6 3.61	0.04-0.20 0.08	0.02-0.37 0.09	<0.01-0.16 0.07	0.03-0.21 0.08
Soluble Organic $NH_3 \cdot N$ (mg l-1)	0.11-0.86 0.30	0.01-0.23 0.11	0.01-3.00 0.52	0.01-0.53 0.12	0.01-0.53 0.12	<0.01-0.15 0.06	<0.01-0.11 0.05

Table 4.--(Continued)

	H.C.	F.W.	S.E.	S.M.	N.M.	L.S.R.	DAM
Particulate $NH_3 \cdot N$ (mg l^{-1})	0.04-0.53 0.24	0.01-0.26 0.09	0.10-7.50 0.91	0.01-0.49 0.15	<0.01-0.12 0.05	<0.01-0.49 0.10	0.01-0.43 0.10
Ortho-PO_4^{-3} (mg l^{-1})	<0.01-0.37 0.08	<0.01-0.08 0.02	0.16-1.95 0.80	<0.01-0.14 0.05	<0.01-0.08 0.05	<0.01-0.13 0.05	<0.01-0.12 0.03
Soluble Organic PO_4^{-3} (mg l^{-1})	<0.01-0.13 0.03	<0.01-0.04 0.01	<0.01-0.72 0.23	<0.01-0.21 0.02	<0.01-0.05 0.01	<0.01-0.06 0.01	<0.01-0.05 0.01
Particulate PO_4^{-3} (mg l^{-1})	<0.01-0.42 0.09	<0.01-0.14 0.04	0.08-2.07 0.90	<0.01-0.10 0.04	<0.01-0.16 0.05	<0.01-0.18 0.05	<0.01-0.29 0.06
Turbidity (N.T.U.)	1.3-68 7.6	0.7-11.0 2.4	1.5-4.6 2.6	0.5-12.0 2.3	0.4-14.0 2.5	0.5-16.5 3.6	0.4-22.0 2.8
Silica (mg l^{-1})	9.2-34.5 24.7	7.3-12.8 8.9	11.7-21.8 17.9	8.7-13.5 9.9	7.4-13.7 9.7	10.2-21.8 15.1	6.5-13.3 9.2
Conductance (micromhos cm^{-1})	141-385 314	60-202 125	423-818 625	63-231 147	64-236 153	213-280 264	70-225 156
pH Range	7.31-8.51	7.34-8.28	6.84-7.74	7.44-8.31	7.45-8.60	7.74-8.34	7.36-8.01
Temperature (°C)	1.0-17.4 10.5	0.2-17.1 10.9	8.4-19.5 14.8	0.7-17.8 11.3	0.3-18.1 11.2	2.3-14.2 9.7	2.1-19.7 12.7
Dissolved Oxygen (mg l^{-1})	7.2-14.1 9.9	7.9-13.4 10.3	8.1-10.1 9.0	8.1-13.4 10.2	8.4-13.1 10.3	8.2-11.7 9.5	5.0-12.5 8.6

forms in L.S.R. The nutrient rich character of both tributaries probably had minimal influence on nutrient levels in the Spokane River due to extensive dilution by the river. Dissolved oxygen concentrations for both tributaries always exceeded 7.0 mg l^{-1} throughout the study.

Sodium plus potassium and bicarbonate were the principle ions of the S.E. Mean sulfate concentrations exceeded 1.0 me l^{-1}. Since the initiation of AWT and the use of alum in the treatment process, sulfate concentrations in the effluent have increased from approximately 0.6 to 1.3 me l^{-1}. As in 1978 (Soltero et al., 1979), nitrate and ammonia were determined to be the major nitrogen fractions in the effluent during 1979. Prior to AWT, the predominant forms were ammonia and particulate nitrogen (Soltero et al., 1973; 1974; 1975; 1976; 1978). The above change in the nitrogenous composition of the S.E. reflects the efficacy of the activated sludge process and solids removal at the new plant. Particulate and orthophosphate were the predominant fractions of the effluent in 1979, but were substantially lower in concentration than before AWT. Dissolved oxygen concentrations of the effluent always exceeded 8 mg l^{-1}.

Comparison of the determined parameters for F.W. (above the sewage outfall) and S.M. (below the outfall) shows that most parameters increased in mean concentration or value following the introduction of S.E. into the river (Table 4). Total inorganic nitrogen ($NO_3^- \cdot N + NO_2^- \cdot N + NH_3 \cdot N$) and orthophosphate concentrations at S.M. were 1.6 and 2.5 times higher, respectively, than at F.W. Similar downstream nitrogen increases had been observed prior to AWT (Soltero et al., 1973; 1974; 1975; 1976; 1978), but orthophosphate concentrations were usually 13 times greater at S.M. than those determined at F.W.

The chemical composition at N.M. was similar to that of S.M. throughout the investigation. This similarity has also been reported in the past (Soltero et al., 1973; 1974; 1975; 1976; 1978; 1979).

Overall, water discharged from Long Lake Dam was slightly higher in dissolved solids, as indicated by conductivity, than that of N.M. All of the major cations and anions appeared to behave in a conservative manner with little change in concentration. The downstream dilution of sodium plus potassium and chloride may have been due to rainfall and/or groundwater infiltration. Mean concentrations of inorganic nitrogen and orthophosphate entering the reservoir during the study from N.M. was nearly the same as that leaving. The most notable change in nitrogen was a decrease in the mean soluble organic fraction.

Estimates of the mean daily phosphate and nitrogen loads in S.E., June through November and prior to and following AWT, are presented in Tables 5 and 6. Overall, mean daily total and orthophosphate loads have decreased approximately 89 and 92 percent, respectively, following AWT. Likewise, overall mean daily total and inorganic nitrogen loads have decreased 82 and 18 percent, respectively. An ANOV and LSD analyses showed that mean daily total and orthophosphate loads in 1979 were significantly lower (P = 0.01) than those prior to AWT, but not different from those determined in 1978. Mean daily total nitrogen loads for 1979 were also found to be significantly less (P = 0.01) than those before AWT Mean daily total inorganic nitrogen loads in 1979 were significantly lower (P = 0.05) than those before AWT except for 1977. The 1979 daily inorganic nitrogen loads were also significantly lower (P = 0.01) from those determined in 1978.

Table 5. Sewage effluent mean daily load (metric tons) of total phosphate (TP) and orthophosphate (TIP) before (1972-1977) and after AWT (1978-1979) for June through November.

	June	July	Aug.	Sept.	Oct.	Nov.	June	July	Aug.	Sept.	Oct.	Nov.
			TP						TIP			
1972	1.36	1.79	1.66	1.84	1.82	1.60	0.85	1.23	1.05	1.38	1.46	1.24
1973	2.11	2.02	1.86	1.77	1.86	1.21	1.39	1.45	1.44	1.33	1.34	0.88
1974	0.85	1.31	1.59	1.53	1.70	2.04	0.56	0.85	0.95	0.92	1.10	1.12
1975	1.27	1.76	2.14	1.87	2.15	2.66	0.70	0.97	1.36	1.34	1.65	1.41
1977	1.27	1.00	1.15	1.53	1.11	0.58	0.63	0.59	0.91	1.35	1.09	0.44
$\bar{x}$	1.37	1.58	1.68	1.71	1.73	1.62	0.83	1.02	1.14	1.26	1.33	1.02
1978	0.06	0.04	0.05	0.12	0.21	0.13	0.01	0.01	0.02	0.09	0.15	0.09
1979	0.15	0.25	0.23	0.25	0.45	0.22	0.07	0.09	0.12	0.11	0.16	0.14
$\bar{x}$	0.11	0.15	0.14	0.19	0.33	0.18	0.04	0.05	0.07	0.10	0.16	0.12

Table 6. Sewage effluent mean daily load (metric tons) of total Kjeldahl nitrogen (TN) and total inorganic nitrogen (TIN = $NO_3^- \cdot N + NO_2^- \cdot N + NH_3 \cdot N$) before (1972-1977) and after AWT (1978-1979) for June through November.

	June	July	Aug.	Sept.	Oct.	Nov.	June	July	Aug.	Sept.	Oct.	Nov.
			TN						TIN			
1972	0.97	1.84	1.47	1.64	1.95	1.78	0.97	1.36	1.27	1.27	1.52	1.59
1973	2.08	2.05	2.53	2.59	3.24	1.87	1.55	1.46	1.52	1.67	2.03	1.65
1974	1.30	2.58	1.92	1.68	2.09	2.07	1.03	1.53	1.42	1.31	1.53	1.51
1975	1.45	2.01	2.57	2.02	3.50	3.01	1.07	1.54	2.05	1.63	2.55	1.96
1977	1.11	1.77	1.35	0.80	0.21	0.70	1.31	1.09	1.11	1.02	0.89	0.98
$\bar{x}$	1.38	2.05	1.97	1.75	2.20	1.88	1.19	1.40	1.47	1.38	1.70	1.54
1978	0.61	0.94	0.29	0.54	0.55	0.46	1.21	1.48	1.15	1.14	1.97	1.57
1979	0.07	0.10	0.11	0.09	0.09	0.15	0.64	0.67	0.97	1.23	1.34	0.85
$\bar{x}$	0.34	0.52	0.20	0.32	0.32	0.31	0.93	1.08	1.06	1.19	1.66	1.21

Prior to AWT, the overall mean monthly increase of total and orthophosphate loads at S.M. were estimated to be 64.1 and 46.4 metric tons, respectively (Table 7). Following AWT, increased loads at S.M. declined to 6.4 and 4.5 metric tons month^{-1} of total and orthophosphate, respectively. Likewise, monthly total nitrogen loads at S.M. have decreased from 240 metric tons before AWT to approximately 60 metric tons following AWT.

Tables 8 and 9 show the percent contribution of total phosphate and nitrogen loads to Long Lake from designated sources along the lower Spokane River system during the months of June through November. The S.E. was the major contributor of phosphate to the reservoir prior to AWT with an estimated contribution of 56.7 percent (Table 8). However following AWT, the Spokane River above its confluence with H.C. (F.W.-H.C.) became the primary contributor of phosphate to the reservoir. In 1979, the Spokane River at Spokane contributed 42.8 percent of the reservoir phosphate load while the S.E. contributed approximately 40 percent. The amount of phosphate contributed to the reservoir by either H.C. or L.S.R. has not changed appreciably since 1972.

During 1979, the primary contributor of nitrogen influent to Long Lake was also the Spokane River above its confluence with H.C. (Table 9). The river above H.C. has essentially been the major source of nitrogen to the reservoir since 1972 with an overall mean contribution of 58.4 percent. The S.E. made up only 8.1 percent of the influent nitrogen load to the reservoir in 1979. The contributions to the reservoir from H.C. and L.S.R. have been similar for all years of study. The minus values for the undetermined sources are possibly the result of nitrogen loss (e.g. denitrification) prior to entering the reservoir.

Table 7. Mean monthly loads (metric tons) of total phosphate (TP), orthophosphate (TIP), total Kjeldahl nitrogen (TN) and total inorganic nitrogen (TIN = $NO_3^- \cdot N + NO_2^- \cdot N + NH_3 \cdot N$) at F.W. and S.M. before (1972-1977) and after AWT (1978-1979) during June through November.

	TP	TIP	TN	TIN
		F.W.		
1972	35.5	5.0	250	160
1973	19.2	3.5	44	52
1974	42.1	11.8	186	144
1975	19.6	5.9	169	141
1977	10.6	2.6	62	49
$\overline{x}$	25.4	5.8	142	109
1978	9.9	1.7	71	93
1979	8.6	1.4	40	63
$\overline{x}$	9.2	1.6	56	78
		S.M.		
1972	111.7	48.3	246	245
1973	86.8	60.8	100	95
1974	98.1	59.3	356	202
1975	72.8	34.7	358	149
1977	78.1	57.4	140	148
$\overline{x}$	89.5	52.1	240	168
1978	14.4	4.0	75	137
1979	16.8	8.1	44	109
$\overline{x}$	15.6	6.0	60	123

Table 8. Percent contribution of the total phosphate load to Long Lake, WA. (mean monthly load, metric tons) from the Spokane River (F.W.-H.C.), H.C., S.E., L.S.R. and undetermined sources (Σ Diff.) before (1972-1977) and after AWT (1978-1979) during June through November.

	F.W.-H.C.	H.C.	S.E.	L.S.R.	Σ Diff.	Total
1972	30.9(35.0)	0.4(0.5)	45.2(51.2)	1.4(1.5)	22.1(25.0)	100(113.2)
1973	19.5(17.0)	2.6(2.3)	63.3(55.1)	1.3(1.1)	13.3(11.6)	100(87.1)
1974	43.5(41.6)	0.5(0.5)	48.0(45.9)	1.9(1.8)	6.2(5.9)	100(95.7)
1975	21.3(19.3)	0.4(0.3)	66.4(60.3)	2.8(2.5)	9.3(8.4)	100(90.8)
1977	18.3(10.2)	0.7(0.4)	60.8(33.8)	1.4(0.8)	18.7(10.4)	100(55.6)
$\bar{x}$	26.7(24.6)	0.9(0.8)	56.7(49.3)	1.8(1.5)	13.9(12.3)	100(88.5)
1978	57.0(9.4)	3.3(0.6)	19.0(3.1)	6.1(1.0)	14.7(2.4)	100(16.5)
1979	42.8(8.5)	0.5(0.1)	39.9(7.9)	3.3(0.7)	13.6(2.7)	100(19.9)
$\bar{x}$	49.9(9.0)	1.9(0.4)	29.5(5.5)	4.7(0.9)	14.2(2.6)	100(18.4)

Table 9. Percent contribution of the total Kjeldahl nitrogen load to Long Lake, WA. (mean monthly load metric tons) from the Spokane River (F.W.-H.C.), H.C., S.E., L.S.R. and undetermined sources (Σ Diff.) before (1972-1977) and after AWT (1978-1979) during June through November.

	F.W.-H.C.	H.C.	S.E.	L.S.R.	Σ Diff.	Total
1972	98.7(249.0)	0.6(1.4)	19.5(49.1)	2.4(6.0)	-21.2(-53.3)	100(252.2)
1973	36.4(42.8)	1.0(1.2)	62.2(73.1)	2.3(2.7)	-1.9(-2.2)	100(117.6)
1974	65.7(183.5)	0.9(2.5)	21.2(59.3)	3.4(9.6)	8.8(24.5)	100(279.4)
1975	53.9(167.8)	0.4(1.3)	23.8(74.2)	3.0(9.4)	18.8(58.6)	100(311.3)
1977	56.4(60.8)	0.9(1.0)	28.1(30.3)	3.1(3.3)	11.6(12.5)	100(107.9)
$\bar{x}$	62.2(140.8)	0.8(0.2)	31.0(57.2)	2.8(6.2)	3.2(8.0)	100(212.4)
1978	85.0(69.6)	1.9(1.6)	21.1(17.3)	3.5(2.9)	-11.5(-9.4)	100(82.0)
1979	106(39.5)	1.3(0.5)	8.1(3.0)	4.0(1.5)	-20.0(-7.4)	100(37.1)
$\bar{x}$	95.5(54.6)	1.6(1.1)	14.6(10.2)	3.8(2.2)	-15.8(-8.4)	100(59.7)

Comparison of the influent and effluent mean daily nitrogen and phosphate loads from Long Lake showed the reservoir to be a nutrient sink for all years of study (Table 10). A mean daily retention of 0.61 and 0.54 metric tons of total and inorganic nitrogen occurred in the reservoir during 1979. Overall mean daily loss of total and inorganic nitrogen to the reservoir prior to AWT was 1.30 and 1.11 metric tons, respectively.

The 1979 daily loss of total and orthophosphate to the reservoir was 0.87 and 0.23 metric tons (Table 10). Mean daily loss before AWT was 0.90 and 0.64 metric tons of total and orthophosphate, respectively. Overall, the loss of phosphate to the reservoir had declined substantially following AWT.

Reservoir

Hydrology

Maximum inflow and discharge from Long Lake occurred in April and May (Table 11). Minimum inflows occurred from July through November. Except for the months of January, March and June, storage changes were minimal. Overall mean retention time for the reservoir was calculated to be 33 days (Table 12). The highest retention times occurred during periods of low flow. A maximum value of approximately 75 days occurred in August.

An ANOV of the mean monthly retention times for the period of June through November showed no statistical difference in retention times between 1979 and other study years (Soltero et al., 1973; 1974; 1975; 1976; 1978; 1979).

Table 10. Mean daily influent and effluent Long Lake, WA. loads (metric tons) of total Kjeldahl nitrogen (TN), total inorganic nitrogen (TIN = $NO_3^- \cdot N + NO_2^- \cdot N + NH_3 \cdot N$), total phosphate (TP) and orthophosphate (TIP) before (1972-1977) and after AWT (1978-1979) during June through November.

		TN	TIN	TP	TIP
1972	influent	14.70	8.43	3.06	1.36
	effluent	15.10	7.88	2.15	0.78
1973	influent	6.04	3.84	2.78	1.94
	effluent	4.90	2.83	1.75	1.11
1974	influent	13.50	6.14	2.71	1.35
	effluent	11.70	4.91	2.09	0.84
1975	influent	13.70	6.19	3.26	1.76
	effluent	11.00	4.18	2.17	1.01
1977	influent	6.02	4.14	2.08	1.44
	effluent	4.82	3.42	1.25	0.92
$\bar{x}$	influent	10.80	5.75	2.78	1.57
	effluent	9.50	4.64	1.88	0.93
1978	influent	6.30	5.40	0.61	0.20
	effluent	5.40	4.50	0.47	0.13
1979	influent	5.29	4.40	1.39	0.30
	effluent	4.68	3.86	0.52	0.07
$\bar{x}$	influent	5.80	4.90	1.00	0.25
	effluent	5.04	4.18	0.50	0.10

Table 11. Mean monthly inflow, discharge and storage change for Long Lake, WA. (1979).

Month	Inflow ($m^3 \times 10^6$)	Discharge ($m^3 \times 10^6$)	Storage Change ($m^3 \times 10^6$)
Jan.	213	270	-57
Feb.	339	347	- 8
Mar.	751	670	81
Apr.	916	916	0
May	1805	1806	- 1
June	541	643	-102
July	196	198	- 2
Aug.	127	125	2
Sept.	175	174	1
Oct.	182	187	- 5
Nov.	200	203	- 3
Dec.	235	233	2

Table 12. Monthly mean storage, discharge and water retention times for Long Lake, WA. (1979).

Month	Mean Storage (m^3)	Mean Discharge (m^3 day^{-1})	Retention Time (days)
Jan.	262×10^6	87.14×10^5	30.1
Feb.	208×10^6	124.07×10^5	16.8
Mar.	280×10^6	216.03×10^5	13.0
Apr.	300×10^6	305.40×10^5	9.8
May	300×10^6	582.57×10^5	5.1
June	300×10^6	214.39×10^5	14.0
July	300×10^6	63.92×10^5	46.9
Aug.	300×10^6	40.22×10^5	74.6
Sept.	300×10^6	58.05×10^5	51.7
Oct.	300×10^6	60.17×10^5	49.9
Nov.	300×10^6	67.71×10^5	44.3
Dec.	300×10^6	75.07×10^5	40.0

Temperature

Figure 2 illustrates the thermal regime at stations 0, 1 and 2 located in the lower end of the reservoir. The reservoir was essentially homothermal during April and May. Thermal stratification began in mid-June and was well established by August. The greatest temperature difference between the surface and bottom for station 0 (8.6°C) occurred on 23 July, while maximum values for stations 1 and 2 (8.0 and 7.9°C, respectively) both occurred on 6 August. Breakdown of stratification began in late September and by October the reservoir was again homothermal.

The general sequence of events and thermal conditions associated with the development, occurrence and breakdown of thermal stratification in the reservoir during 1979 did not appear to be appreciably different from conditions described in previous investigations (Soltero _et al_., 1973; 1974; 1975; 1976; 1978; 1979).

Conductivity

Conductivity steadily increased from April to mid-July (Fig. 3). Chemical stratification was apparent in late June and persisted until September. A cell of high conductive water, located at the depth of the power penstocks, was evident from August through September. This internal current, a result of Spokane River interflow, has been previously described as a function of density flow, thermal stratification and seasonal inflow and outflow (Soltero _et al_., 1973; 1974; 1975; 1976; 1978). By October, increased vertical mixing associated with the breakdown of thermal stratification resulted in the establishment of chemical homogeneity throughout the reservoir.

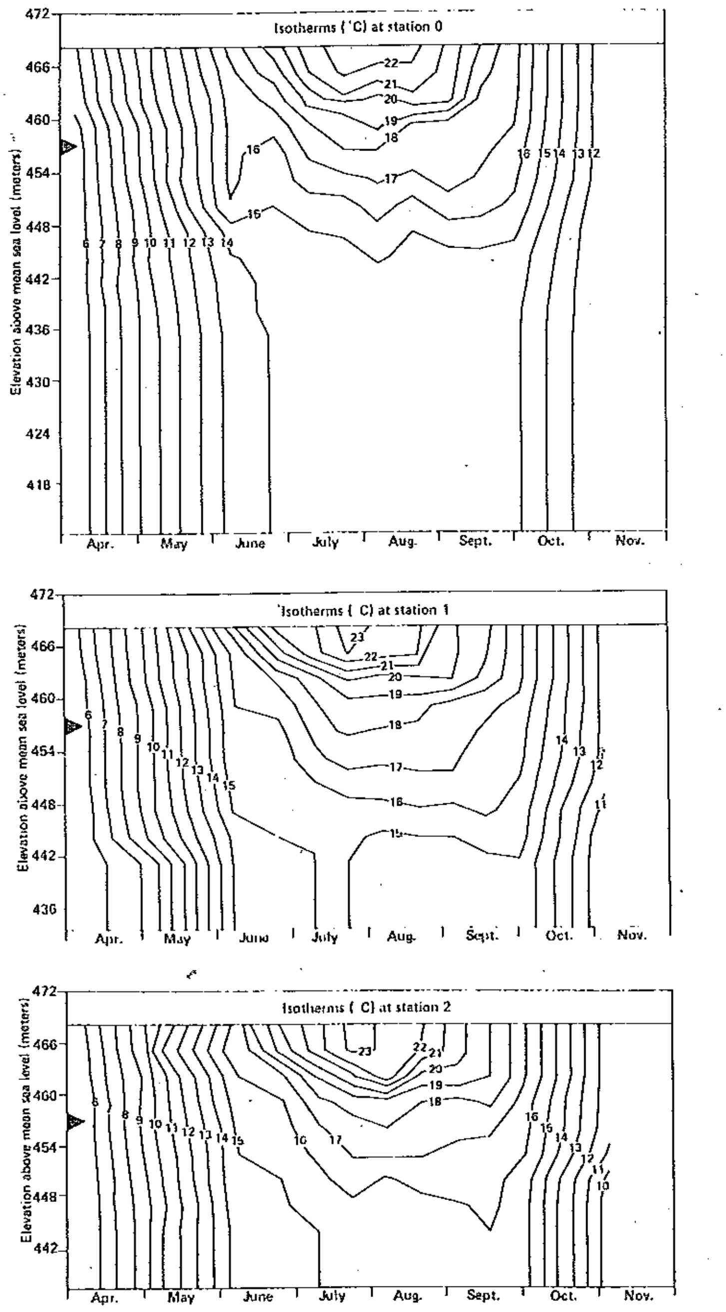

Figure 2. Isotherms (^{0}C) at stations 0, 1 and 2, Long Lake, WA. (1979)

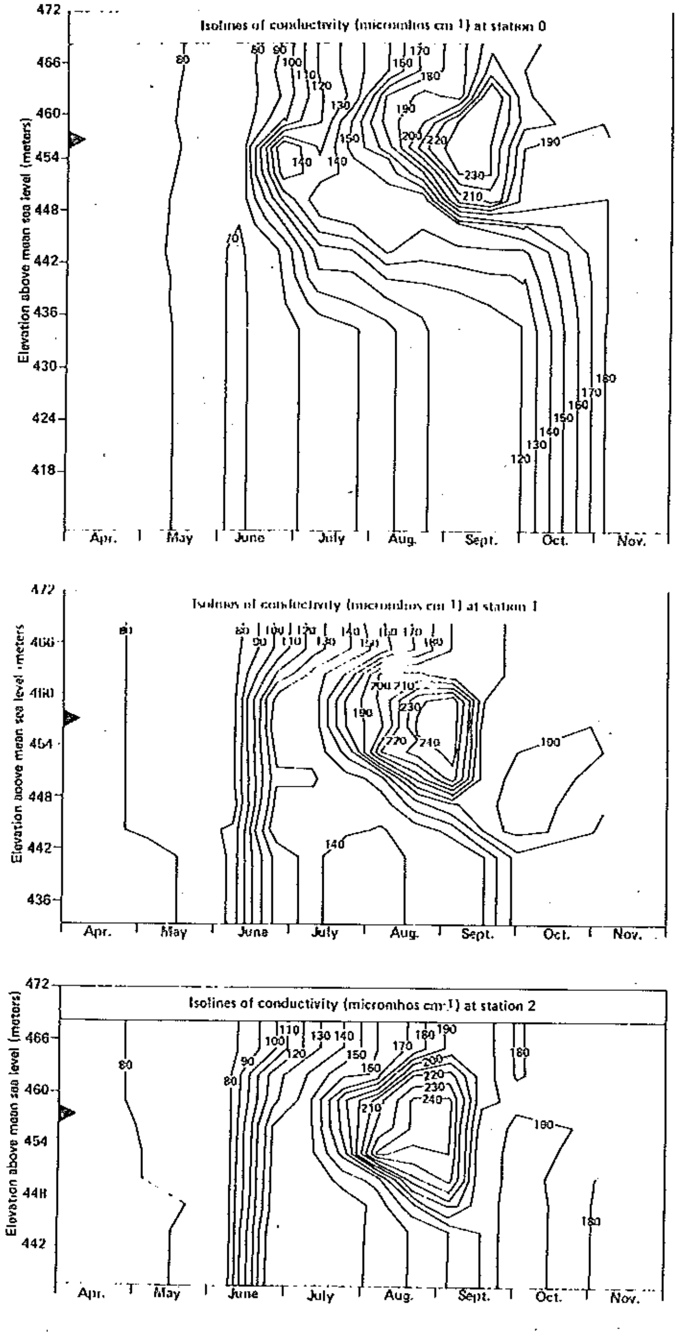

Figure 3. Isolines of conductivity (micromhos cm^{-1}) at stations 0, 1 and 2, Long Lake, WA. (1979)

Water Chemistry

Nitrate nitrogen was the main inorganic nitrogen fraction at all stations (Table 13). Mean concentrations steadily increased down reservoir from a low of 0.32 mg l^{-1} at station 4 to a high of 0.39 mg l^{-1} at station 1. Orthophosphate concentrations ranged from less than 0.01 to 0.34 mg l^{-1} with an overall mean of 0.04 mg l^{-1}. Mean dissolved oxygen, conductance and temperature values usually decreased from station 4 to station 0. Values of pH ranged from a low of 6.0 to a high of 9.5.

Dissolved oxygen concentrations exceeded 9.0 mg l^{-1} throughout the water column from April to early June (Fig. 4). Concentrations in the water mass below the power penstock elevation (hypolimnion) began to decline with the development of thermal stratification in mid-June. Hypolimnetic stagnation reached a maximum in August at station 2 and at stations 0 and 1 in September. The greatest oxygen concentration differential between the surface and the bottom was 11.3 mg l^{-1} at station 1 on 17 September. Surface concentrations generally exceeded 9.0 mg l^{-1} throughout the period of summer stagnation. Following fall overturn in October, dissolved oxygen concentrations exceeded 7.0 mg l^{-1} throughout the reservoir.

The degree of hypolimnetic anoxia in 1979 was similar to that observed in 1978 following AWT (Soltero _et al._, 1979), and that observed before AWT in the higher flow years of 1972, 1974 and 1975 (Soltero _et al._, 1973; 1975; 1976). However, the extent of anoxia during the lower flows of 1973 and 1977 (Soltero _et al._, 1974; 1978) was substantially greater than that observed in 1979.

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Table 13. Range and mean of the determined water quality parameters at each Long Lake, WA. sampling station (1979).

	0	1	2	3	4
$NO_3^- \cdot N$ (mg l^{-1})	0.07-0.95 0.37	0.10-0.98 0.39	0.11-0.90 0.38	<0.01-1.10 0.33	0.09-0.93 0.32
$NO_2^- \cdot N$ (mg l^{-1})	<0.001-0.031 0.003	0.001-0.059 0.006	<0.001-0.064 0.007	<0.001-0.024 0.004	0.001-0.006 0.003
$NH_3 \cdot N$ (mg l^{-1})	<0.01-0.34 0.09	0.01-0.36 0.10	0.01-0.41 0.12	<0.01-0.26 0.11	0.01-0.20 0.10
Ortho-PO_4^{-3} (mg l^{-1})	<0.01-0.12 0.04	<0.01-0.21 0.05	<0.01-0.34 0.05	0.01-0.20 0.04	<0.01-0.09 0.04
Dissolved Oxygen (mg l^{-1})	0.1-13.2 7.2	0.1-13.1 7.6	0.1-13.5 8.1	5.1-13.2 9.6	8.7-14.7 10.9
Conductance (micromhos cm^{-1})	68-244 135	69-342 150	75-243 155	72-250 157	68-247 154
pH Range	5.98-9.14	6.27-9.31	6.00-9.23	7.16-9.26	7.43-9.48
Temperature (°C)	5.2-22.6 14.7	5.2-22.7 15.1	5.0-22.9 15.2	5.7-24.1 16.2	5.9-23.6 16.5

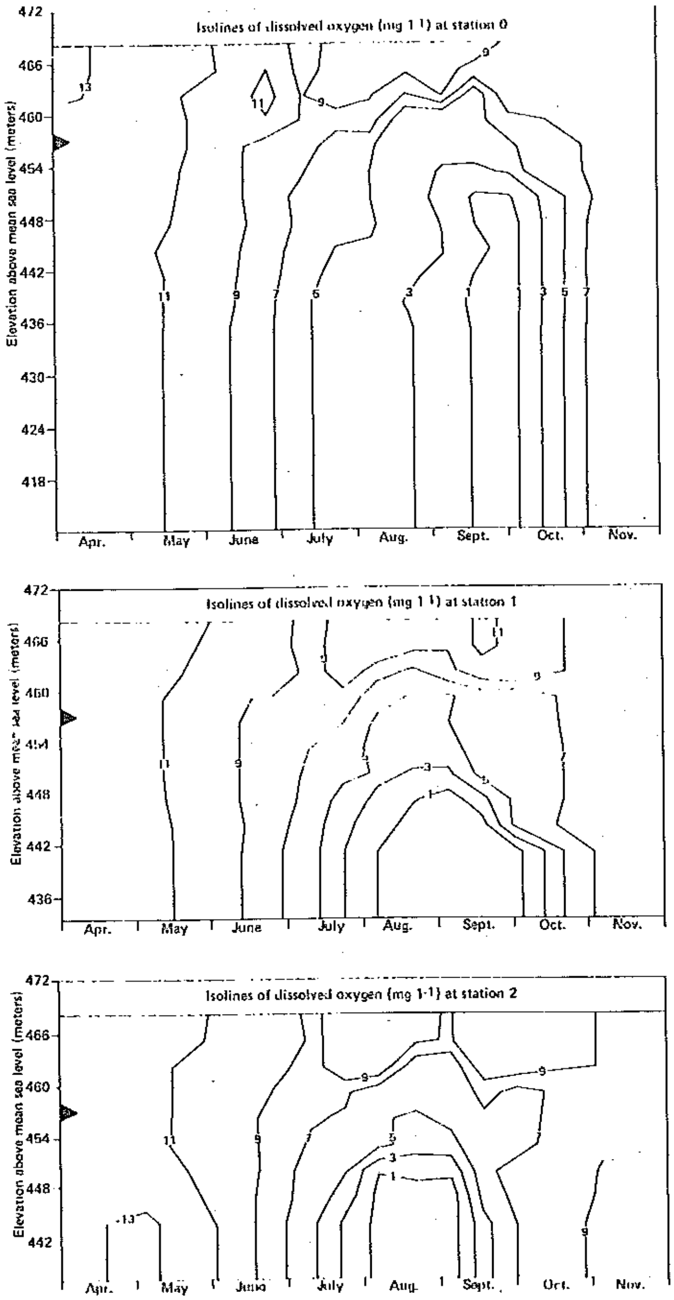

Figure 4. Isolines of dissolved oxygen (mg l^{-1}) at stations 0, 1 and 2, Long Lake, WA. (1979)

Prior to the development of thermal stratification in June, nitrate nitrogen concentrations were less than 0.30 mg l^{-1} throughout the water column (Fig. 5). With the development of anoxia, nitrate nitrogen levels in the hypolimnion increased with concentrations generally exceeding 0.50 mg l^{-1} at stations 1 and 2 and 0.30 mg l^{-1} at station 0. Also during August and September, a cell of nitrate rich water greater than 0.70 mg l^{-1} $NO_3^-\cdot N$ existed at the power penstock elevation. Following fall turnover, nitrate nitrogen concentrations exceeded 0.30 mg l^{-1} throughout the reservoir.

Orthophosphate concentrations were generally less than 0.05 mg l^{-1} throughout the water column prior to the onset of thermal stratification (Fig. 6). During summer stagnation, hypolimnetic concentrations increased with values exceeding 0.10 mg l^{-1} PO_4^{-3}. Maximum hypolimnetic concentrations for stations 0, 1 and 2 were 0.11, 0.21 and 0.34 mg l^{-1}, respectively. Surface orthophosphate concentrations were usually 0.01 mg l^{-1} throughout the growing season. Following fall turnover, concentrations were essentially less than 0.05 mg l^{-1} PO_4^{-3} throughout the reservoir.

A comparison of orthophosphate concentrations at stations 0, 1 and 2 for all study years (Soltero _et al_., 1973; 1974; 1975; 1976; 1978; 1979) showed a hypolimnetic maximum of 2.40 mg l^{-1} in 1973 and a low of 0.29 mg l^{-1} in 1978. In addition, the length of time associated with orthophosphate hypolimnetic maxima during 1978 and 1979 was approximately one month, whereas investigations prior to AWT have shown maxima lasting approximately three months. Following fall turnover in 1978 and 1979 orthophosphate concentrations throughout the water column were

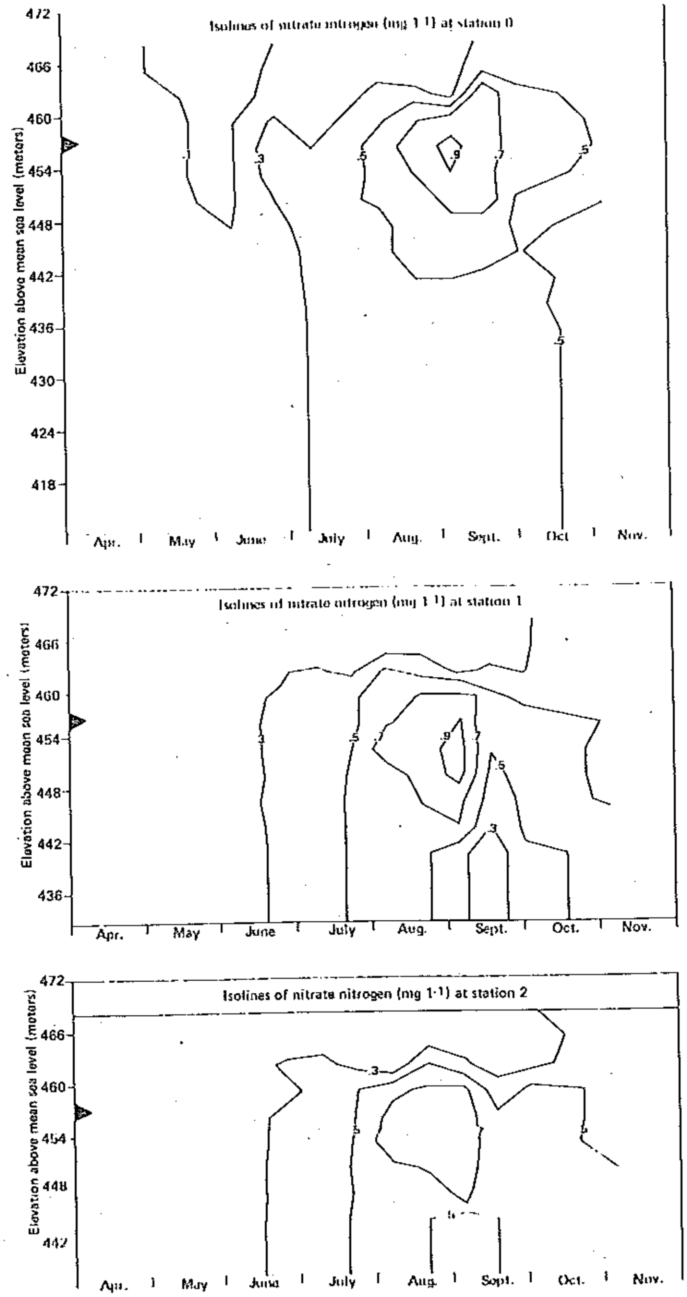

Figure 5. Isolines of nitrate nitrogen (mg l^{-1}) at stations 0, 1 and 2, Long Lake, WA. (1979)

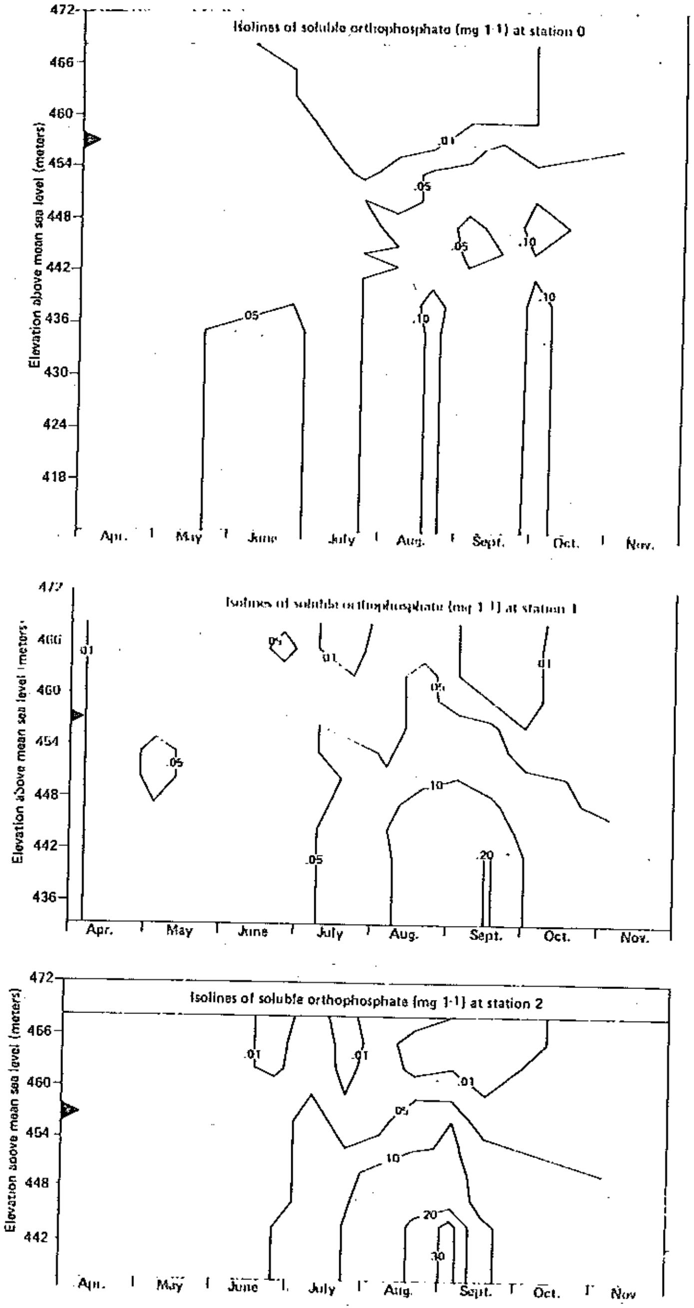

Figure 6. Isolines of orthophosphate (mg 1^{-1}) at stations 0, 1 and 2, Long Lake, WA. (1979)

approximately 0.05 mg l^{-1} while concentrations determined in previous studies were usually greater than 0.10 mg l^{-1}.

Nitrate nitrogen was the principal inorganic nitrogen form in the euphotic zone at all stations (Table 14). Orthophosphate concentrations ranged from less than 0.01 to 0.06 mg l^{-1} with an overall reservoir mean of 0.02 mg l^{-1}. Dissolved oxygen concentrations were usually greater than 7.0 mg l^{-1}. Silica concentrations ranged from 1.4 mg l^{-1} at station 3 to 10.7 mg l^{-1} at station 4. The highest mean euphotic zone depth occurred at stations 1 and 2,while the lowest values occurred at stations 0 and 4.

Prior to AWT, overall mean euphotic zone nitrate nitrogen and orthophosphate concentrations were 0.25 and 0.06 mg l^{-1}, respectively (Table 15). Since AWT, euphotic zone nitrate nitrogen and orthophosphate concentrations have increased and decreased 28 and 67 percent, respectively. Total inorganic nitrogen to orthophosphate concentration ratios (TIN/TIP) for euphotic zone waters prior to AWT were approximately 5:1 (Table 15). However, since the initiation of phosphorus removal at the treatment plant, TIN/TIP ratios have increased to approximately 20:1.

Light

Water clarity progressively improved from station 4 to station 0 (Fig. 7). Mean Secchi disk visibilities for stations 0 and 4 were 3.5 and 2.3 meters, respectively. In addition, turbidity and light extinction decreased down reservoir with a mean turbidity value of 2.9 N.T.U. at station 4 and 1.4 N.T.U. at station 0. Mean extinction coefficient values ranged from 0.97 at station 4 to 0.58 at station 0.

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Table 14. Range and mean of the determined water quality parameters for the euphotic zone at each Long Lake, WA. sampling station (1979).

	0	1	2	3	4
$NO_3^- \cdot N$ (mg l^{-1})	0.08-0.59 0.34	0.11-0.57 0.30	0.11-0.53 0.28	0.10-0.48 0.22	0.13-0.80 0.29
$NO_2^- \cdot N$ (mg l^{-1})	0.001-0.009 0.003	0.001-0.008 0.004	0.001-0.011 0.004	0.001-0.006 0.003	<0.001-0.006 0.003
$NH_3 \cdot N$ (mg l^{-1})	0.02-0.12 0.08	0.02-0.13 0.08	0.04-0.14 0.08	0.05-0.14 0.09	0.05-0.15 0.10
Ortho-PO_4^{-3} (mg l^{-1})	<0.01-0.05 0.02	<0.01-0.04 0.02	0.01-0.04 0.02	<0.01-0.04 0.02	0.01-0.06 0.03
Dissolved Oxygen (mg l^{-1})	6.9-13.2 9.2	7.6-13.0 9.6	7.2-12.9 9.9	8.7-12.5 10.3	9.4-12.3 10.8
Conductance (micromhos cm^{-1})	72-222 148	75-206 150	76-214 148	72-192 146	69-201 153
pH Range	7.26-8.77	7.34-9.14	7.47-9.16	7.43-9.06	7.47-9.08
Temperature (°C)	6.6-20.4 16.6	5.8-21.4 16.8	5.6-22.8 17.0	6.0-23.5 17.5	6.0-22.4 16.6
Turbidity (N.T.U.)	0.7-3.6 1.4	0.9-4.4 1.7	0.9-4.2 1.7	1.4-4.9 2.3	1.5-6.2 2.9
Silica (mg l^{-1})	3.8-9.4 7.1	3.0-9.4 6.8	1.8-9.8 6.6	1.4-10.1 5.9	2.5-10.7 7.1
Euphotic Zone Depth (m)	5.0-13.0 5.1	3.0-13.0 8.4	5.0-12.0 8.1	3.0-9.0 6.8	3.0-7.0 5.4

Table 15. Mean euphotic zone nitrate, nitrite and ammonia nitrogen concentrations; orthophosphate concentrations; and total inorganic nitrogen (TIN = $NO_3^- \cdot N$ + $NO_2^- \cdot N$ + $NH_3 \cdot N$) to orthophosphate (TIP) concentration ratios over all dates and sampling stations prior to (1972-1977) and following AWT (1978-1979) during the period of June through November (Long Lake, WA.).

Date	$NO_3^- \cdot N$ (mg l^{-1})	$NO_2^- \cdot N$ (mg l^{-1})	$NH_3 \cdot N$ (mg l^{-1})	Ortho-PO_4^{-3} (mg l^{-1})	TIN/TIP
1972	0.44	0.012	0.10	0.09	6.1
1973	0.13	0.006	0.05	0.06	3.1
1974	0.24	0.002	0.06	0.05	6.0
1975	0.22	0.005	0.10	0.05	6.5
1977	0.21	0.005	0.16	0.07	5.4
$\overline{x}$	0.25	0.006	0.09	0.06	5.4
1978	0.33	0.007	0.10	0.02	21.9
1979	0.31	0.004	0.09	0.02	20.2
$\overline{x}$	0.32	0.006	0.10	0.02	21.1

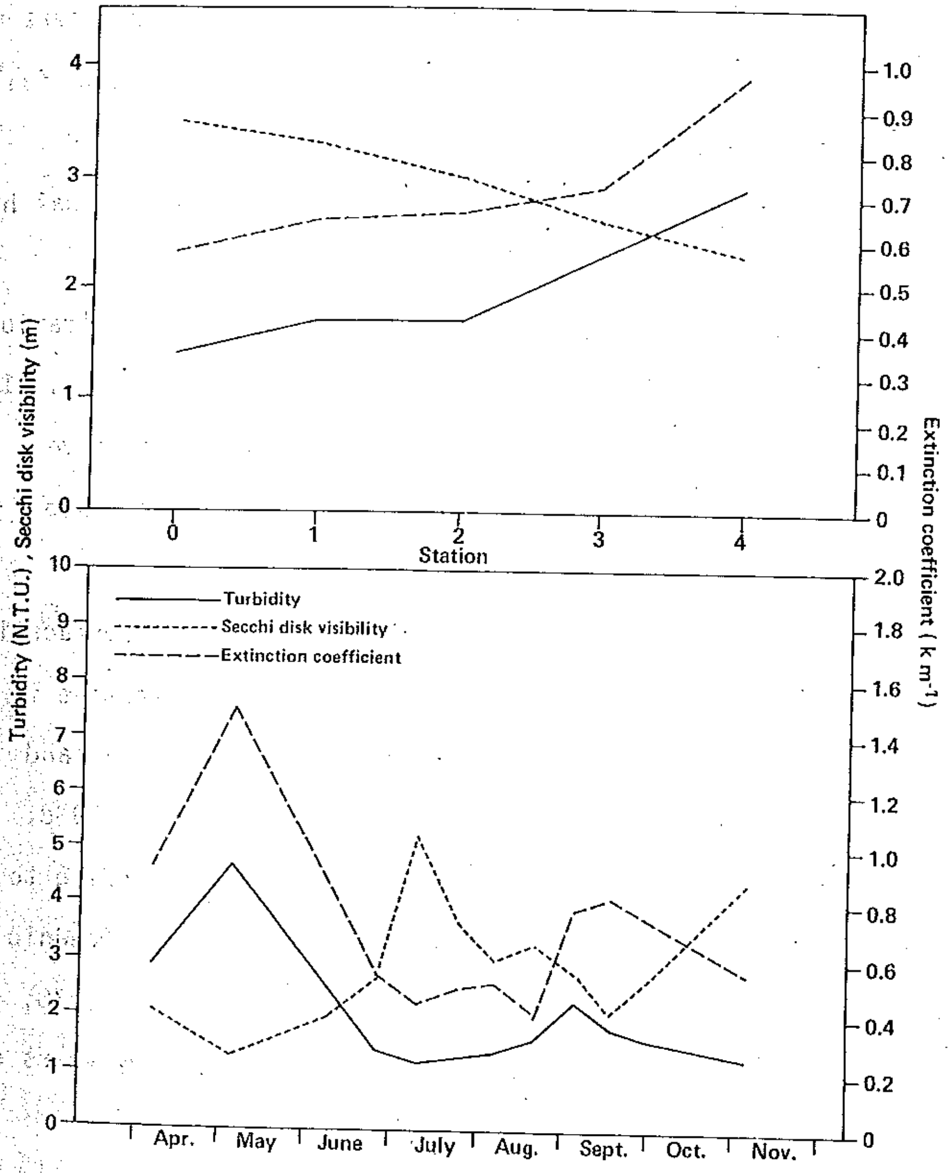

Figure 7. Mean turbidity (N.T.U.), Secchi disk visibility (m) and extinction coefficients (k m^{-1}) by station over all dates and by date over all stations, Long Lake, WA. (1979)

Mean reservoir Secchi disk visibility steadily increased from a seasonal low of 1.3 meters on 4 May to a seasonal high of 5.2 meters on 9 July. Visibility then declined through the summer months until fall turnover in September. In contrast, mean reservoir turbidity and extinction coefficient values tended to decline from their seasonal highs (4.6 N.T.U. and 1.50 k m^{-1}) in May.

Water clarity has previously been found to improve from station 4 to station 0 (Soltero *et al*., 1973; 1974; 1975; 1976; 1978; 1979). In addition, maximum turbidity and extinction coefficient values have occurred during periods of high stream flow in the spring and late summer phytoplankton pulses.

The amount of solar radiation reaching the reservoir surface is primarily dependent upon the degree of cloud cover. To determine if significant differences existed in solar radiation between 1979 and previous study years (Soltero *et al*., 1973; 1974; 1975; 1976; 1978; 1979), an ANOV was made on the monthly average sky cover (sunrise to sunset) taken at the Spokane International Airport (National Oceanic and Atmospheric Administration, 1972; 1973; 1974; 1975; 1977; 1978; 1979). No significant differences were evident between study years on an annual basis or for the period June through November.

Phytoplankton Standing Crop and Chlorophyll a

One hundred species in sixty-seven genera of algae were identified from the euphotic zone composites collected at the five established sampling stations on Long Lake (Table 16).

Table 17 presents the rank of the major phytoplankton species according to mean biovolume and represents approximately 97 percent of

Table 16. Phytoplankton species observed during the study, Long Lake, WA. (1979).

DIVISION: CHLOROPHYTA

Class: Chlorophyceae

Ankistrodesmus falcatus G. M. Smith
Acanthosphaeria Zachariasi Lemmermann
Carteria cordiformis (Carter) Diesing
Carteria sp. Diesing
Chlamydomonas sp. Ehrenberg
Chlorella vulgaris Beyerinck
Chlorella sp. Beyerinck
Chlorogonium elongatum (Dang.)
Chodatella Chodati (Bernard) Ley.
Coccomonas orbicularis Stein.
Coccomonas sp. Stein
Coelastrum microporum Naeg.
Coelastrum reticulatum (Dang.) Senn
Crucigenia rectangularis (Naeg.) Gay
Crucigenia tetrapedia (Kirch.) West & West
Crucigenia sp. Morren
Dictyosphaerium pulchellum Wood
Dictyosphaerium sp. Naegeli
Dysmorphococcus varabilis Takeda
Echinosphaerella limnetica G. M. Smith
Excentrosphaera virdis Moore
Franceia Droescheri (Lemm.) G. M. Smith
Gloeocystis gigas (Kuetz.) Lagerheim

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Table 16.--(Continued)

Golenkinia radiata Chodat

Haematococcus lacustris (Girod.) Rostafinski

Kirchneriella obesa (W. West) Schmidle

Kirchneriella sp. Schmidle

Lagerheimia quadriseta (Lemm.) G. M. Smith

Micractinium pusillum Fresenius

Mougeotia sp. Agardh.

Oocystis sp. Naeg.

Palmellococcus sp. Chodat

Pandorina morum Bory

Pascheriella tetras Korshikov

Pediastrum duplex Meyen

Pediastrum tetras (Ehrenb.) Ralfs

Planktosphaeria gelatinosa G. M. Smith

Quadrigula closterioides (Bohlin) Printz

Quadrigula sp. Printz

Scenedesmus abundans (Kirch.) Chodat

Scenedesmus arcuatus Lemmermann

Scenedesmus bijuga (Turp.) Lagerheim

Scenedesmus dimorphus (Turp.) Kutz.

Scenedesmus incrasatutus Bohlin

Scenedesmus quadricauda (Turp.) de Brebisson

Scenedesmus sp. G. M. Smith

Schroederia Judayi G. M. Smith

Schroederia setigera (Schroed.) Lemmermann

Table 16.--(Continued)

Sphaerocystis Schroeteri Chodat

Staurastrum paradoxum Meyen

Tetradesmus Smithii Prescott

Treubaria crassispina G. M. Smith

Treubaria setigerum (Archer) G. M. Smith

DIVISION: EUGLENOPHYTA

Class: Euglenophyceae

Cryptoglena pigra Ehrenberg

Euglena acus Ehrenberg

Eutreptia sp. Perty

DIVISION: CHRYSOPHYTA

Class: Bacillariophyceae

Asterionella formosa Hassall

Caloneis sp. Cleve

Cyclotella sp. Kutz.

Cymbella ventricosa Kutz.

Fragilaria crotonensis Kitton

Gomphonema acuminatum Ehrenberg

Melosira granulata (Ehr.) Ralfs

Melosira italica (Ehr.) Kutz.

Navicula sp. Bory

Nitzschia sp. Hassall

Rhizosolenia eriensis H. L. Smith

Stephanodiscus sp. Ehrenberg

Synedra rumpuns Kutz.

Table 16.--(Continued)

Synedra sp. Kutz.
Synedra ulna (Nitasch) Ehrenberg
Tabellaria fenestrata (Lyngb.) Kutz.
Class: Chrysophyceae
Chromulina sp. Cienkowski
Chrysochromulina sp. Lackey
Dinobryon divergens Imhof.
Dinobryon sp. Ehrenberg
Kephyrion sp. Pascher
Lagynion sp. Pascher
Mallomonas acaroides Perty
Mallomonas caudata Iwanoff
Mallomonas producta (Zach.) Iwanoff
Mallomonas sp. Perty
Ochromonas sp. Wystozki
DIVISION: PYRROPHYTA
Class: Dinophyceae
Ceratium hirundinella (O.F.M.) Dujardin
Glenodinium pulvisculus (Ehr.) Stein
Glenodinium sp. Stein
DIVISION: CYANOPHYTA
Class: Cyanophyceae
Anabaena circinalis Robenhorst
Anabaena Felisii (Menegh.) Bornet & Flahault
Anabaena flos-aquae (Lyngb.) de Brebisson

Table 16.--(*Continued*)

- *Anabaena oscillarioides* Bory
- *Anabaena* sp. Bory
- *Anabaena spiroides* Klebahn
- *Microcystis aeruginosa* (Kuetz.) Elenkin (cells)
- *Microcystis aeruginosa* (Kuetz.) (colony)
- *Oscillatoria* sp. Vaucher
- *Synechococcus* sp. Nageli

DIVISION: CRYPTOPHYTA

- Class: Cryptophyceae
  - *Chilomonas Paramaecium* Ehrenberg
  - *Cryptomonas ovata* Ehrenberg
  - *Cryptomonas* sp. Ehrenberg
  - *Rhodomonas* sp. Karsten

Table 17. Rank of the major phytoplankton species according to mean biovolume (mm^3 l^{-1}) based on collections from all stations, Long Lake, WA. (1979).

Rank	Taxon	Biovolume
1	Fragilaria crotonensis	2.046
2	Melosira italica	1.549
3	Coelastrum reticulatum	0.185
4	Anabaena spiroides	0.170
5	Anabaena circinalis	0.157
6	Anabaena sp.	0.126
7	Asterionella formosa	0.093
8	Oocystis sp.	0.092
9	Cryptomonas sp.	0.090
10	Melosira granulata	0.089
11	Microcystis aeruginosa (colony)	0.084
12	Rhizosolenia eriensis	0.074
13	Sphaerocystis Schroeteri	0.064
14	Microplankton	0.058
15	Tabellaria fenestrata	0.051
16	Synedra sp.	0.042
17	Staurastrum paradoxum	0.028
18	Rhodomonas sp.	0.023
19	Chlamydomonas sp.	0.022
	Microcystis aeruginosa (cells)	0.022
20	Coelastrum microporum	0.019

the total estimated mean biovolume during the study. The Bacillariophyceae was the dominant class comprising 77.6 percent of the total mean biovolume. *Fragilaria crotonensis* and *Melosira italica* were the primary contributors and had the greatest overall mean biovolumes; 2.046 and 1.549 $mm^3\ l^{-1}$, respectively. *Fragilaria* ranked second in occurrence while *Melosira* ranked ninth (Table 18). *Asterionella formosa* ranked seventh on a biovolume basis but occurred in 77 percent of the samples. *Melosira granulata* and *Rhizosolenia eriensis* ranked tenth and twelfth, respectively, in biovolume and occurred in less than 40 percent of the samples.

The Cyanophyceae accounted for approximately 11 percent of the total mean biovolume. *Anabaena spiroides*, *Anabaena circinalis* and *Anabaena* sp. were the major contributors and ranked fourth, fifth and sixth, respectively, in biovolume. *Anabaena* sp. ranked seventh in occurrence (55 percent) while *A. spiroides* and *A. circinalis* occurred in less than 18 percent of the samples. *Microcystis aeruginosa* ranked seventeenth in occurrence.

The Chlorophyceae was the third major class of algae contributing 8.1 percent of the total mean biovolume. *Coelastrum reticulatum* had a mean biovolume of 0.185 $mm^3\ l^{-1}$ (ranking third) and ranked eleventh in occurrence. *Oocystis* sp. was the second major contributor to chlorophycean biovolume and ranked eighth in both occurrence and biovolume. *Sphaerocystis Schroeteri* ranked thirteenth in biovolume and occurred in approximately 50 percent of the samples.

Cryptomonas sp. and *Rhodomonas* sp. were the dominant members of the Cryptophyceae contributing 2.2 percent to the total mean biovolume. *Cryptomonas* sp. ranked ninth in biovolume and sixth in occurrence.

Table 18. Rank of the major phytoplankton species according to occurrence (%) based on collections from all stations, Long Lake, WA. (1979).

Rank	Taxon	Occurrence
1	Microplankton	100
2	*Fragilaria crotonensis*	90
3	*Rhodomonas* sp.	85
4	*Asterionella formosa*	77
5	*Chlamydomonas* sp.	70
6	*Cryptomonas* sp.	68
7	*Anabaena* sp.	55
8	*Oocystis* sp.	52
9	*Melosira italica*	50
10	*Sphaerocystis Schroeteri*	47
11	*Coelastrum reticulatum*	45
12	*Synedra* sp.	43
13	*Schroederia Judayi*	42
14	*Ankistrodesmus falcatus*	40
15	*Navicula* sp.	33
16	*Rhizosolenia eriensis*	32
17	*Microcystis aeruginosa* (cells)	30
18	*Melosira granulata*	28
19	*Golenkinia radiata*	22
20	*Scenedesmus quadricauda*	18

Rhodomonas sp. ranked only eighteenth in biovolume but occurred in 85 percent of the samples (ranking third).

Microplankton were seen in every sample but accounted for only 1.1 percent of the total biovolume. These were unidentifiable organisms of usually five microns or less in size.

Figure 8 shows the mean phytoplankton biovolumes by class and station over all sampling dates. The Bacillariophyceae dominated the standing crop at all sampling sites with values ranging from a minimum of 3.49 $mm^3 l^{-1}$ at station 4 to a maximum of 4.18 $mm^3 l^{-1}$ at station 3. Maximum standing crops of the Chlorophyceae occurred at stations 1 and 3. The greatest Cyanophyceae standing crop occurred at station 4 due to a fall pulse of A. circinalis and A. spiroides.

Diatoms made up the major portion of the phytoplankton community during the spring and early summer months throughout the reservoir (Fig. 9). Pulses of Chlorophyceae and Cryptophyceae then followed in the summer. The blue-greens then succeeded in early fall with the greatest standing crop occurring at station 4. Following fall turnover, the diatoms regained their dominance with a maximal pulse in early October.

Chlorophyll a concentrations appeared to follow phytoplankton biovolumes better than phytoplankton numbers on both a volume and areal basis (Figs. 10 and 11). Correlation coefficients (Table 19) between various combinations of these parameters showed that, in general, chlorophyll a concentrations gave a reasonable estimate of phytoplankton standing crop. Chlorophyll a concentrations ranged from a low of 2.16 mg m^{-3} at station 1 on 5 November to a high of 31.93 mg m^{-3} on 4 September at station 4.

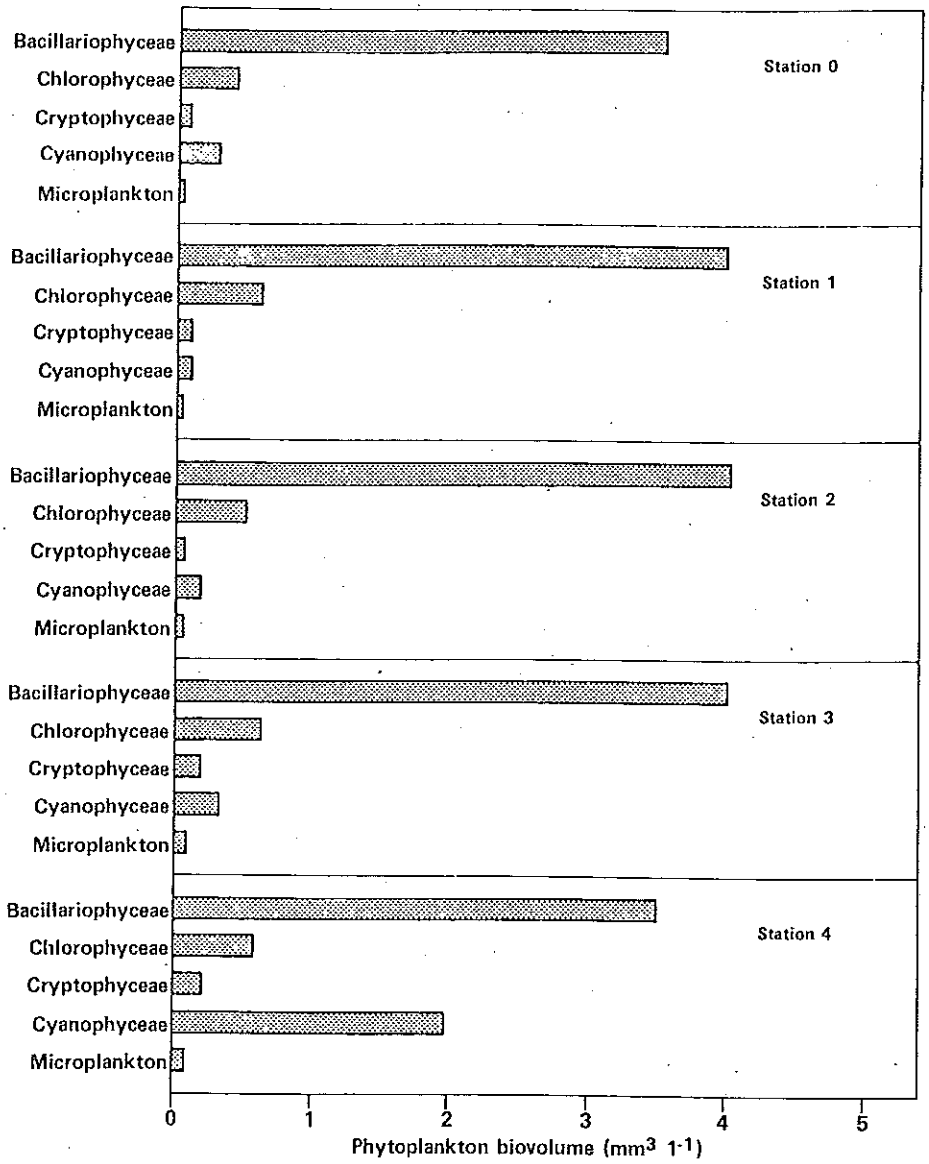

Figure 8. Mean phytoplankton biovolume by class and station over all dates, Long Lake, WA. (1979)

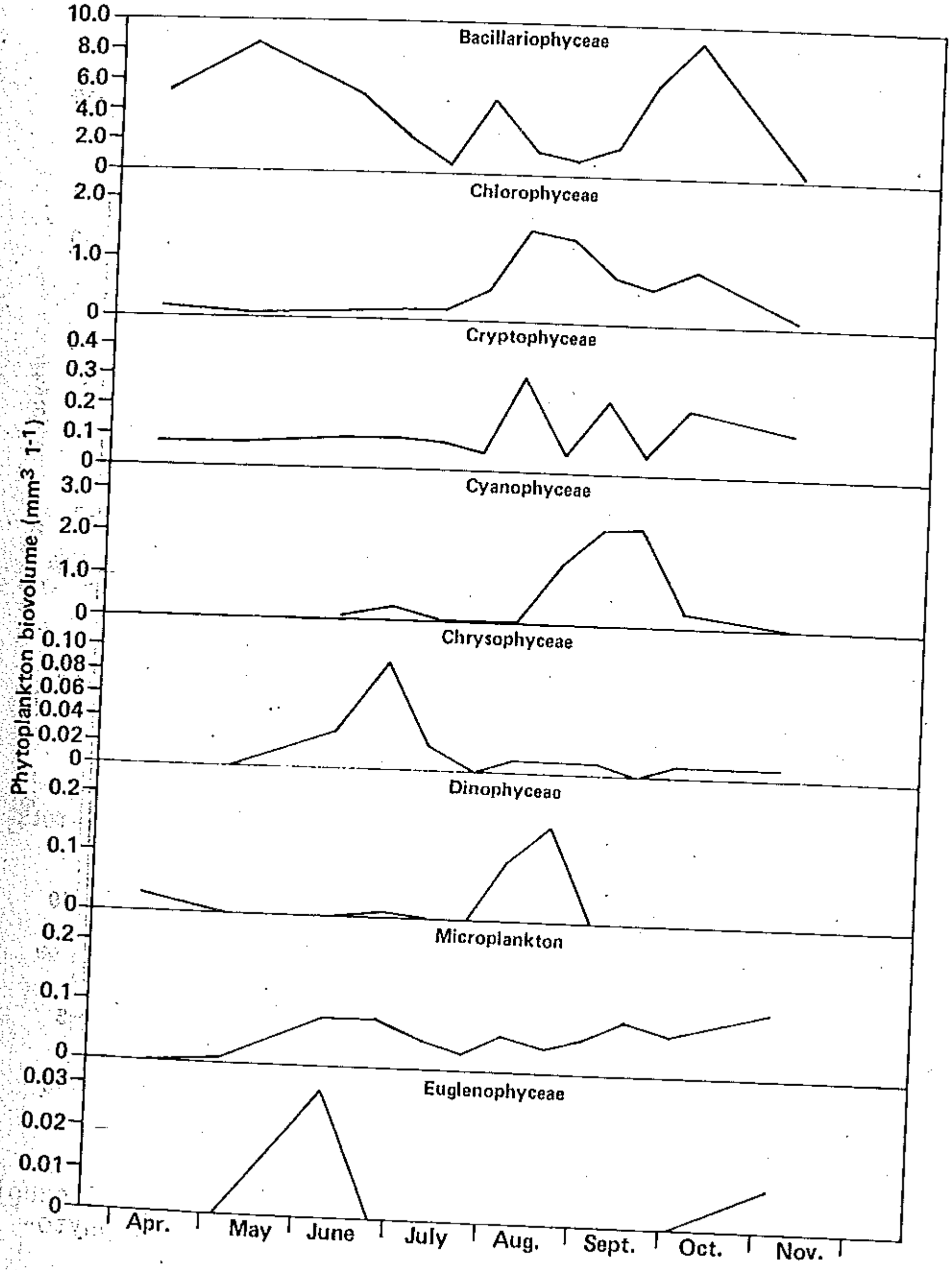

Figure 9. Mean phytoplankton biovolume by class and date over all stations, Long Lake, WA. (1979)

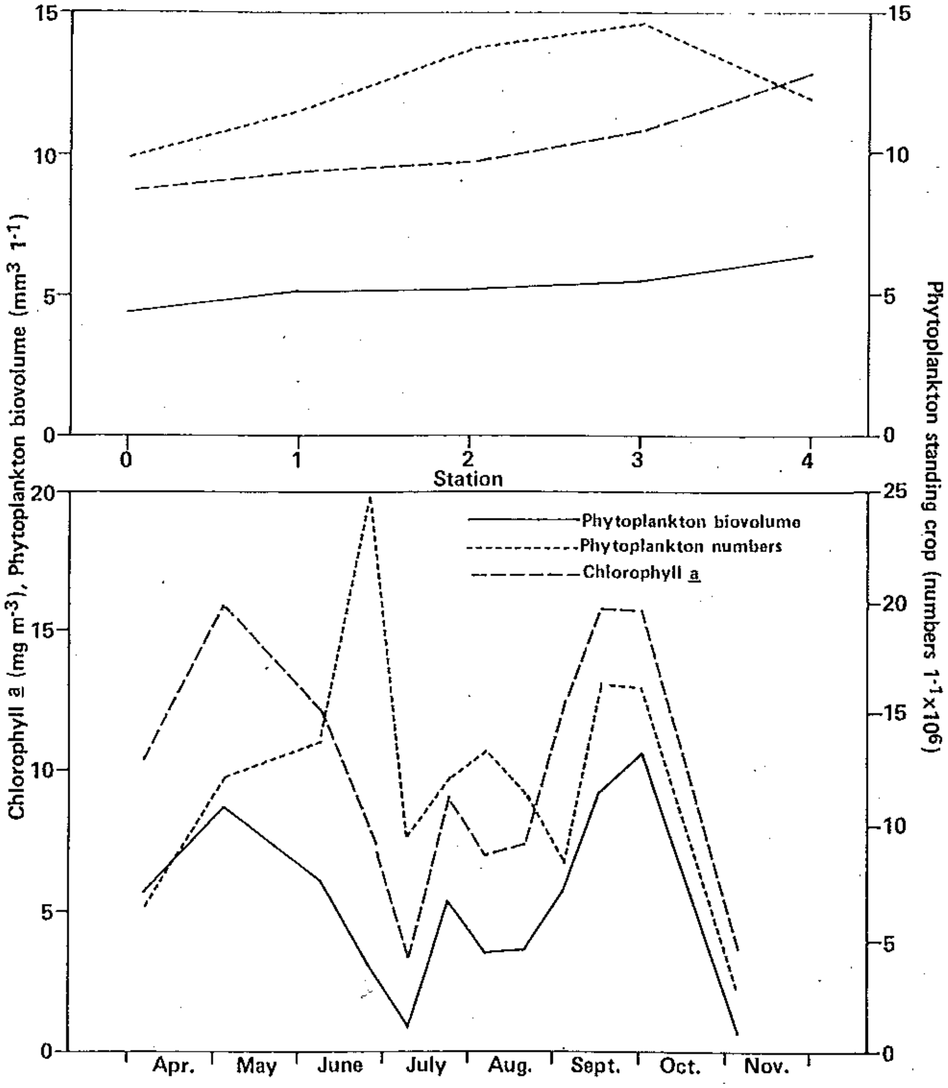

Figure 10. Mean phytoplankton biovolume (mm^3 l^{-1}), phytoplankton numbers (# l^{-1} x 10^6) and chlorophyll a concentrations (mg m^{-3}) by station over all dates and by date over all stations, Long Lake, WA. (1979)

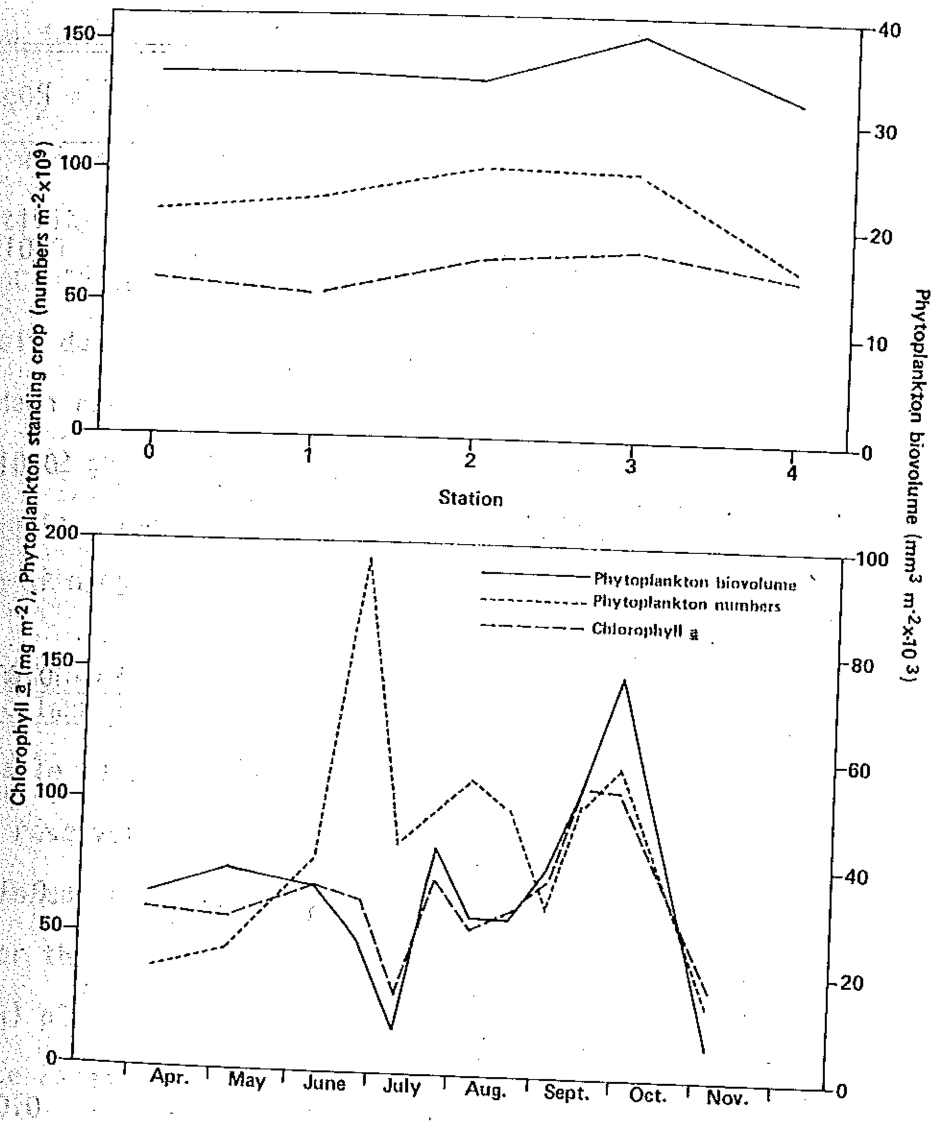

Figure 11. Mean phytoplankton viovolume (mm^3 m^{-2} x 10^3), phytoplankton numbers (# m^{-2} x 10^9) and chlorophyll a concentrations (mg m^{-2}) by station over all dates and by date over all stations, Long Lake, WA. (1979)

Table 19. Correlation coefficients (n = 60) for chlorophyll a concentration versus phytoplankton biovolume versus phytoplankton numbers, Long Lake, WA. (1979).

Variables	r	P
Chlorophyll a (mg m^{-3}) vs. Phytoplankton biovolume (mm^3 l^{-1})	0.83	0.01
Chlorophyll a (mg m^{-2}) vs. Phytoplankton biovolume (mm^3 m^{-2} x 10^3)	0.79	0.01
Chlorophyll a (mg m^{-3}) vs. Phytoplankton numbers (# l^{-1} x 10^6)	0.37	0.01
Chlorophyll a (mg m^{-2}) vs. Phytoplankton numbers (# m^{-2} x 10^9)	0.32	0.05
Phytoplankton biovolume (mm^3 l^{-1}) vs. Phytoplankton numbers (# l^{-1} x 10^6)	0.41	0.01
Phytoplankton biovolume (mm^3 m^{-2} x 10^3) vs. Phytoplankton numbers (# m^{-2} x 10^9)	0.27	0.05

Primary Productivity

Mean daily primary productivity increased from station 0 to station 4 (Fig. 12). The highest mean value of 1.57 gC m^{-2} day^{-1} (4.19 gO_2 m^{-2} day^{-1}) occurred at station 3. A photosynthetic quotient of one was used to convert units of carbon to oxygen. A maximal daily productivity value was also recorded at station 3 on 20 August (3.73 gC m^{-2} day^{-1}; 9.96 gO_2 m^{-2} day^{-1}). The overall mean value for the reservoir during the study was 1.13 gC m^{-2} day^{-1} (3.02 gO_2 m^{-2} day^{-1}).

Mean reservoir productivity increased to a maximum of 2.63 gC m^{-2} day^{-1} (7.02 gO_2 $m^{-2}day^{-1}$) in late July during a pulse of *Fragilaria crotonensis*. Thereafter, productivity declined reaching the seasonal minimum of 0.06 gC m^{-2} day^{-1} (0.16 gO_2 m^{-2} day^{-1}) on 5 November.

Relationship of Areal Phosphate Loading to Phytoplankton Production

Table 20 shows the relationships of total areal phosphate loading to mean reservoir orthophosphate and chlorophyll *a* concentrations, phytoplankton standing crop, and primary productivity for all years of study for the period of June through November. The expression $L\ (1 - R)\ p^{-1}$ relates specific total areal phosphate loading (L) to phosphate retention (R) and flushing rate (p) (Dillon, 1975). Total phosphate loading to Long Lake during 1979 was slightly higher than in 1978 and was attributable to increased mean daily phosphate loading from the sewage effluent (Table 5). Nevertheless, a paired t-test showed that the mean post-AWT (1978, 1979) loading values were significantly lower ($p = 0.01$) than the pre-AWT (1972-1977) values.

Increased phosphate loading in 1979 resulted in a corresponding increase in chlorophyll *a* standing crop and primary productivity.

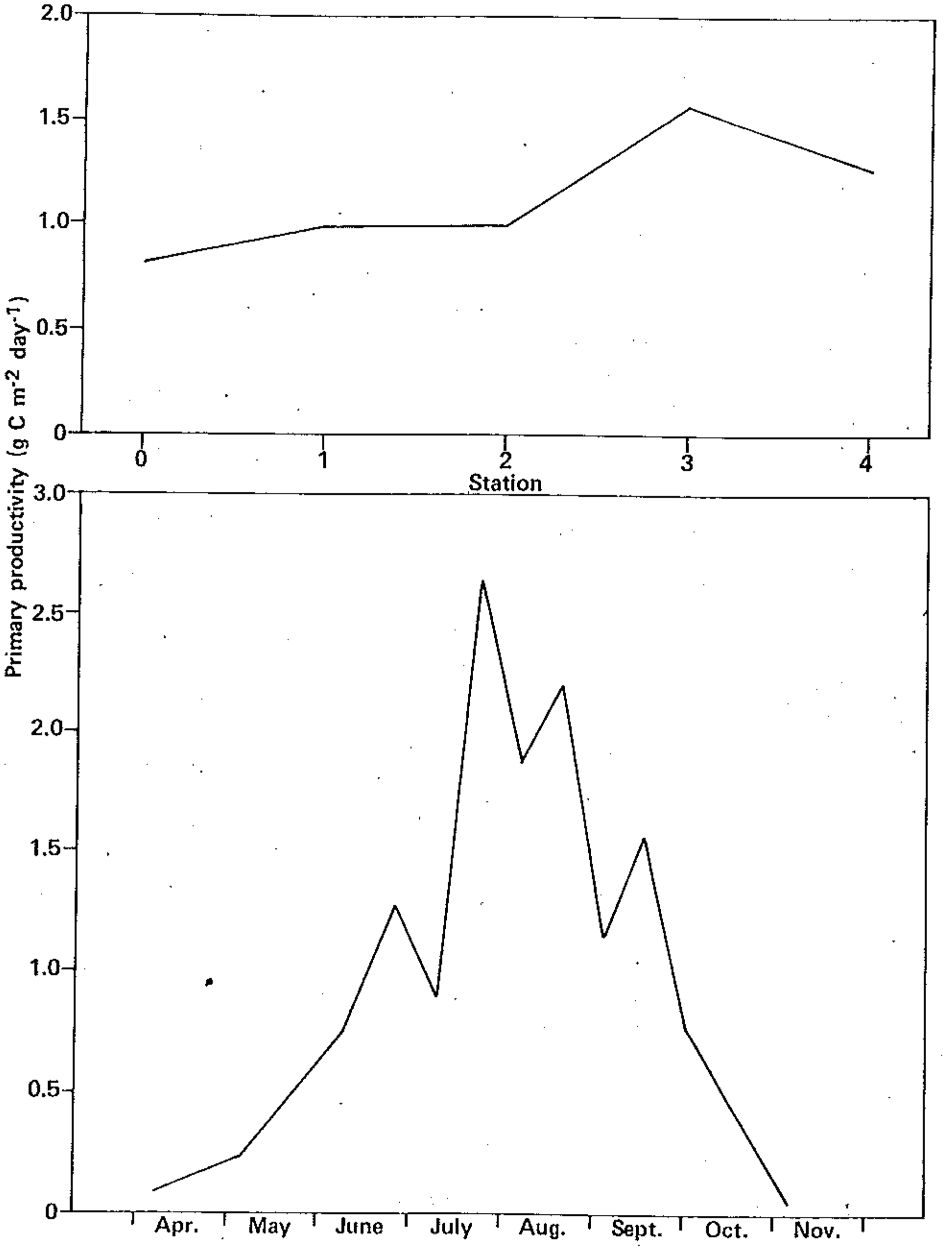

Figure 12. Mean primary productivity (gC m^{-2} day^{-1}) by station over all dates and by date over all stations, Long Lake, WA. (1979)

Table 20. Total areal phosphate loading in relation to mean orthophosphate and chlorophyll a concentrations, phytoplankton biovolume and primary productivity in Long Lake, WA. for all study years during the period of June through November.

Year	$L (1 - R) p^{-1}$ (g $PO_4^{\equiv}$ m^{-2})	Orthophosphate (mg PO_4 l^{-1})	Chlorophyll a (mg m^{-3})	Phytoplankton (mm^3 l^{-1})	Primary Productivity (g C m^{-2} day^{-1})
1972	2.17	0.14	12.34	6.22	1.61*
1973	4.02	0.30	19.86**	10.56**	2.10
1974	1.60	0.11	10.90	6.58	0.82
1975	2.03	0.10	11.87	7.64	1.05
1977	2.90	0.25	14.12	8.90	1.45
$\overline{x}$	2.54	0.18	13.82	7.98	1.41
1978	0.74	0.05	8.79	42.41	0.94
1979	0.88	0.04	9.44	4.92	1.32
$\overline{x}$	0.81	0.05	9.12	23.67	1.13

*July-November
**June-October

Paired t-testing of pre and post-AWT chlorophyll a values showed that post values were significantly lower than pre-AWT values ($P = 0.05$). Despite the increase in phosphate loading during 1979 overall phytoplankton biovolume decreased.

The relationship between phosphorus concentration and phosphorus loading to mean summer chlorophyll a concentrations has been well established (Sakamoto, 1966; Dillon, 1974; Bachmann and Jones, 1974; Vollenweider, 1976). Figure 13 shows the relationship of areal total phosphorus loading to mean chlorophyll a concentration for all years of study, June through November (1973 chlorophyll a, June - October). Total phosphorus loading to Long Lake, using Dillon's (1975) method, appears to be an excellent predictor of mean chlorophyll a concentration. The linear regression equation for the relationship is:

$$y = 9.70\ (x) + 5.99 \qquad R^2 = 0.95$$

where y = mean chlorophyll a concentration during June through November and x = total areal phosphate loading for the same period of time.

By incorporating hydraulic load and mean depth into a calculation of phosphorus loading, Vollenweider (1976) has also found a good correlation ($r = 0.87$) between calculated total phosphorus load values and mean chlorophyll a concentration for 60 lakes. Table 21 shows a comparison of predicted mean chlorophyll a concentrations using Vollenweider's equation, chlorophyll a estimates from the present study regression equation based on Dillon's loading concepts and the actual mean chlorophyll a levels for each year of study. In general, estimates from the study regression equation more closely approximated actual chlorophyll a concentrations. However, paired t-tests showed that no

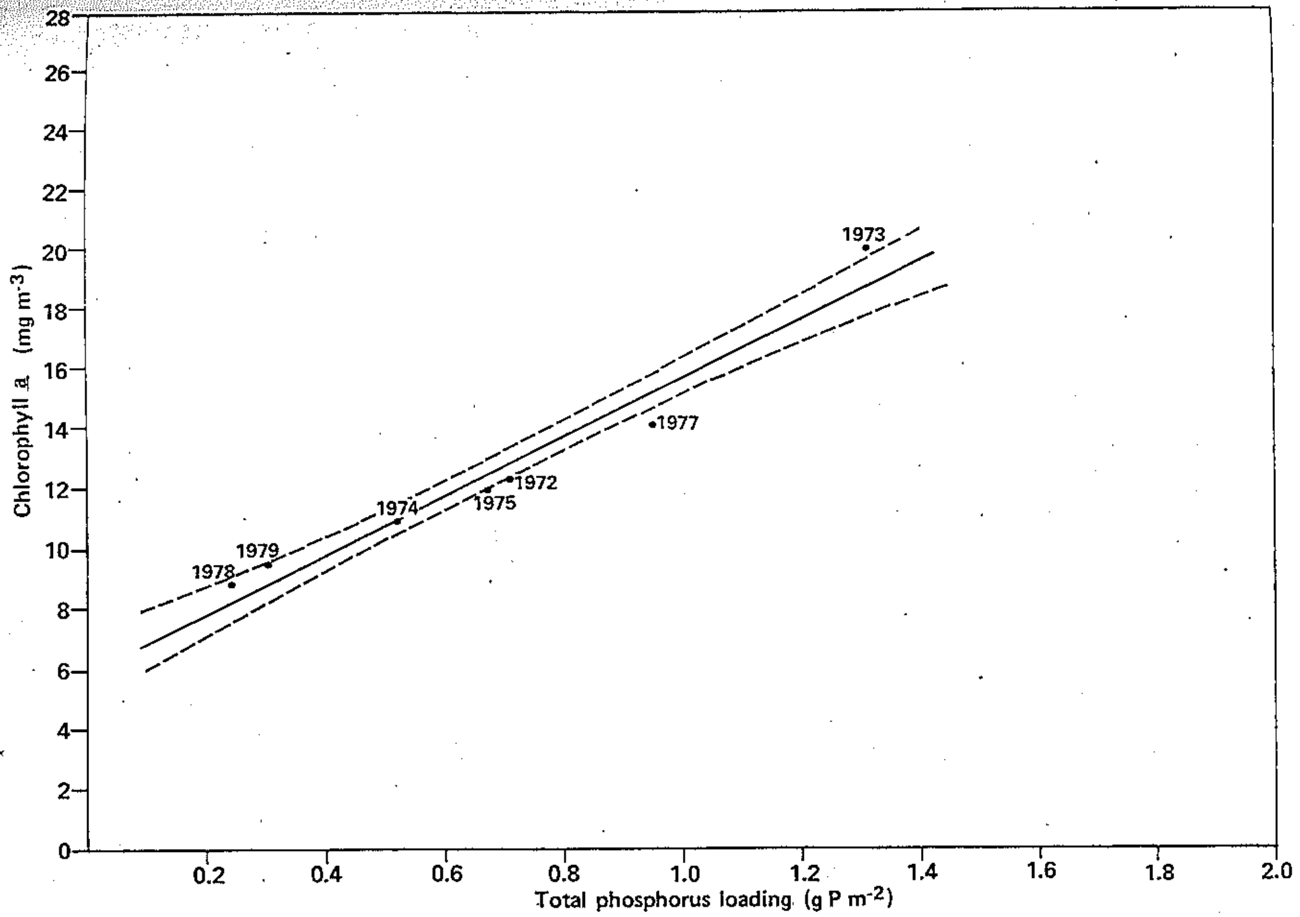

Figure 13. Total areal phosphorus loading (gP m^{-2}) and mean chlorophyll a concentrations (mg m^{-3}) in Long Lake, WA. for all years of study during the period of June through November (1973 chlorophyll a June-October). Dashed lines are the 95 percent confidence interval.

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Table 21. Comparison of predicted chlorophyll *a* concentrations using Vollenweider's (1976) equation, the present study regression equation based on Dillon's (1975) loading concepts and actual mean chlorophyll *a* concentrations in Long Lake, WA. for all study years during the period of June through November.

Year	Vollenweider chlorophyll *a* (mg m^{-3})	Study regression equation chlorophyll *a* (mg m^{-3})	Actual chlorophyll *a* (mg m^{-3})
1972	13.18	12.88	12.34
1973	22.76	18.70	19.86*
1974	9.57	11.03	10.90
1975	13.09	12.39	11.87
1977	18.03	15.21	14.12
$\bar{x}$	15.33	14.04	13.82
1978	4.39	8.32	8.79
1979	9.96	8.80	9.44
$\bar{x}$	7.18	8.56	9.12

*June-October

statistical significant differences exist between either of the estimated or actual values.

In addition to developing an equation to estimate mean chlorophyll a concentrations, Vollenweider (1976) developed an equation for critical phosphorus loading. Critical loading was defined as the maximum allowable total phosphorus load to a system where total phosphorus concentration at spring overturn would be less than 10 mgP m^{-3} but with an upper limit of 20 mgP m^{-3}. With the work of Sakamoto (1966), and Dillon and Rigler (1974) it has become common limnological knowledge to suggest that total phosphorus concentration at spring overturn greater than 10 mgP m^{-3} can promote summer phytoplankton standing crops (as measured by chlorophyll a concentrations) indicative of eutrophic waters. Vollenweider's equation for critical loading was given as:

$$L_c = 10 \text{ to } 20\ (q_s)\ (1 + \sqrt{z/q_s})$$

where L_c is the critical phosphorus loading, q_s is the hydraulic load (= total discharge for the period of time under consideration/lake surface area), and z is the mean depth. The numbers 10 and 20 represent the maximum allowable concentrations of total phosphorus (mgP m^{-3}) at spring overturn. In contrast to critical phosphorus loading, specific surface loading (L_p) does not utilize hydraulic load or mean depth but is merely the relationship between total phosphorus input to a system to lake surface area. Thus, the relationship between critical and specific surface loading appears to be that the critical concentration of total phosphorus at spring overturn is primarily dependent upon the specific surface phosphorus and hydraulic loads. The closer the L_p

value is to the calculated L_c value, the more likely the assumed critical total phosphorus concentration will occur.

Table 22 shows the computed values for critical loading for 10 and 20 mgP m^{-3} total phosphorus concentrations and the specific surface loading for each year of study, June through November (183 days). The mean daily total phosphorus load for 1979 was higher than what was observed in 1978 but substantially lower than pre-AWT values. The specific surface loading of 1979 (3.99 gP m^{-2}) was greater than the suggested critical loading level of 2.12 gP m^{-2}.

Zooplankton Standing Crop

A total of 52 species in 37 genera were identified during the study (Table 23). Twelve species were Cladocera, four Eucopepoda and 36 Rotifera.

The major zooplankton were ranked by percent occurrence (Table 24) and abundance (Table 25). Bosmina longirostris and cyclopoid copepidites occurred in 100 percent of the samples but only ranked fifteenth and eleventh, respectively, according to abundance. Keratella cochlearis and nauplius larvae were also present in every sample and were the two most abundant organisms in the 1979 samples ranking first and second, respectively. Other frequently encountered species included Synchaeta pectinata and Polyarthra vulgaris which ranked fourth and fifth, respectively, according to abundance. Cyclops bicuspidatus thomasi, while present in every sample, ranked only seventeenth in abundance and Pompholyx sulcata, which ranked fifth in occurrence, was a major contributor in abundance, ranking third. Collectively, the Rotifera comprised 60.2 percent of total numerical standing crop followed by

Table 22. Mean daily total phosphorus load (metric tons), hydraulic load (q_s), specific surface loading (L_p) and critical loading (L_c; using 10 and 20 mg m^{-3} total phosphorus as threshold values) to Long Lake, WA. for all study years during the period of June through November (183 days).

Year	Mean daily total P load to Long Lake (metric tons)	q_s (m)	L_p (g P m^{-2})	$L_{c(10)}$ (g P m^{-2})	$L_{c(20)}$ (g P m^{-2})
1972	1.00	128.3	8.78	1.72	3.43
1973	0.91	56.8	7.99	0.86	1.71
1974	0.88	165.8	7.73	2.14	4.30
1975	1.06	138.3	9.31	1.83	3.66
1977	0.68	54.5	5.97	0.83	1.65
$\overline{x}$	0.91	108.7	7.96	1.48	2.95
1978	0.20	80.7	1.76	1.15	2.30
1979	0.45	73.4	3.99	1.06	2.12
$\overline{x}$	0.33	77.1	2.88	1.11	2.21

Table 23. Zooplankton species observed during the study, Long Lake, WA. (1979).

- PHYLUM: ARTHROPODA: Subphylum: Mandibulata
  - Class: Crustacea
    - Subclass: Branchiopoda
      - Order: Cladocera
        - *Alona guttata* Sars
        - *Bosmina longirostris* (O. F. Müller)
        - *Ceriodaphnia lacustris* Birge
        - *Ceriodaphnia reticulata* (Jurine)
        - *Chydorus sphaericus* (O. F. Müller)
        - *Diaphaphanosoma leuchtenbergianum* Fischer
        - *Daphnia galeata* Sars *mendotae* Birge
        - *Daphnia pulex* Leydig
        - *Daphnia retrocurva* Forbes
        - *Daphnia schodleri* Sars
        - *Leptodora kindtii* (Focke)
        - *Leydigia quadrangularis* (Leydig)
    - Subclass: Copepoda
      - Order: Eucopepoda; Suborder: Calanoida
        - *Diaptomus siciloides* Lilljeborg
      - Order: Eucopepoda; Suborder: Cyclopoida
        - *Cyclops bicuspidatus thomasi* Forbes
        - *Cyclops vernalis* Fischer
        - *Mesocyclops edax* (Forbes)
- PHYLUM: ROTIFERA
  - Class: Monogononta

Table 23.--(*Continued*)

Order: Ploima

Anuraeopsis cristata Berzins

Ascomorpha sp. Perty

Asplanchna brightwelli Gosse

Brachionus angularis Gosse

Brachionus calyciflorus Pallas

Collotheca balatonica Varga

Collurella obtusa Hauer

Euchlanis dilatata Ehrenberg

Gastropus minor (Rousselet)

Kellicottia longispina (Kellicott)

Keratella cochlearis (Gosse)

Keratella quadrata (O. F. Müller)

Lecane luna (O. F. Müller)

Leptadella ovalis (O. F. Müller)

Monostyla lunaris Ehrenberg

Monostyla quadridentata

Notholca labis Gosse

Notholca squamula Müller

Pleosoma hudsoni Imhof

Pleosoma truncatum (Levander)

Polyarthra vulgaris Carlin

Synchaeta pectinata (Ehrenberg)

Trichocerca capucina (Wierzejski)

Trichocerca cylindrica Imhof

Table 23.--(*Continued*)

- *Trichocerca longiseta* (Shrank)
- *Trichocerca multicrinis*
- *Trichocerca pusilla* Jennings
- *Trichotria tetractis*

Order: Flosculariacea
- *Conochilus unicornis* Hlava
- *Filinia longiseta* Ehrenberg
- *Hexarthra mira* Hudson
- *Pompholyx sulcata* Hudson

Order: Collothecacea
- *Collotheca balatonica* Varga

Class: Diogononta

Order: Bdelloida
- *Philodina* sp. Ehrenberg
- *Rotatoria citrinus*
- *Rotatoria neptunica*

Table 24. Rank of the major zooplankton species of Long Lake, WA. according to occurrence (%) based upon collections from all stations (1979).

Rank	Taxon	Occurrence
1	*Bosmina longirostris*	100
	Cyclopoid copepidites	100
	Cyclops bicuspidatus thomasi	100
	Keratella cochlearis	100
	Nauplii	100
	Polyarthra vulgaris	100
	Synchaeta pectinata	100
2	*Daphnia retrocurva*	92
3	*Diaptomus siciloides*	83
4	*Ascomorpha* sp.	75
	Collotheca balatonica	75
	Diaphanosoma leuchtenbergianum	75
	Mesocyclops edax	75
5	*Kellicottia longispina*	67
	Leptodora kindtii	67
	Monostyla lunaris	67
	Pompholyx sulcata	67
6	*Brachionus angularis*	58
	Conochilus unicornis	58
7	*Daphnia galeata mendotae*	50
	Filinia longiseta	50
	Trichocerca capucina	50

Table 24.--(Continued)

Rank	Taxon	Occurrence
8	*Asplanchna brightwelli*	42
	Euchlanis dilatata	42
	Keratella quadrata	42
	Trichotria tetractis	42

Table 25. Rank of the major zooplankton species of Long Lake, WA. according to abundance (%) based upon collections from all stations (1979).

Rank	Taxon	Abundance
1	*Keratella cochlearis*	23.28
2	Nauplii	18.68
3	*Pompholyx sulcata*	14.07
4	*Synchaeta pectinata*	7.78
5	*Polyarthra vulgaris*	7.48
6	*Daphnia retrocurva*	6.87
7	*Diaptomus siciloides*	5.08
8	*Ascomorpha* sp.	2.47
9	*Conochilus unicornis*	2.27
10	*Diaphanosoma leuchtenbergianum*	2.15
11	Cyclopoid copepidites	2.04
12	*Daphnia galeata mendotae*	1.72
13	*Mesocyclops edax*	1.51
14	*Collotheca balatonica*	1.22
15	*Bosmina longirostris*	1.05
16	*Trichocerca capucina*	0.45
17	*Cyclops bicuspidatus thomasi*	0.27
18	*Asplanchna brightwelli*	0.23
19	*Brachionus angularis*	0.21
20	*Cyclops vernalis*	0.17

the Eucopepoda (27.8 percent). The Cladocera were the least significant contributor comprising only 12.0 percent of the total zooplankton standing crop.

Rotifers were most abundant at stations 0 and 1 and least numerous at station 2 (Fig. 14). Cladocerans were also more prominant at station 0 and were equally abundant between the remaining four stations. The copepods were most prevalent at station 0 with the lowest density occurring at station 4.

Figure 15 shows the seasonal fluctuations of the zooplankton on a numerical basis. The seasonal data also reveals the numerical dominance of the rotifers throughout the year beginning with a major peak in early June, primarily of K. cochlearis. Keratella was also responsible for the peak in late July. Other major rotifer peaks were observed in August and September as a result of increased numbers of Pompholyx sulcata and Synchaeta pectinata, respectively.

An early peak of copepods was observed in July due to increased numbers of Diaptomus siciloides (Fig. 15). Combined numbers of D. siciloides and Mesocyclops edax allowed the Eucopepoda to reach a seasonal high in early August. Cladoceran maximum density occurred in early July and reflected a Daphnia retrocurva maximum during that period.

Successional patterns for the major rotifers are depicted in Figure 16. Synchaeta pectinata appeared in early April along with a few individuals of Polyarthra vulgaris. May and June was marked by increased numbers of S. pectinata and P. vulgaris as well as pulses of Keratella cochlearis and Conochilus unicornis. All major rotifer species declined in mid-July, with the exception of K. cochlearis which experienced a major pulse. Early August was characterized by increased numbers of

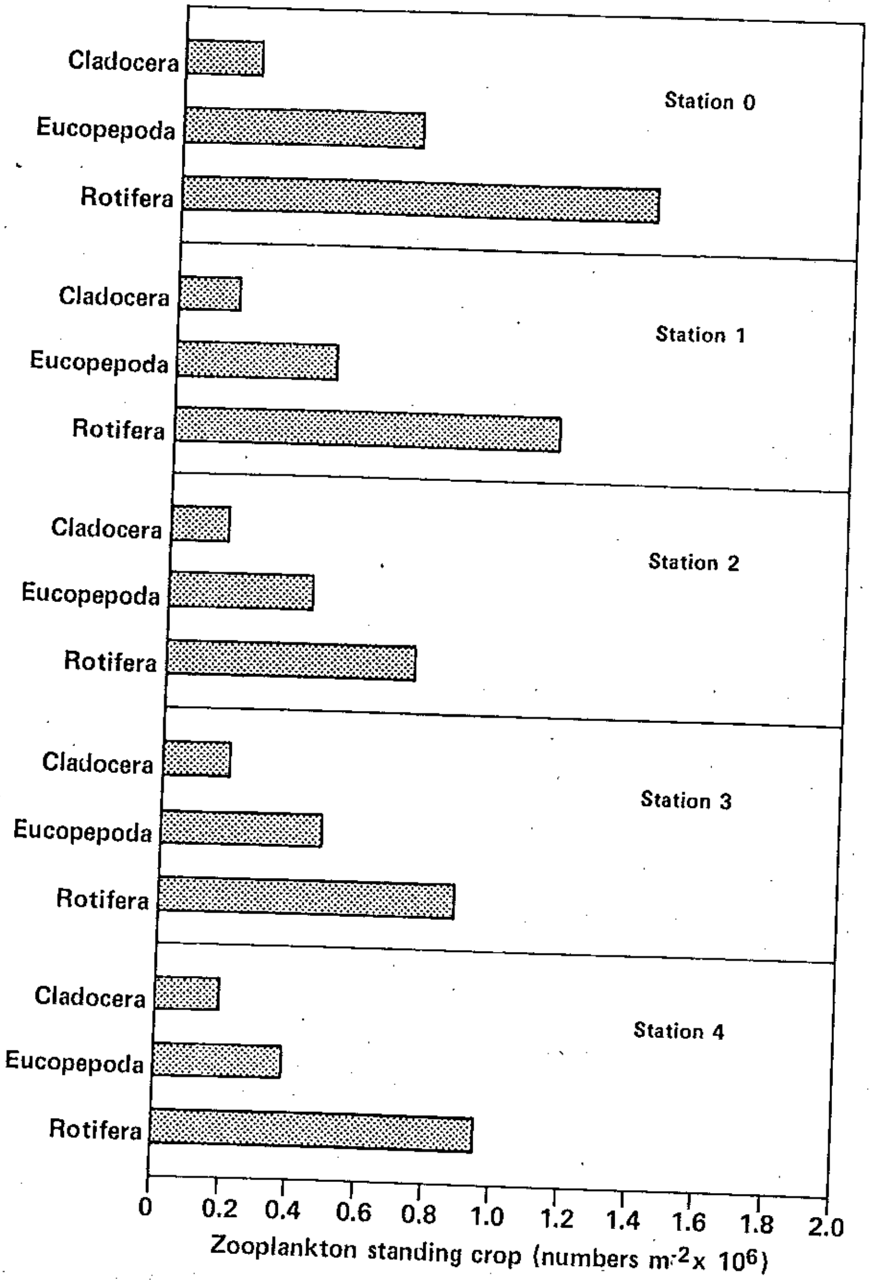

Figure 14. Mean zooplankton standing crop (# $m^{-2} \times 10^6$) by order and station over all dates, Long Lake, WA. (1979)

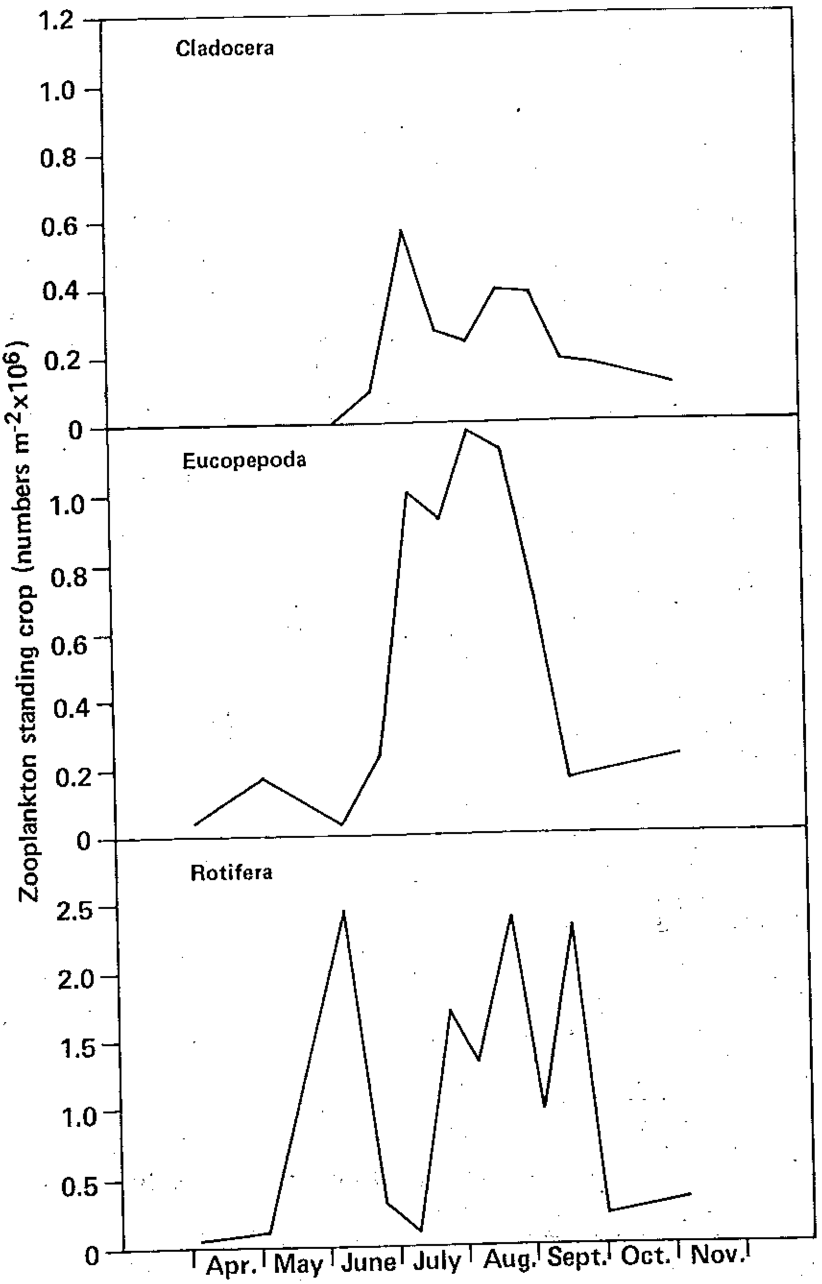

Figure 15. Mean zooplankton standing crop (# $m^{-2} \times 10^6$) by order and date over all stations, Long Lake, WA. (1979)

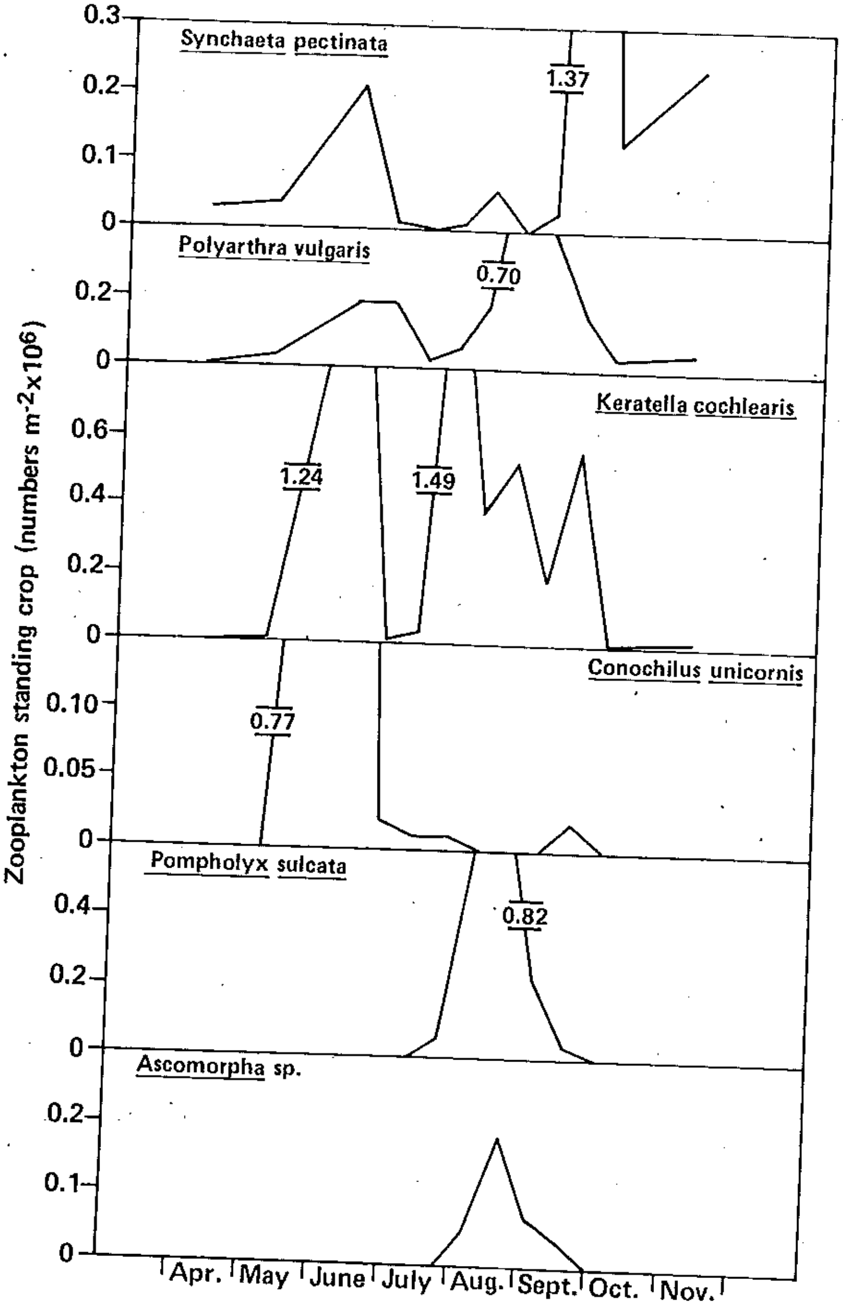

Figure 16. Successional trends of the predominant rotifers by date over all stations, Long Lake, WA. (1979)

S. pectinata, *P. vulgaris*, *P. sulcata* and *Ascomorpha* sp. *Synchaeta pectinata* and *K. cochlearis* standing crops increased again in September with only *S. pectinata* finishing the sampling season in appreciable numbers in October and November.

Cyclops bicuspidatus thomasi established early dominance among the Crustacea in April and remained the predominant species until late June (Fig. 17). Early July saw increased numbers of the cladocerans, *Bosmina longirostris*, *Daphnia retrocurva* and *Diaphanosoma leuchtenbergianum*. The copepods *D. siciloides* and *M. edax* were also present in July with a predominance of these two species occurring in August along with *D. leuchtenbergianum*. Crustacean standing crops declined in early September with only *C. bicuspidatus thomasi* and *B. longirostris* showing increased numbers in October and November.

Zooplankton-Phytoplankton Relationships

The zooplankton-phytoplankton relationships for 1979 are illustrated in Figure 18. While the greatest phytoplankton biovolume was evident at station 3, the largest mean zooplankton standing crop was observed at station 0, while the lowest mean zooplankton density occurred at station 2.

The early phytoplankton pulse in April and May, primarily of diatoms, was followed by a zooplankton pulse of rotifers in late May and June. Diatoms were again prevalent in the phytoplankton pulse of July which stimulated a considerable rotifer response in August, preceded by a smaller peak of cladocerans and copepods in July. An early September bloom of *Anabaena* was followed by a decline in zooplankton standing crop, possibly due to the inhibitory effects of extracellular toxins

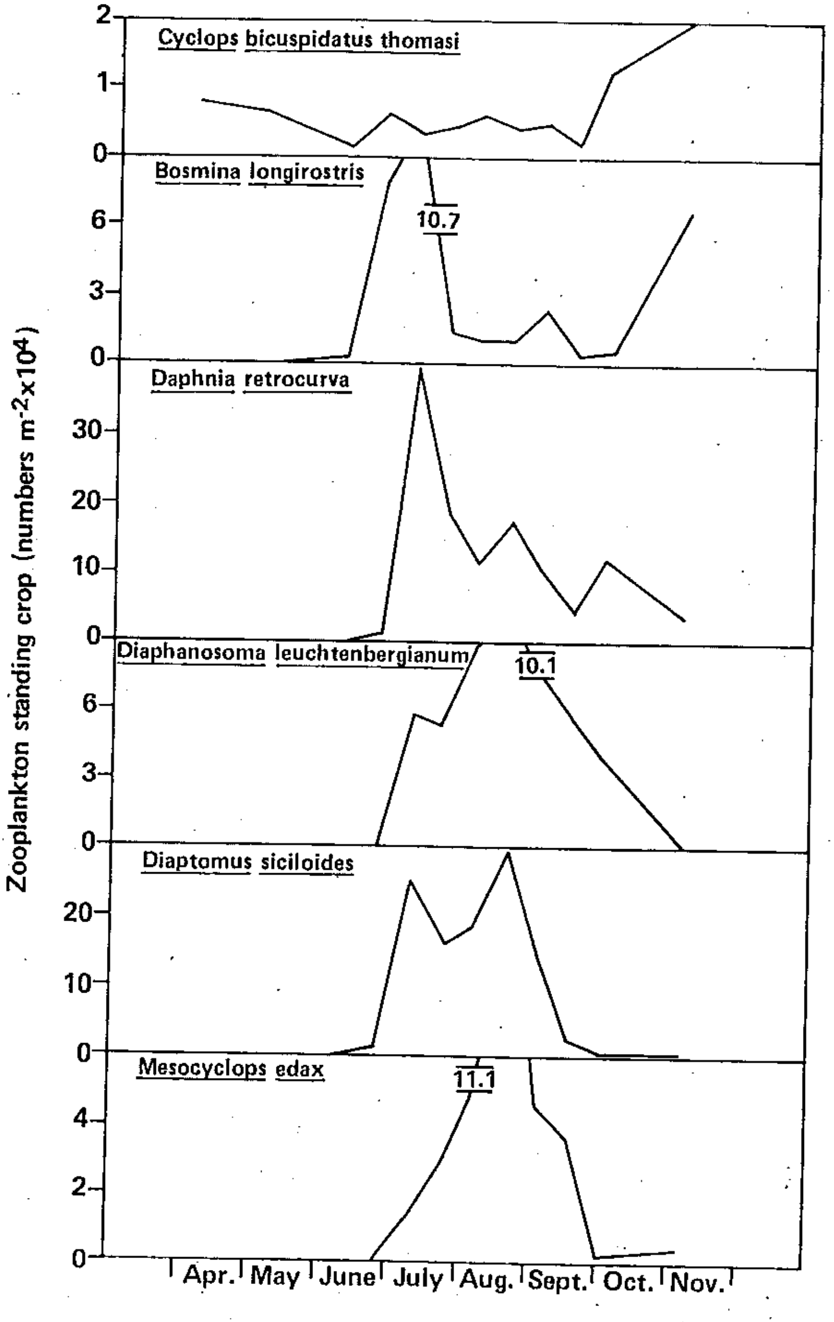

Figure 17. Successional trends of the predominant crustaceans by date over all stations, Long Lake, WA. (1979)

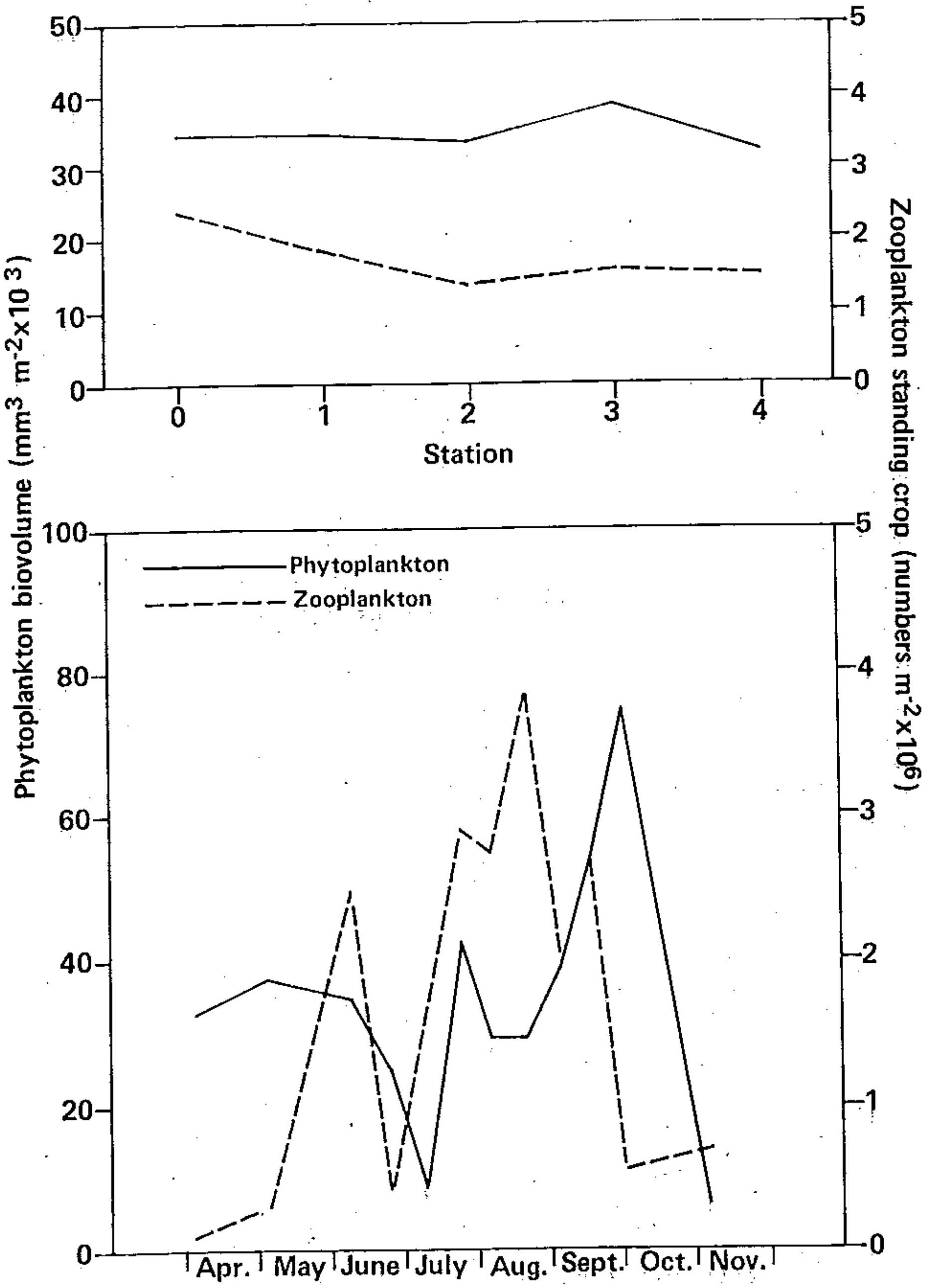

Figure 18. Mean zooplankton density (# $m^{-2} \times 10^6$) and phytoplankton biovolume (mm^3 $m^{-2} \times 10^3$) by station over all dates and by date over all stations, Long Lake, WA. (1979)

often associated with many blue-green algae (Arnold, 1971). A small increase in zooplankton numbers was noted in October following the recurrence of the diatoms in late September.

1979 Zooplankton Findings versus Previous Study

Comparison of 1979 zooplankton data with the 1978 zooplankton study (Soltero _et al_., 1979) showed similar results for the two years. The zooplankton were dominated numerically by the rotifers followed by the Eucopepoda and then by the Cladocera. The six most abundant species in 1978 were also the most abundant in 1979 (Table 25). The successional patterns for both years were similar to those observed prior to AWT (Soltero _et al_., 1973; 1974; 1976; 1978).

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