MOSES LAKE QUALITY: RESULTS OF DILUTION, SEWAGE DIVERSION AND BMPs - 1977 THROUGH 1988

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by

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EXECUTIVE SUMMARY

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The restoration of 2,700 hectares of Moses Lake essentially began in 1962 with a study to determine the cause(s) for the lake's poor quality by Sylvester and Ogelsby (1964). With continued persistence by the late Clinton Connelly, Director of the Moses Lake Irrigation and Rehabilitation District, research continued on the lake to evaluate the feasibility and efficacy of adding low-nutrient Columbia River water from the Bureau of Reclamation's nearby East Low Canal in order to lower lake nutrient content and thereby reduce the abundance of nuisance algae, as recommended by Sylvester and Ogelsby. Fifteen years later, with funding from EPA's Clean Lakes Program and an important clarification that the lake would more accurately be "diluted" rather than "flushed," regular dilution water additions began in March, 1977. Dilution water has been added regularly during the spring since 1977 (except 1984), usually continuing through much of the critical summer period, dilution water was distributed to previously undiluted Pelican Horn in 1982, treated sewage effluent was diverted from Pelican Horn in 1984, and best management practices (BMPs) in the watershed to further reduce nutrient inputs began in 1987. As a result, water quality in 1988 was the highest ever observed.

The lake was sampled at 8 stations along transects at a depth of 0.5 m during March through September for total and soluble reactive phosphorus (TP and SRP), nitrate and total nitrogen (TN), chlorophyll _a_ (and algal biovolume) and water transparency. Samples were collected at other depths to determine whole lake concentrations, but water quality improvements were judged with data from transects, which were also sampled in 1969-1970 before treatment. Samples were also regularly collected from five inflows to the lake.

Dilution water was added to Parker horn during a total of 971 days over the 12-year period. The average annual input for the actual days of addition was 169.4 x 10^6 m^3/yr, (24.2 m^3/s, 847 cfs), which represented an average flushing rate for the whole Parker Horn of 17%/day. For the April to September period (183 days) average dilution inflow was 130 x 10^6 m^3/yr (8.2 m^3/s, 287 cfs) for an average spring-summer flushing rate in Parker Horn of 5.8%/day. Including normal flows from Crab Creek, the average flushing rate was 7.8 %/day in Parker Horn. Flushing for the whole lake averaged 0.27%/day and 0.73%/day for dilution water only and including normal flows, respectively.

Lake quality improved markedly in response to dilution with TP and chl _a_ decreasing by at least 50% and transparency increasing by 100% since dilution

began in 1977. Additional improvements, primarily in South Lake, occurred following sewage diversion so that average values for Parker Horn and South Lake combined during 1986-1988 were 45 $\mu g/L$, 17 $\mu g/L$ and 1.6 m for TP, chl a and transparency, respectively. The original goals for TP and chl a were 50 and 20 $\mu g/L$, respectively. Part of the cause for this improvement was due to a marked decrease in Crab Creek TP and SRP content apart from dilution or sewage diversion. Although the lake's trophic state is still eutrophic, (TP >25 $\mu g/L$, chl a > 10 $\mu g/L$ and transparency <2 m), it is dramatically improved from its pre-treatment hypereutrophic state in 1969-1970 (TP = 154 $\mu g/L$, chl a = 57 $\mu g/L$, transparency = 0.8 m).

Pelican Horn quality in general changed little following pumping. Although algal biomass was reduced by a high flushing rate (19%/day) in upper Pelican Horn, algal growth was actually benefited in lower Pelican by added nitrogen, which was severely limiting. Diversion of sewage, however, reduced TP by about 90%, because the effluent contained ten times higher TP content than the Horn's groundwater input. As a result, chl a declined to levels near that in other lake sections, but transparency has improved little (0.4 to 0.6 m) due to the shallowness of Pelican Horn and its active carp population.

BMPs that began in 1987 probably did not contribute to the added improvement in lake quality observed in 1986-1988, because only about one-fifth of the total irrigated area (1612/8380 hectares) actually had nutrient and water controls instituted. Thus, only 5% reduction in watershed nutrient loss probably occurred, instead of the anticipated 25% reduction, amounting to an expected decrease in lake content of only 1.3%. The minor effect so far was confirmed by the lack of change observed in Crab Creek nitrogen content.

Blue-greens have and still do dominate the plankton algae in Moses Lake. Although the blue-green fraction in Parker Horn and South Lake had declined following dilution, they subsequently returned to their nearly total dominance during summer (75-90% of biovolume) with Aphanizonenon and Microcystis being the principal bloom formers. Small green algae still dominate the biomass in Pelican Horn as before sewage diversion. Although blue-greens became important (50% of biovolume) immediately following diversion, they have since declined to only 5%. Experimental results indicate that the continued dominance by blue greens in Parker Horn and South Lake and greens in Pelican Horn may be due to, respectively, lower and higher free CO_2 concentrations.

Phosphorus usually limits algal growth now, whereas nitrogen was limiting before (1969-1970) and during the earlier years after dilution began (1977-1979). This shift to P limitation was caused by a gradual decrease in P in the Crab Creek inflow, diversion of sewage effluent and the Mt. St. Helens' ashfall, which reduced internal P loading from lake sediments. The switch to P limitation has resulted in dilution water, which effectively reduces stream and lake N content more than P, being less effective at controlling algal growth in recent years. For example, P and chl a in Parker Horn did not decline in 1984, the year with no dilution, although transparency was less due to no reduction in non-algal particulate matter.

Internal loading of P from lake sediments was determined as the residual after surface outflow P and change in lake P were subtracted from inflow P. Internal loading has become relatively more important since sewage diversion and the decline in Crab Creek P content. The quantity of internal loading varied greatly from year to year--almost five times as much variation as external loading--and increased with wind mixing (as indicated by decreased relative thermal resistance, RTR, in the water column) and to a lesser extent with flushing rate.

Predictive models for TP were developed for Lower Parker Horn, Rocky Ford Arm and Lower Pelican Horn and predicted TP was usually within the expected variation due to sampling error. The effect of further decreases in TP loading to Crab Creek and dilution water addition on TP and chl a in Parker Horn can be predicted using the range in RTR observed over the past nine years as a measure of prediction uncertainty. An enhancement of internal loading from increased flushing (increased dilution water addition) in Parker Horn may be due to a resulting increase in the concentration gradient between sediment and overlying water.

Nutrient concentration data from inflow streams and the lake do not reveal any significant changes that could be due to the recently instituted BMPs and the detention basin on lower Rocky Ford Creek. This is probably because only 20% of the anticipated land area had thus far been included for irrigation water and fertilizer control by the end of this project (1988) and rooted macrophytes had not developed in the detention basin until 1989. However, further reduction in lake nutrient content and improved lake quality may yet occur if full compliance is attained for BMPs and if the detention basin retains nutrients as expected.

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Acknowledgments

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Introduction

The restoration of Moses Lake has taken about 25 years. The effort essentially began in 1962 with a study for the MLIRD (Moses Lake Irrigation and Rehabilitation District) by Sylvester and Oglesby (1964). Their conclusion was that the lake's hypereutrophic state was caused by the excessive load of phosphorus and nitrogen from the inflow streams and treated sewage effluent. The large population of carp was also considered to be related to the lake's highly turbid appearance. There was no accounting in their nutrient budget for internal loading of phosphorus via exchange processes between bottom sediment and overlying water or by excretion from carp. Internal loading is now known to contribute on the average about one third of the total load of phosphorus to the lake.

The late Clinton Connelly, Director of the MLIRD, commissioned that initial study with the hope of improving the quality of Moses Lake. Through Mr. Connelly's persistence that goal has finally been achieved. The success that Sylvester and Ogelsby (Ogelsby, 1969) had with the dilution of Green Lake in Seattle, beginning in 1965, encouraged an attempt to apply the same treatment to Moses Lake in the late 1960s. Moses Lake could be diluted rather easily by diverting low nutrient Columbia River water from the Eastlow Canal into Parker Horn via an existing route - Rocky Coulee Wasteway and Crab Creek (Figure 1). Unfortunately, a failure to communicate an understanding of the treatment process to other irrigation districts that use the outflow from Moses Lake precluded the start of dilution in 1968. At that time the proposed treatment was referred to as "flushing", a term that conveyed the wrong meaning. In fact, "dilution" is a more accurate description of the process, because with the addition of low nutrient East Low Canal water, the nutrient content in the outflow of Moses Lake would be lowered.

Research nevertheless continued on the lake to evaluate the potential of dilution as a lake quality control. Experiments in plastic bags held in upper Parker Horn during the summer of 1970 showed that algal growth (maximum biomass) could be reduced by adding Eastlow Canal water and the reduction was proportional to the fraction of dilution water added (Welch et al., 1972). At the same time, a two year data base on the lake and its inflows was developed with support from the Environmental Protection Agency (EPA) and the Washington Water Research Center. That data base would serve to represent pretreatment conditions. Through Mr. Connelly's persistence, another nitrogen and phosphorus budget was constructed based on the 1969-1970 data base, along with supplemental inflow data, to determine the relative importance of the specific sources (e.g. irrigation return flow, fish hatcheries, etc.). The high N and P concentrations in Crab Creek were considered to be coming primarily from irrigation returns, via surface and/or subsurface routes (Welch, et al., 1973). Dilution was an obvious procedure to reduce those concentrations, especially in Crab Creek. But it was not until "Clean Lake" funds became available in 1976 through EPA that large volumes of East Low Canal water could be added to Moses Lake for purposes of eutrophication control. Brown and Caldwell Engineers, Inc. initially managed the dilution project for MLIRD starting in 1977 and except for 1984, dilution water has entered the lake for varying periods between April and September of each successive year.

The next goal was to distribute dilution water to other parts of the lake where dilution water did not reach, e.g. the Rocky Ford (Main) Arm and Pelican

Horn. Dilution water was actually found to be driven from Lower Parker Horn by the wind half way through the Rocky Ford Arm reaching Airman's Beach in adequate quantities (Welch et al., 1983). But movement into Pelican Horn was ineffective. Therefore, upper Parker Horn (diluted) water has been pumped at the rate of 1.4 m^3/s (50 cfs) to upper Pelican Horn via an underground pipe beginning in 1982 (Figure 1). In 1984 sewage effluent was diverted from Pelican Horn and in 1987 a system of BMPs (Best Management Practices) was implemented in the Crab Creek watershed to limit the agricultural use of water and fertilizer. Of the 8,380 ha (20,954 a) of irrigated land in the Crab Creek drainage and the 104 potential participants, 36 farms representing 2,140 ha (5,346 a) have actually been involved in controls to date (Bain, 1988). In addition, a dam was constructed on Rocky Ford Creek in 1985 to prevent carp from migrating into the creek and at the same time provide a trap for incoming phosphorus in the backwater pond. Also, a septic tank policy was developed for near-lake residences.

This report summarizes the large amount of water quality data collected from Moses Lake and its inflows since restoration began in 1977. The purpose is to evaluate the overall response to dilution-water addition (1977-1983, 1985-1988), diversion of treated sewage effluent in 1984 and BMPs in 1987-1988. Ashfall from the eruption of Mount St. Helens in May, 1980 was a natural event that produced significant effects in the lake. However, those effects appeared to be relatively short-lived. Evidence from cores showed that the 3-4 cm (1.2-1.6 inch) ashlayer sunk well below the sediment surface within two years after the ashfall (Welch et al., 1985).

The other goal of the project was to develop a modelling approach to predict the lake response to changes in nutrient loading. A dynamic wind-mixing and algal-growth model dependent on nitrogen was developed in order to predict the maximum algal biomass during the April to September period in lower Parker Horn and South Lake. Part of the spring-summer growth cycle was verified for 1979 data when nitrogen was limiting growth, although the fall decline was predictable at one station, but not the other (Marquis, 1985). That approach was not pursued further, because; 1) phosphorus (P) became limiting after the ashfall, and 2) a more empirical approach aimed at average concentrations may be more reliable. Therefore, a steady-state P model, which includes internal loading, was developed for four segments of the lake. Average chlorophyll a and Secchi transparency could then be predicted from P concentrations, which is a conventional approach for management purposes.

Description of Moses Lake

Moses Lake was formed naturally by blowing sand dunes that dammed Crab Creek. The level is stabilized at about 319 m (1046 ft) in summer (318 m in winter) by dams constructed in 1929 by the City of Moses Lake and in 1963 by the U.S. Bureau of Reclamation. The lake is large (2790 ha) and relatively shallow with an average depth of 5.6 m (Table 1). Only the South Lake segment, with a maximum depth of 11.5 m, permanently stratifies thermally during the summer. Therefore, 80% of the lake can be considered polymictic. That is, stratification can develop during calm periods, but then will completely mix when windy conditions return, with the degree of mixing dependent on wind speed and depth.

Table 1. Basin characteristics of Moses Lake.[1] (from Patmont, 1980)

Basin	Depth (m)	Area (ha)	Volume (x 10^6 m^3)	% Lake Volume	Representative Station Sample Location
1. Upper Parker Horn	Z_m=4.0; $\bar{Z}$=1.8	79.3	1.44	0.9	5-T
2. Lewis Horn	Z_m=5.5; $\bar{Z}$=3.5	90.0	3.19	2.0	6-T[2]
3. Lower Parker Horn	$\bar{Z}$=4.9	221.3	10.88	7.0	7
a. strata 1	0-1.0		2.39	1.5	7-T, 7-S
b. strata 2	1.0-3.5		4.46	2.9	7-2
c. strata 3	3.5-9.5		4.02	2.6	7-B
4. Cascade	Z_m=11.5; $\bar{Z}$=8.4	385.8	32.32	20.8	8-C, 8-T
5. South Lake	$\bar{Z}$=7.2	441.4	31.85	20.5	9
a. strata 1	0-1.0		4.75	3.1	9-T, 9-S
b. strata 2	1.0-3.5		10.54	6.8	9-2
c. strata 3	3.5-8.5		13.16	8.5	9-6
d. strata 4	8.5-11.5		3.40	2.1	9-B
6. Lower Pelican Horn	$\bar{Z}$=3.5 m	236.7	8.37	5.4	10
a. strata 1	0-1.0		2.76	1.8	10-T, 10-S
b. strata 2	1.0-3.5		4.86	3.1	10-2
c. strata 3	3.5-7.0		0.75	0.5	10-B
7. Middle Pelican Horn	Z_m=2.2; $\bar{Z}$=1.3	81.6	1.07	0.7	11-T
8. Upper Pelican Horn	Z_m=2.2; $\bar{Z}$=1.4	49.2	0.69	0.4	19
9. Rocky Ford Arm	Z_m=9.5; $\bar{Z}$=5.5	1204.6	65.85	42.3	12-C, 12-T
Moses Lake - Total	$\bar{Z}$ = 5.6	2789.9	155.66	100.0	

[1] Based on lake level of 319 m MSL.

[2] Not sampled during this study.

The two principal inflow streams are Crab Creek and Rocky Ford Creek. The latter is a spring-fed stream and enters the Rock Ford Arm, while Crab Creek drains nearly 80% of the 6000 km^2 watershed. However, their May - September mean flows are more similar; Crab Creek = 3 m^3/s and Rocky Ford Creek = 2 m^3/s. The flushing rate for the whole lake at those flow rates is about 1/year. Most of the summer flow in Crab Creek is return flow from irrigated lands in the lower drainage; prior to the Columbia Basin Irrigation Project, Crab Creek did not flow in summer. However, most of the drainage is devoted to dry land farming and high runoff can originate from that large area in the spring if there is a substantial snow pack. Thus, in some years the flushing rate of the whole lake, based on annual flows, can be 2/year.

The city of Moses Lake lies largely in the area between Parker and Pelican Horns (Figure 1). The population of the city and urban fringe is about 20,000. The lake is used extensively for recreation; boating, swimming and fishing. The lake is well known for its crappie and bass fishing. Commercial fishing for carp has occurred on the lake, with an annual harvest of 0.9 million kg (2 million lb).

Sampling Procedures and Analysis

Sampling Methods

Sampling design followed that of previous years (Welch et al. 1969; Carlson and Welch, 1983). Horizontal transect samples were collected at a depth of 0.5 m to correct for patchiness of algae and chemical constituents, while vertical profiles were taken at transect midpoints to determine the extent and effects of stratification (Figure 1).

The lake was sampled at eight stations for all constituents approximately twice monthly from March through September and monthly for the remainder of the year (Figure 1). During December through February only lake inflows and outflows were sampled for nutrient content due to ice cover. Stations 8 and 12 were discontinued after June, 1987. Post-treatment sampling was not begun in Upper Pelican Horn (19) until 1982, and was inconsistent after 1985 due to boat access difficulties.

Transect samples were taken using a pitot-type tube attached to the underside of the boat, while traveling at a constant velocity along the mid-line (transect) of each basin area. Discrete samples for vertical profiles were taken using a 4 L VanDorn-type sampler at the midpoint of transects 7, 9, and 10. Sample depths were at the surface, 2 m, 6 m (at Station 9 only), and within 1 m of the lake bottom. Composite vertical samples, representing a water column average, were taken at Stations 8 and 12 using a flexible PVC tube.

A qualitative survey of macrophyte abundance was conducted in September, 1983. Macrophytes were sampled by dragging a garden rake along the lake bottom along both sides of Upper Parker Horn from Lewis Horn to the Alder Street bridge (see transect for Station 5, Figure 1).

Water Analysis

Most analytical methods remained the same during the pre-treatment years of 1969-1970 and throughout the post-treatment years, 1977-1988. There were some exceptions, however, mostly as instruments changed.

Alkalinity was determined within 24 hours of sampling by potentiometric titration to endpoint pH 4.8 (Corning model 5 pH meter) with 0.2 N H_2SO_4 (APHA, 1958). Determination of pH was made in the field in samples from discrete depths from each vertical profile with a VWR Scientific mini-pH meter and with a Cole-Parmer model 5985-80 Digi-Sense pH meter after 1985. Vertical temperature profiles were made at one meter intervals using a YSI model 54A temperature probe. Samples of bottom water at 0.5 m above the sediment surface were taken for the determination of dissolved oxygen and analyzed by the Winkler method (APHA, 1985). These data are stored in computer format and only raw data from 1986-1988 are presented in this report.

Cell volume of phytoplankton was determined in transect samples only. The samples were preserved with Lugol's acid-iodine solution and stored in the dark. Cell counting was performed using a Palmer-Maloney cell at 200 power. The basis of cell counts was either 100 Whipple grids or a total cell count of 300, whichever occurred first. Counts of filamentous algae were determined by the method of Olson (1972). Phytoplankton biovolume was estimated by multiplying the cell count for a particular genus by an appropriate mean cell volume for that genus. Mean cell volumes were determined using stage micrometer measurements and the procedures of Emery (1972) or Olson (1972).

Plant pigment chlorophyll a (chl a) was analyzed using the florometric method of Strickland and Parson (1972) through 1986. Corrections for phaeophytin were made by acidifying the extract. The fluorometric readings were calibrated against the spectrophotometric method according to the procedures of Lorenzen (1976). Except for September, chl a data for 1984 were found to be invalid due to a gradually declining photomultiplier tube in the fluorometer. Chl a values for 1984 were therefore estimated with regression equations of biovolume on chl a using past data (Weiher, 1986). After 1986, chl a was determined directly on a Perkin-Elmer Lambda 3 spectrophotometer (Srickland and Parsons, 1972). Both methods were employed on a portion of 1985 and 1986 samples to insure a consistency in results.

Samples for soluble nitrogen as nitrate plus nitrite ($NO_3 + NO_2$-N) were filtered three to six hours after collection through 0.45 μm glass fiber filters. The filtrate was frozen in acid-washed polyethylene bottles until analyzed. Filters were soaked in distilled water for at least 24 hours prior to use. Soluble nitrogen was determined using the colorimetric cadmium reduction method (EPA, 1979) and a Technicon II autoanalyzer. An Alpkem autoanalyzer was used for 1987 and 1988 samples. Total nitrogen (TN) samples were unfiltered and stored frozen. TN was analyzed with the ultraviolet light oxidation method of Strickland and Parsons (1972) with some major modifications through 1986. Persulfate oxidation was used for 1987 and 1988 samples (Solorzano and Sharp, 1980). Following oxidation, samples were analyzed in the same manner as $NO_3 + NO_2$-N.

Samples for soluble reactive phosphorus (SRP) were filtered and preserved by freezing. SRP was determined by the acid molybdate heteropoly blue method

(Strickland and Parsons, 1972). Total phosphorus (TP) samples were unfiltered and were preserved unfrozen with 1 N H_2SO_4 (four drops per 60 ml bottle). Samples were digested by the persulfate method and the digestate was analyzed for SRP (Strickland and Parsons, 1972). Standards and blanks were processed along with each sample set for all nutrient analyses.

Quality control and assurance were performed regularly for nutrient analyses. Standards and blanks were processed for each set of samples. Precision was $\pm$ 5% or less for all levels of SRP, TP and NO_3, except for the lowest level of SRP, with accuracy (% recovery) being within $\pm$ 8% of the true value, except for the low level of SRP (Table 2). Accuracy was similar for TN, but precision on the high levels was lower ($\pm$ 24 %).

Secchi disc depth was determined at each vertical profile station, at the north end of the Station 5 transect and usually at the midpoint of the Station 19 transect when it was possible to enter that section with a boat. Specific conductance was determined in the laboratory with a Barnstead model 5 conductivity bridge (APHA, 1985).

Constituent Variability

Two series of replicate sample collections were conducted in 1983 to determine the variability, or error (precision), in estimating the mean concentrations of NO_3, TP and chl a in the lake (Marquis, 1985). Samples from three, normal transects were collected on the same day from stations 7 and 9 as indicated by the longitudinal lines in Figure 1. These transects were repeated in identical fashion. Three additional transects at each station were traversed in an irregular fashion so as to sample diagonally across the lake and obtain the influence of more nearshore concentrations. A third type of collection was made by collecting a series (along the transect) of water column composites by lowering and filling a flexible plastic tube from the surface to bottom.

Table 3 shows the results of this exercise. The sampling (plus analytical) precision error was similar for TP and chl a, being on the order of $\pm$ 20% or less (means 10% and 13%) regardless of station or sample procedure. NO_3, on the other hand, showed much higher sampling errors, especially when concentrations were low. This would be expected because NO_3 is soluble and available for algal uptake, and was therefore subject to rapid, uneven depletion and, hence, a patchy distribtion. Small absolute variations would produce large relative errors when concentrations are low. However, the sampling (and analytical) errors for TP and chl a are comfortably low and indicate that the sampling procedures were effective in estimating surface concentration throughout the lake segments. Also, transect means may be a good estimator of water column mean concentrations. There would be times when the latter is not true, however.

Water Budgets

Water budgets were computed from gauged inputs and outputs, while subsurface flow (groundwater) was determined by the difference between inflow and outflow, considering lake volume according to:

Table 2. Univ. of Washington, Environmental Engineering and Science 1987 Laboratory Quality Data (EPA Composite Samples: WP284), n=number of replicates.

	GIVEN CONC. (μg/l)	MEASURED MEAN(μg/l)	SD(μg/l)	CV(%)	RECOVERY(%)
SRP	6.25	5.42	0.62	11.5	87
(n=6)	12.5	11.85	0.64	5.4	95
	50.0	49.51	2.28	4.6	99
TP	10	9.30	0.49	1.95	93
(n=9)	25	23.00	1.26	5.48	92
	100	93.05	1.81	5.27	93
NO_3-N	17.5	16.9	0.8	4.73	97
(n=4)	35.0	36.2	1.8	4.97	103
	140	142.4	2.4	1.69	102
TN	80	72	8.6	8	90
(n=4)	320	296	21.8	24	93

Table 3: Sample variability data for Parker Horn and the lower lake, obtained during field sampling in the summer of 1983.

DATA TYPE	SAMPLE VARIABILITY					
	NO3		TOTAL P		CHL A	
DATE	X±SD (µg/l)	CV(%)	X±SD (µg/l)	CV(%)	X±SD (µg/l)	CV(%)
PARKER HORN						
IDENTICAL PATH TRANSECTS (n=3)						
7/4/83	14±1	4	63±4	6	18.7±0.6	3
10/25/83	586±60	10	88±19	22	7.6±1.8	24
IDENTICAL AND IRREGULAR PATH TRANSECTS (n=6)						
7/4/83	13±3	23	65±6	9	17.3±3.0	18
10/25/83	551±90	16	93±13	14	7.6±1.2	16
IDENTICAL AND IRREGULAR TRANSECTS, AND COMPOSITE (n=15)						
7/4/83	16±7	47	70±16	22	16.2±3.0	18
10/25/83	559±82	15	99±21	21	7.4±1.2	16
LOWER LAKE						
IDENTICAL PATH TRANSECTS (n=3)						
7/7/83	25±19	75	115±5	4	13.3±1.1	8
10/25/83	132±11	8	139±10	7	14.3±1.3	9
IDENTICAL AND IRREGULAR PATH TRANSECTS (n=6)						
7/7/83	15±16	104	115±9	8	12.9±1.0	7
10/25/83	193±122	63	136±8	6	15.2±3.2	21

$$GW = \text{Outflow} - \text{Inflow} + \Delta \text{ Lake Storage}$$

The lake's outflow is controlled by both the Moses Lake Irrigation and Rehabilitation (MLIRD) outlet and the U.S. Bureau of Reclamation (USBR) outlet. The USBR outlet consisted of four stoplog-controlled openings and one center radial gate until the fall of 1982, when five radial gates were installed. Equations for flow calculations for the outlet prior to this change were given by Patmont (1980). The corrected equations that apply since then are follows:

$$Q = (B)(5.60\ h_1^{3/2})$$

B = Backwash correction due to baffle blocks

h_1 = Moses Lake elev. - datum (meters)

1) The radial gate's datum was 317.5 m (1041.3 ft) and was independent of the stoplogs' elevations (datums).

2) When $h_1 > h_2 > 0$, the backwash correction, B, is

   $B = [1-(h_2/h_1)^{3/2}]^{.385}$

   where h_2 = Potholes elev. - datum

3) When $h_2 > h_1$, backwash is not reduced, thus

   $B = [1-(h_1/h_2)^{3/2}]$

The MLIRD outlet structure consists of six, 6-foot diameter pipes controlled by circular gates. When a gate was below the water surface, flow was calculated assuming the opening acts as a nozzle with a constriction coefficient of 0.61 (Henderson, 1966) and a datum of 1040.9 feet (317.3 m). When a gate was above the water surface, the flow rate was determined by the following equation (Henderson, 1966):

$$Q = .48\ (g^{.5})(D^{.6})E^{1.9}$$

where Q = flow (cfs)

g = gravitational acceleration (ft/s^2)

D = pipe diameter (feet)

E = lake elevation - datum (feet)

The actual flow was assumed to be 90% of this theoretical flow due to energy loss. Prior to 1980, flow equations for the MLIRD outlet were given by Patmont (1980).

Other outflows include irrigation pumpage and evaporation. Irrigation pumpage was assumed to remain the same from one year to the next yet vary from month to month as given by Sylvester and Oglesby (1964). This is a valid assumption based on the consistency of irrigation amounts between years in the watershed (Bain et al., 1988). Evaporation was calculated from pan evaporation data at O'Sullivan Dam provided by the USBR or from a nomograph as specified by Patmont (1980). Subsurface exfiltration was assumed to be negligible, because the groundwater table surrounding the lake is at a higher elevation than the lake (Bain et al., 1985).

The U.S. Geological Survey (USGS) gauges both Rocky Ford Creek (Station 13) and Crab Creek (Station 3). The USBR provides records for East Low Canal discharge into Rocky Coulee Wasteway. This wasteway's flow is not gaged but was estimated by a conductivity mass balance from 1977 through 1979 (Patmont, 1980). Use of the average monthly flow determined by this method (3×10^6 m^3/mo) continued in the 1980s, because flow from this spring-fed wasteway should have remained relatively constant and suspected downstream inputs of high conductivity groundwater to Crab Creek often invalidated the mass balance method. A higher flow was assumed in 1984, because there was significantly higher than normal precipitation that year. Precipitation was obtained for either the Moses Lake or Ephrata station from National Oceanic and Atmospheric Administration (NOAA) climatological data.

Lake surface elevations were obtained from either the USBR or USGS. These data were used to calculate both outflows and changes in storage.

Nutrient Budgets

Budgets were constructed for TP only and during May - September. TP input, or load, from the various sources (Crab Creek, sewage effluent, and Rocky Ford Creek) was computed as the sum of the products of mean flow for each period and the P concentration for that period. Periods were chosen such that the determined P content was the mid point. Groundwater input was the product of water budget residuals (excess of input minus output and evaporation) and P content determined at Station 14.

Net internal phosphorus loading and net sedimentation (S) were determined from the following mass balance:

$$S = J_{ext} - TP_{out} - \Delta \text{ Lake TP}$$

where J_{ext} = External TP load

TP_{out} = TP in irrigation pumpage water and leaving through the USBR and MLIRD outlets

Δ Lake TP = Change in volume weighted lake TP

A negative S indicates positive net internal phosphorus loading. A positive S indicates that net sedimentation has occurred.

The amount of phosphorus entering or leaving the lake was calculated as a summation of each flow multiplied by its respective total phosphorus concentrations. The averaged TP from the surface and transect in the South Lake was assumed to represent the outflow concentration. The TP loadings from dilution water in East Low Canal (Station 1) and Rocky Coulee Wasteway (Station 2) were found from:

$$J_{ext(1\&2)} = TP_2(Q_1 + Q_2)$$

where $J_{ext(1\&2)}$ = mass to TP passing Station 2
TP_2 = TP concentration at Station 2
Q_1 = dilution water flow volume
Q_2 = flow volume of Rocky Coulee Wasteway

The individual contribution from Rocky Coulee Wasteway determined by mass balance was then:

$$J_{ext2} = J_{ext(1\&2)} - TP_1Q_1$$

This produced a negative value if $TP_1 > TP_2$. This situation occurred briefly during the Spring of 1986 and 1987, perhaps indicating that significant phosphorus quantities either settled out or were utilized by periphyton.

An average TP groundwater concentration was used during the 1980s for the following reasons. The May through September TP concentration from the spring (Station 14) varied from 48 µg/L in 1982 to 146 µg/L in 1986, although it would be expected that the contents of an aquifer would remain rather stable. The SRP fraction of the total varied from 0.3 in 1986 to 1.0 in 1979, although that too would be expected to remain stable. This value should approach 1, because an aquifer would not be expected to have the velocity required to carry particulates. Therefore, the higher phosphorus values are probably due to stirred up sediment particles. During the first three years of groundwater sampling (1977, 1979, and 1980), collection was made in a stream flowing out of the hillside at Station 14 where sediment entrainment was not a problem. SRP was averaged for the nine years for which there were adequate data and then divided by the SRP fraction for the first three years. This yielded a fraction of 0.75 and a resulting TP concentration of 51 µg/L. This value was compared with concentrations from two wells (W-3 and W-4) and one spring (SP-7) which were close to the lake. The wells were monitored in 1983 and 1986 and yielded a Spring-Summer, two-year average of 50 µg/L (Bain, 1987), which supports using the value of 51 µg/L.

The whole lake TP was found by allocating volumes to each lake sample as specified in Table 1.

Nitrogen budgets were computed in 1977 and 1978, but the uncertainty of so many unknowns (N fixation and denitrification) makes interpretation difficult. The role of N has been considered more specifically with flow-weighted inflow nitrate concentrations, which was a good indicator of lake algal biomass when N was exclusively limiting growth.

Relative Thermal Resistance

The dimensionless quantity, Relative Thermal Resistance (RTR), is defined as the difference in densities between two water layers divided by the density difference of water at 4°C and water at 5°C (Wetzel, 1983). This is directly related to the energy required to mix those layers, and therefore is largely a function of wind power. Temperature also affects RTR, because density differences become significantly greater at higher temperatures. The RTR of the entire water column was used here as an index of the energy required to mix the column.

Prior to 1980, no surface temperatures were taken at Station 12, so that RTR in Rocky Ford Arm could not be computed for that period. Because this arm represents 61% of the lake volume, the effects of thermal resistance on internal phosphorus loading were not evaluated for those years.

Algal Dominance Experiment

An experiment was conducted in Lower Parker Horn (7) and Lower Pelican Horn (10) from June 30 to July 10, 1986 to determine if the buoyancy and dominance by blue-green algae were caused by changes in pH/CO_2 in the water column. Pelican Horn has always been dominated by green algae, even since sewage diversion, while blue greens have dominated the blooms in Parker Horn. Thus, the goal was to determine if these differences in algal composition were related to changes in water column pH/CO_2.

Both sites have maximum depths of about 5 m and are oriented northeast to the lake's main axis. Water samples were collected in the morning (0700) and afternoon (1400) every other day from surface, 0.5, 1.5 and 4.0 m at two sites in each of Parker and Pelican Horns. Samples were analyzed for pH, alkalinity, nitrate-N, chl a, and biovolume. Free CO_2 (or H_2CO_3*) was determined from in situ pH and alkalinity using equilibrium constants from Stumm and Morgan (1981) and corrections for conductivity from Ahlgren and Ahlgren (1975).

Phosphorus Model

The model used to describe changes in lake TP is based on the following mass balance (Vollenweider, 1975; Larsen et al., 1979).

$$(dTP/dt) = (J_{ext}/V) - \rho\ TP - \sigma\ TP \qquad (1)$$

where TP = Volume weighted total phosphorus (μg/L)

J_{ext} = External phosphorus load (mg/year)

V = Lake Volume (m^3)

ρ = Flushing rate (1/year)

σ = Sedimentation rate coefficient (1/year)

The steady state derivation of the above is:

$$TP = J_{ext}/[V(\sigma+\rho)] \qquad (2)$$

The major assumptions of this model are that:

1) The lake is completely mixed.

2) Steady-state has been reached.

3) Phosphorus sedimentation is proportional to the lake's concentration.

The first assumption is invalid for Moses Lake because the lake's various arms do not completely mix with each other and the lake does stratify. Lack of complete horizontal mixing was dealt with by segmenting the lake as described in the following section. Stratification is not a major problem if column weighted means are used, but is a problem if a certain stratum's phosphorus concentration is to be predicted. Although the second assumption is not completely met, all but Rocky Ford Arm were predicted to reach at least 95% of steady-state during the summer based on flushing rates. The third assumption's validity is predominantly based on the model's accuracy. Using an average σ, Jones and Bachmann (1976) found a high correlation ($r = 0.918$) between predicted and actual summer mean TP concentrations for 143 lakes.

If phosphorus can be predicted and also limits algal growth, then an empirical model can be used to predict chl a from TP (Jones and Bachmann, 1976).

Lake Segmentation

The lake was divided into segments due to the observed differences in nutrient concentrations, chl a, and limiting nutrient. This was done with the intent of concentrating on Lower Parker Horn and the South Lake because they can be most influenced by dilution water and are used more by recreationists than either Rocky Ford Arm or Pelican Horn.

The lake was segmented as shown in Figure 2. TP outputs from Stations 7, 8, and 10 were predicted and then used as inputs for modeling the TP concentration at Station 9.

The amount of groundwater entering each segment was allocated as follows: Pelican Horn - 80%; Parker Horn - 10%; Rocky Ford Arm - 10%. The 80% contribution to Pelican Horn was estimated by Carlson (1983) from a sodium mass balance. A groundwater contour map was employed with similar results by noting that flow is proportional to the gradient of the groundwater level, and assuming that the hydraulic conductivity of the soil was the same around the lake. The remaining flow was estimated from this second approach to be equally distributed between Parker Horn and the Rocky Ford Arm.

Water pumped from Upper Parker Horn into Pelican Horn was assumed to have the same TP concentration as the combined flows from Stations 1, 2, and 3, due to the pump's proximity to the mouth of Crab Creek. For this reason, it was also assumed that the pumped water never entered Parker Horn. The irrigation water pumped from Pelican Horn was assumed to have a TP concentration equal to the column weighted mean TP at Station 10.

Model Calibration

All of the variables in the TP model could be adequately measured except σ. The equation is typically calibrated by assuming $\sigma = \rho^n$, where n is an

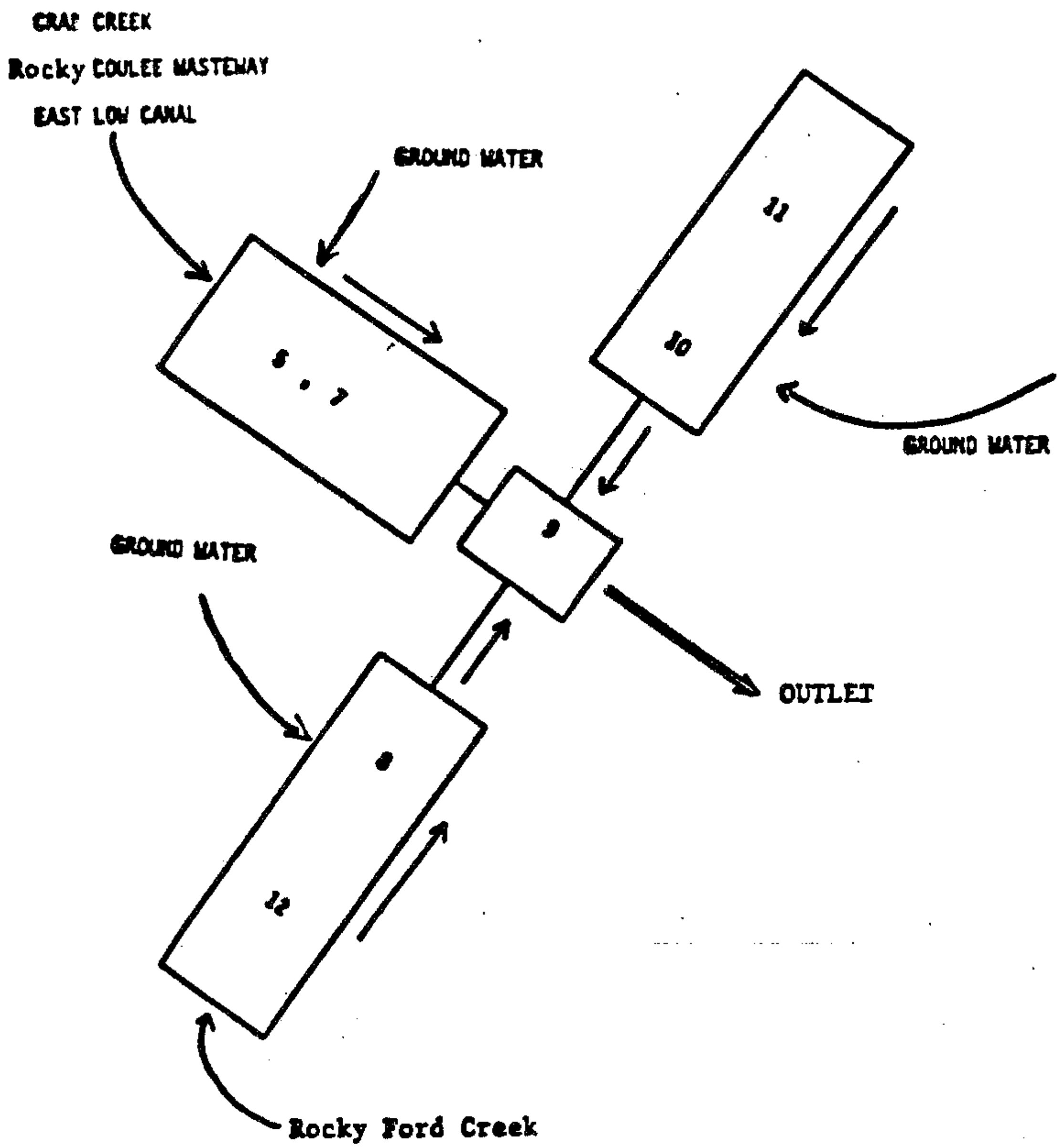


Figure 2. Assumed water movement and basin divisions for the TP models (Modified from Okereke, 1987).

integer between 0 and 1 (Larsen and Mercier, 1976; Shuster, 1985). In lakes or lake sections such as Parker Horn, which have negative sedimentation rate coefficients, obviously σ cannot equal ρ^n, regardless of n. The reason σ can be negative is that the rate of TP transport out of the sediment, which may exceed the gross TP sedimentation rate, is included within this coefficient. As previously pointed out, RTR and ρ are believed to significantly affect this upward transport of P in Moses Lake. Therefore, σ was regressed against both these variables in calibrating the model.

Error Analysis

A spatial estimate of error was derived for the May through September measured mean TPs. Variability was assessed from two years of data and extrapolated to the other years assuming that variability was a percentage of the mean. Sampling and spatial variability were estimated from replicate transect data collected twice during the Summer of 1983 (Table 3). The mean coefficient of variation (CV) for TP using the four data sets was 10% for the identical transects and 9% for the combination of identical and irregular transects. Because the addition of irregular transects to this error analysis did not increase the CV, the implication is that sampling variability is approximately 10% and the spatial variability is negligible when sampled by transect within a segment of Moses Lake. The sampling error for the May through September mean was found by dividing this 10% by the square root of the number of samples (Reckhow and Chapra, 1983).

The temporal variability was estimated by using transect data at Stations 7, 8, 9, and 10 for 1977 when samples were collected weekly (see Jones, 1988). Every other trip was grouped to simulate a pair of twice monthly sampling regimes, and every fourth trip was selected to form four monthly sampling regimes. The average difference between monthly and weekly means was 12% and between twice monthly and weekly was 5.3%. It was assumed that the wekly values represented the true value and therefore a temporal CV of 5.3% was used. The sampling CV and temporal CV were combined using the following:

$CV_2 = (CV_s^2 + CV_t^2)^{1/2}$ (Reckhow and Chapra, 1983)

where CV_c = combined CV

CV_s = sampling CV

CV_t = temporal CV

The 95% confidence interval was found using the student-t statistic. For a summer mean with 10 samples, this interval was found to represent +/- 14% of the mean.

Results

Dilution Water and Pumping

Low nutrient dilution water has been added to Parker Horn during the spring, and sometimes during the summer as well, since 1977 (Table 4). There

Table 4. Volume and flow rate of Columbia River dilution water added to Parker Horn and the periods of pumping -(1.43 m^3/s) Parker Horn water to Pelican Horn during 1977-1985.

Year	Dilution Period	10^6 m^3	m^3/s	Pumping Period	10^6 m^3
1977	3/20-5/7	139.3	33.9		
	5/22-6/4	11.8	10.6		
	8/14-9/18	52.3	17.5		
	(96 days)	203.4			
1978	4/20-6/18	112.5	21.9		
	(60 days)				
1979	4/3-6/4	134.5	25.3		
	7/11-8/28	67.6	16.5		
	9/20-10/18	56.1	23.4		
	(138 days)	258.2			
1980	2/18-3/1	5.8	6.2		
	5/13-6/10	27.6	11.5		
	(39 days)	33.4			
1981	3/27-4/15	13.9	8.6		
	4/16-4/22	9.9	19.3		
	4/23-4/29	13.8	26.9		
	4/30-5/15	27.3	21.3		
	(49 days)	64.9			
1982	3/28-4/3	5.1	8.6	7/1-9/30	8.8
	4/4-5/14	48.9	14.3	(72* days)	
	5/15-7/9	107.6	22.9		
	7/10-7/23	15.9	14.3		
	7/24-8/5	8.8	8.6		
	(126 days)	186.3			
1983	4/3-4/27	35.2	17.1	4/2-9/14	20.2
	4/28-7/6	50.6	8.6	(165 days)	
	(94 days)	85.8			
1984		---		4/10-10/1	21.3
				(174 days)	
1985	4/21-4/24	2.2	8.6	4/24-10/20	21.9
	4/25-8/7	166.9	18.7	(179 days)	
	8/8-9/10	27.2	9.6		
	(140 days)	196.3			

*pump inoperative for 20 days in period

Table 4 (continued)

Year	Dilution Period	10^6 m^3	m^3/s	Pumping Period	10^6 m^3
1986	5/1-5/14	20.55	15.86	4/8-9/25	21.0
	5/15-6/4	14.68	8.09	(170 days)	
	6/15-7/2	32.37	20.82		
	7/3-7/9	5.99	9.91		
	8/20	0.73	8.50		
	8/21-8/27	12.36	20.43		
	(63 days)	86.69			
1987	5/1-5/13	18.79	16.73	4/3-10/13	23.9
	5/14-6/3	46.25	25.49	(193 days)	
	6/4-6/10	10.28	16.99		
	6/10-6/17	5.68	8.21		
	6/18-6/24	0.51	0.85		
	(51 days)	81.50			
1988	3/27-4/2	9.42	15.58	4/5-10/4	22.5
	4/3-4/30	129.32	53.45	(182 days)	
	5/1-6/11	90.78	25.02		
	6/12-7/2	16.10	8.87		
	8/28-9/18	10.28	5.41		
	(115 days)	255.89			

was no dilution water added in 1984, the year sewage effluent was diverted. However, water was added through the summer period during three of the past four years (Table 4). Dilution is most effective if additions continue throughout the summer because of the quick (within a month) return to Crab Creek nutrient levels in Parker Horn once dilution water addition has ceased (Welch et al., 1984).

The volumes of dilution water added have been highly variable from year-to-year, depending on the demands for irrigation water from Pot Holes reservoir, which receives the outflow from Moses Lake. Pot Holes must have storage capacity if dilution water is routed from the East Low Canal through Moses Lake. Low demand for irrigation water from Pot Holes during cool summers reduces the capacity available to accommodate increased flows of diluted Moses Lake water. Except for 1984, the smallest volume added was in 1980 following the Mount St. Helens ashfall, which resulted in low irrigation demand. Average flow rates of dilution water into Parker Horn have ranged from 6.2 to 53.5 m^3/s or a time-weighted annual mean flow of 24.2 m^3/s (169.4 X 10^6 m^3) for the 971 days over 12 years (Table 4). That time-weighted mean flow represents a mean flushing rate in Parker Horn (12.32 X 10^6 m^3) of 17%/day for the days when water actually entered. The lake would more realistically respond to an average April - September (183 days) flushing rate in which the 12-year average dilution water input was 130 X 10^6 m^3 (8.2 m^3/s) or 5.8%/day. If the normal Crab Creek flow is included, the mean flushing rate is 7.8%/day. The maximum April - September mean flushing rate was 11%/day, which occurred in 1982 and 1988. The short period, maximum flushing rate was in 1988 during April 3 - April 30 - 39%/day and the average over the nine years with short-period high inflows was 20%/day (Table 4).

The normal April - September flushing rate for the whole lake was 0.27%/day. The 12-year mean flow of dilution water (8.2 m^3/s) for the six-month period increased the whole lake flushing rate 2.7 times to 0.73%/day. Of course dilution water did not reach all parts of the lake uniformly, but the lake basin is relatively flat providing extensive exposure and long fetches for effective mixing by the wind (Welch et al., 1982). As a result, and with the exception of remote parts of upper Rocky Ford Arm, distribution of dilution water throughout the lake was rather complete, especially with pumping from Parker Horn to Pelican Horn (Figure 1).

Pumping of Parker Horn water through Pelican Horn was a consistent process (1.4 m^3/s or 50 cfs) that included most of the six-month period (Table 4). That flow provided a flushing rate for all of Pelican Horn of 1.2%/day, but in the shallow upper and middle sections, the rate was 7%/day.

Lake Quality

Nutrients, chl a and transparency: The improvement in lake quality of at least 50%, in terms of nutrient and algal concentrations, and 100% in water transparency since dilution began in 1977 has been extensively documented (Welch and Patmont, 1980; Welch et al., 1982; Carlson and Welch, 1983; Welch et al., 1984; and Welch and Weiher, 1987). Actually, there have been further improvements in recent years (1986-1988). Lower Parker Horn (7) and South Lake (9) have been the principal lake sections used to indicate overall response to dilution. The 1988 results show the greatest improvement thus far; TP and chl a were the lowest ever observed in Lower Parker Horn (42 and 13 μg/L) and South Lake (40 and 7 μg/L) representing a respective decrease from 1969-1970 of 72% and 82% in Lower Parker and 74% and 84% in South Lake (Table 5). The highest average transparency yet at either site was 2.4 m in South Lake. That is a 240% improvement compared to pre-treatment years. The value in Lower Parker was the second highest recorded for a 268% improvement.

The response in Lower Parker Horn and South Lake is illustrated in Figures 3-5. TP appears to have finally declined below the 50 μg/L level, which was an original goal of the dilution project. Chl a also appears to have reached a lower equilibrium level in South Lake averaging 12 ug/L during 1986-1988 compared to 18 μg/L prior to 1984, while those two means were similar in Lower Parker Horn (24 and 21 ug/L). Transparency shows little or no trend throughout the dilution years, in spite of the exceptionally high value (2.4 m) in South Lake in 1988 (Figure 4). Years with relatively higher transparency were also years of large volumes of dilution water; 1977, 1979, 1982, 1985 and 1988 (Table 4). This effect on clarity, independent of algal content, is due to dilution of non-algal particles. Similarly, very poor transparency existed in 1984, the year with no dilution water, even though there was no appreciable change in chl a from previous years.

The very high concentrations of TP and chl a in 1985 were obviously unusual, especially considering that nearly 200 X 10^6 m^3 of dilution water was added that year over the longest period (140 days) in the 12 dilution years (Table 4). While transparency was comparable to other dilution years, TP, chl a, and NO_3 equaled and usually exceeded levels in other dilution years, except for 1980, the ashfall year (Table 5). The cause was probably due to unusual water-column mixing and high internal loading of TP that year and will be discussed later in relation to TP loading and modeling.

While these sections of the lake are no longer hypereutrophic, they are still eutrophic, which is often characterized as TP > 25 μg/L, chl a > 10 μg/L and transparency < 2 m (Welch, 1988). Means for the last three years in Lower Parker and South Lake are 45 μg/L TP, 17 μg/L chl a, and 1.6 m transparency.

Levels of chl a, and transparency in the Rocky Ford Arm have remained rather stable since dilution began, although there has been a decrease in TP (Table 5). Unfortunately, Station 12 was not established in pre-treatment years so relative improvement from dilution can not be determined. Although dilution water reaches well into the Rocky Ford Arm (12) producing around one third dilution water at times, it receives less input than Lower Parker and South Lake. Nevertheless, quality has usually been as high and sometimes higher in the Rocky Ford Arm (Table 5).

The response in Pelican Horn is quite different, because it was largely isolated from dilution until pumping from Upper Parker Horn began in 1982 and sewage effluent was diverted (from 11) in 1984. Only TP showed a significant decrease (35%) in Pelican Horn following pumping (Figure 2). Chl a changed little and transparency not at all (Figures 4-5). Actually, there was a very substantial decrease in algal biovolume due to washout, especially in Upper Pelican Horn (19) where dilution water input raised the flushing rate to 19%/day. Chl a did not decrease in proportion to algal biovolume, however, because the algae in Pelican Horn were severely limited by N and as a result

Table 5. May - September Transect Means For Each Station, 1969-70, 1977-88, in Moses Lake

	Upper Pelican (19)	Middle Pelican (11)	Lower Pelican (10)	South Lake (9)	Cascade (8)	Rocky Ford (12)	Lower Parker (7)	Upper Parker (5)
Secchi Depth (m)								
1969-70	0.40	0.40	0.90	1.00	1.00		0.60	0.50
1977			1.30	1.80	1.70		1.30	0.90
1978		0.40	1.10	1.70	1.80	1.40	1.20	0.80
1979		0.50	1.20	1.70	2.10	1.80	1.50	1.30
1980		0.30	0.70	1.30	1.60	1.60	1.00	0.60
1981				1.40	1.60	1.50	1.20	
1982	0.40	0.40	0.90	1.70	2.60	1.70	1.70	1.20
1983	0.40	0.30	0.80	1.20	1.70	1.40	1.40	1.00
1984		0.40	0.80	1.10	1.10	1.10	0.90	0.90
1985		0.56	1.00	1.43	1.55	0.56	1.48	1.36
1986		0.57	0.95	1.26	1.70	1.26	1.40	1.11
1987		0.59		1.37		1.67	1.44	1.39
1988		0.80		2.39		1.70	1.61	1.42
Chlorophyll a (ug/L)								
1969-70	50	48	54	42	34		71	101
1977			30	26	24		33	46
1978		39	17	15	9	15	16	17
1979		39	27	23	18	19	29	23
1980		41	18	11	9	21	18	27
1981		46	20	19	19	24	26	
1982	40	41	40	19	9	16	15	23
1983	22	41	20	16	13	17	24	30
1984*		30		14		12	27	
1985	16	20	63	41	12	21	71	23
1986		13	10	9	22	17	16	22
1987		11		20		20	34	26
1988		11		7		15	13	10
NO3+NO2-N (ug/L)								
1969-70	11	10	38	25	87		69	181
1977			41	50	75		73	185
1978		26	10	16	51	68	43	213
1979		43	12	19	28	40	52	211
1980		50	92	148	169	173	285	425
1981		31		34	79	118	71	
1982	219	39	46	20	32	69	90	179
1983	113	28	20	27	74		65	175
1984	108	19	56	43	138	157	159	337
1985	65	30	57	75	147	131	141	174
1986		24	57	9	35	29	32	431
1987	96	29		10		34	24	103
1988	32	12		18		49	61	215

	Upper Pelican (19)	Middle Pelican (11)	Lower Pelican (10)	South Lake (9)	Cascade (8)	Rocky Ford (12)	Lower Parker (7)	Upper Parker (5)
Phytoplankton Volume (mm^3/L)								
1969-70							25	
1977				8				
1978				7			8	
1979				23			20	
1980				5			10	
1981				5		7	7	
1982	37	27	19	8			8	7
1983		33	15	10			27	
1984		16	7	5			15	
1985		19		38		25	78	
1986		17		6		15	8	
1987		10		11		10	19	
1988		11		3		8	6	
Total P (ug/L)								
1969-70	559	920	186	156	133		152	189
1977			93	90	67		78	96
1978		715	99	86	67	68	58	85
1979		533	111	83	66	82	67	82
1980		668	125	87	70	84	83	113
1981		718	121	80	58	65	67	
1982	244	617	138	79	55	55	54	45
1983	125	332	138	88	54	89	64	60
1984	131	230	81	56	59	60	64	82
1985	82	93	93	88	62	98	116	49
1986		76	46	42	48	64	45	58
1987	76	70		42		48	54	65
1988		84		40		61	43	45
SRP (ug/L)								
1969-70	269	643	52	48	39		28	30
1977			60	55	37		26	20
1978		420	22	24	14	11	9	9
1979		462	27	27	16	18	9	13
1980		367	27	23	18	12	9	7
1981				21	7	7	5	
1982	25	364	35	29	10	10	5	5
1983	13	189	49	40	23	19	13	7
1984	9	32	8	7	11	10	5	6
1985	0	4	4	5	19	23	4	13
1986		9	7	9	5	7	10	8
1987	3	5		4		6	4	5
1988		4		7		6	5	5

	Upper Pelican (19)	Middle Pelican (11)	Lower Pelican (10)	South Lake (9)	Cascade (8)	Rocky Ford (12)	Lower Parker (7)	Upper Parker (5)
Total N (ug/L)								
1969-70				(1400)			(1500)	
1977			605	614	605		531	650
1978		854	558	523	437	498	438	656
1979		884	698	556	496	587	518	697
1980		1,027	692	634	669	812	726	961
1981		812	500	510	494	538	516	
1982	1,215	1,090	520	647	429	560	438	386
1983	752	712	628	431	398	505	464	740
1984		1,099	833	761	815	935	697	1,044
1985	896	1,012	1,147	1,144	895	1,034	814	705
1986		515	577	616	580	891	506	675
1987		178		332		356	486	526
1988		504		213		371	252	361

* Calculated from biovolume-chl a relationships (Weiher, 1986)

TOTAL PHOSPHORUS

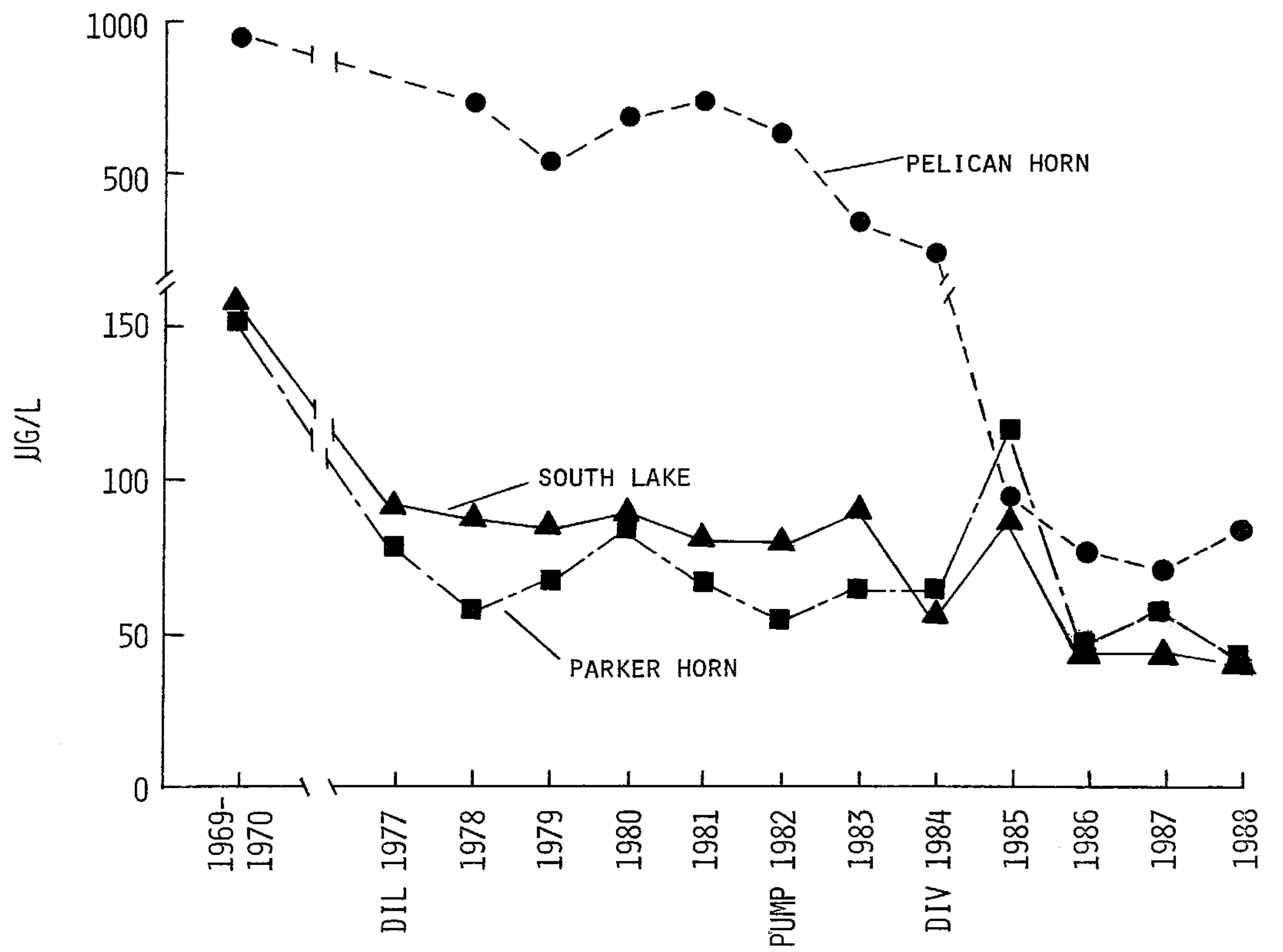


Figure 3. Mean total phosphorus concentrations from transect samples (0.5m depth) in three sections of Moses Lake (7,9,11) during May-September in pre-treatment years (1969-1970) and post-treatment years (1977-1988). DIL - start of dilution water addition to Parker Horn, PUMP = start of dilution water pumping from Upper Parker Horn to Upper Pelican Horn and DIV = diversion of sewage effluent from Middle Pelican Horn.

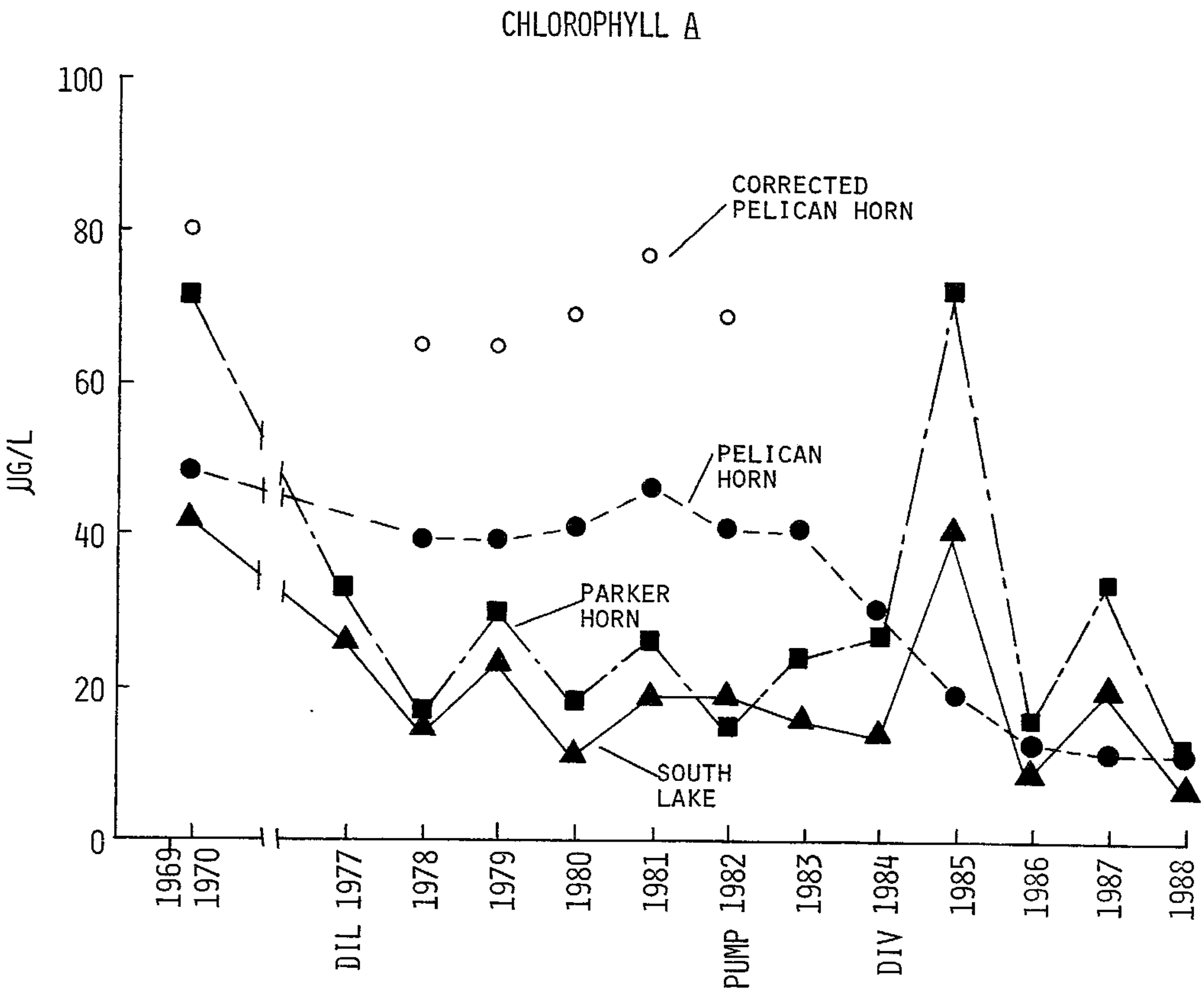


Figure 4. Mean chlorophyll concentrations from transect samples (0.5m depth) in three sections of Moses Lake (7,9,11) during May-September in pre-treatment years (1969-1970) and post-treatment years (1977-1988). See Figure 3 for legend.

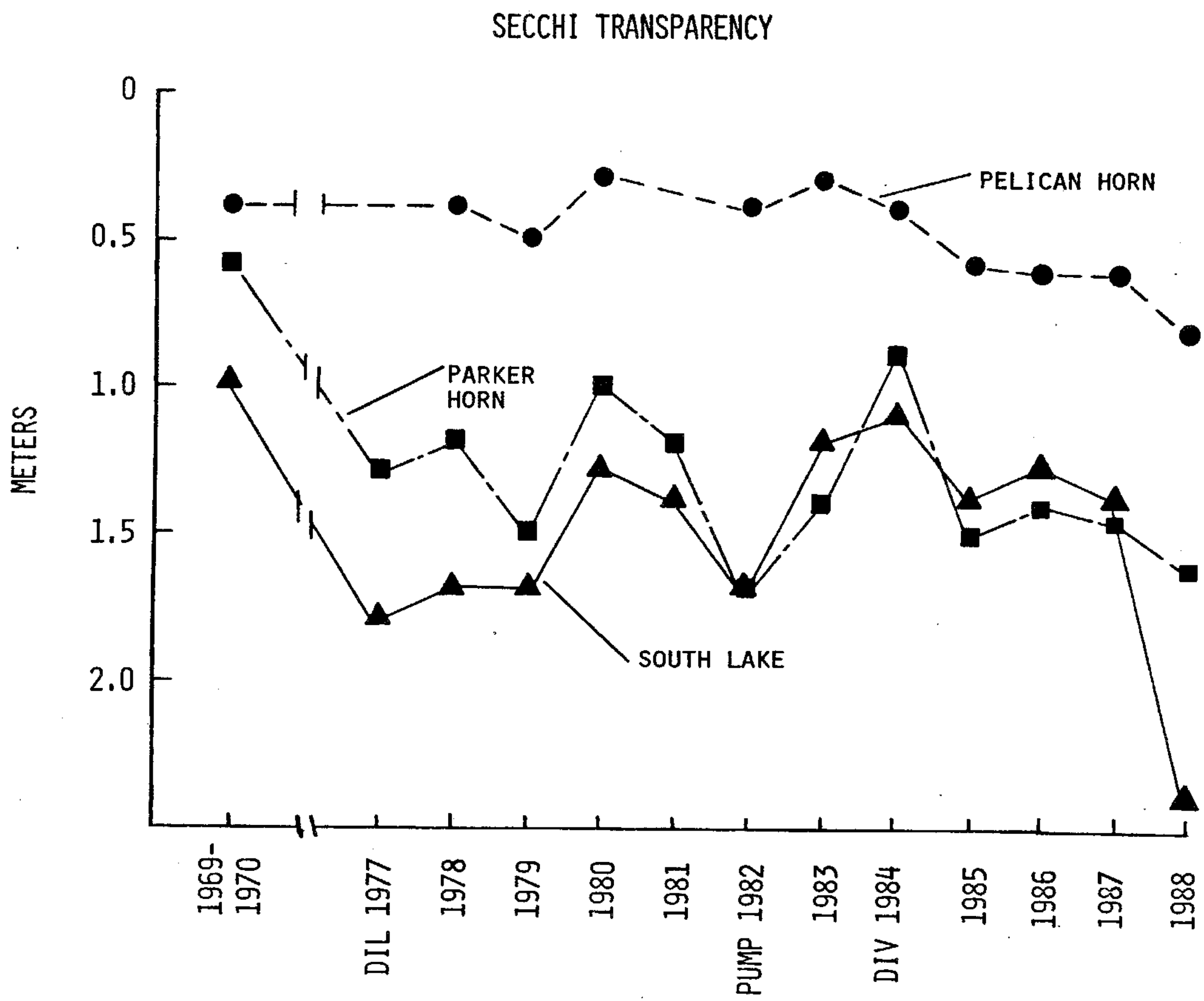


Figure 5. Mean Secchi transparency in three sections of Moses Lake (7,9,11) during May-September in pre-treatment years (1969-1970) and post-treatment years (1977-1988). See Figure 3 for legend.

had low cell chl a content. Although biovolume decreased in Pelican Horn, cell chl a increased in response to increased N input (Carlson, 1983).

As expected, diversion dramatically and promptly reduced the TP content in Pelican Horn to what appears to be an equilibrium level of about 10% of the pre-treatment level (Figure 3). Sewage effluent and ground water were the only sources of P to Middle Pelican Horn, so diversion reduced loading by about 96% (sewage effluent contained 100 times the TP concentration in ground water). The concentration in Pelican Horn (1986-1988 mean of 77 μg/L) is still higher than in Lower Parker Horn and South Lake (1986-1988 mean of 45 μg/L). Chl a has also declined promptly to a level comparable to other parts of the lake (Figure 4). Although algae in Pelican Horn still appear to be N limited (1986-1987 chl a:biovolume ratio = 0.95), the N:P ratio (NO_3-N:SRP) has increased from a mean of 0.34 before (1982-1984) to 4.8 after diversion (1986-1988). The N:P ratio in 1969-1970 was 0.02. Algae in Parker Horn have a chl a:biovolume ratio of 1.8 ± 0.9 (n=10). To illustrate how chl a would have compared in pre-treatment years, chl a in Pelican Horn was corrected based on the chl a:biovolume ratio in Parker Horn (Figure 4).

SRP in Middle Pelican Horn decreased even more dramatically, from a pre-treatment level of 643 μg/L in 1969-1970 to around 4 μg/L the last two years. That represents a decrease of more than 99%. There is now more NO_3 than SRP in Pelican Horn (Table 5).

In spite of the improvement in nutrient content and algal biomass, transparency in Pelican Horn is still rather low although it has in fact doubled from the 0.4 m, pre-treatment level (Table 5; Figure 5). The shallowness of Upper and Middle Pelican Horn and the large concentration of carp may prevent any further improvement in transparency.

The extent to which sewage diversion from Pelican Horn may have affected remote Rocky Ford Arm (12) and Lower Parker Horn, as well as the effect of other watershed measures (BMPs in 1987 and detention basin on Rocky Ford Creek in 1985) is of interest. Mean nutrient concentrations were lower during 1986-1988 throughout the lake, although in some instances the variability was so great (especially with NO_3) that the differences are not statistically significant (Table 6). The differences in TP and SRP are greatest in Middle Pelican Horn and South Lake, and least in Lower Parker and Rocky Ford Arm, which would be expected if the effect were due to sewage diversion because the latter lake sections are farther from the source. Some effluent was probably transported to those areas by wind action, however, the movement would have been impeded by the I90 bridge constriction and the opposing general direction of flow. NO_3 also decreased markedly in all areas, although as mentioned above there were large variations (Table 6). Although the timing is appropriate to ascribe the cause for the decreases in nutrient content at the two uplake Stations (7 and 12) to diversion, the case is not clear-cut, due to the above mentioned factors and the fact that changes were greater and more significant at the sites closest to the effluent. Also, as noted before, chl a and transparency did not change in Rocky Ford Arm or Lower Parker Horn from before to after diversion, although they did change in Pelican Horn and South Lake.

The BMPs (water and nutrient conservation) that began in 1987 were probably not the cause for the observed nutrient decreases during 1986-1988.

Table 6. Comparison of transect nutrient content between data from 1977-1984, before sewage diversion, with 1986-1988, after diversion.

	(11) M. Pelican	(9) So. Lake	(12) R. Ford Arm	(7) L. Parker
TP				
1977-1984	555 ± 193	81 ± 11	72 ± 13	67 ± 10
1986-1988	77 ± 7	41 ± 1	58 ± 9	47 ± 6
SRP				
1977-1984	360 ± 104[1]	31 ± 12[1]	12 ± 4	10 ± 7
1986-1988	6 ± 3	7 ± 3	6 ± 1	6 ± 2
NO_3-N				
1977-1984	34 ± 11	30 ± 13[2]	90 ± 47[2]	79 ± 38[2]
1986-1988	22 ± 9	12 ± 5	38 ± 10	39 ± 20
TN				
1977-1984	925 ± 149	585 ± 101	634 ± 170	541 ± 111
1986-1988	399 ± 191	387 ± 207	539 ± 305	415 ± 141

[1] 1984 omitted because of non-equilibrium levels the diversion year
[2] 1980 omitted because of exceptionally high concentration the ashfall year

Only about one fifth of the irrigated area (1612 ha of 8380 ha) was actually included for water and nutrient controls (Bain, 1988). A 25% reduction in nutrient loss from the full area (8380 ha) would amount to only about a 5% reduction from that actually included (1612 ha). Five percent of the volume of undiluted surface and ground water flow in Crab Creek (28 X 10^6 m^3) divided by the sum of that flow plus the average dilution water input (81 X 10^6 m^3) for the May-September period amounts to a potential change in nutrient concentration of only 1.3%. To confirm the lack of effect of BMPs thus far, there has been no change in the May-September NO_3 content of Crab Creek (Table 7). The largest effect was expected in N (Bain, 1988). Therefore, the BMPs thus far implimented were probably not responsible for the observed nutrient reductions.

The reduction of TP in Rocky Ford Creek in 1985 may have influenced lake concentrations, especially in Rocky Ford Arm and Lower Parker Horn. TP loading from Rocky Ford Creek since 1985 has averaged 3600 $\pm$ 710 kg, compared to 4632 $\pm$ 919 kg prior to 1985, or a 22% decrease. The observed decrease in TP in Rocky Ford Arm for the same periods was 20%. Bain et al. (1985) had suggested that a 37% reduction in Rocky Ford Creek TP could be expected as a result of a detention pond finished in 1985 on the lower creek. The loading decrease may not be real due to the year-to-year variability in flow. This is indicated by a smaller decrease in volume-weighted TP concentrations between the same time periods (174 vs 151 μg/L). Results of nutrient content in samples collected upstream and downstream of the detention pond in 1987-1988 bear this out. They show essentially no effect of the pond (Table 7).

Algal composition: The algae in Moses Lake has always been dominated by blue-green species, primarily Microcystis and Aphanizomenon, but also including Anabaena, Lyngbya and Coelosphaerium. A typical seasonal pattern of composition change is illustrated for South Lake (9) in Figure 6. Diatoms bloom early in March and April, while blue greens begin to dominate in June. Summer blooms of Microcystis and Aphanizomenon can account for nearly all of the biovolume. This pattern is about the same throughout the lake and by mid July (day 196) three fourths or more of the biovolume is blue greens, except in Pelican Horn (Figure 7). Since dilution began, blue greens have averaged 66 $\pm$ 21 % of the biovolume in South Lake and 81 $\pm$ 12 % in Lower Parker Horn (Appendix A).

Upper and Middle Pelican Horn have been notorious for their green, opaque appearance in aerial photographs. That opaqueness is due to relatively small, green algae, such as, Gloeocystis, Pediastrum, Oocystis, and Scenedesmus. These taxa are common in sewage lagoons, which Middle and Upper Pelican Horn resembled prior to sewage diversion. Because light attenuation is a function of particles in the water, a given biovolume composed of small single cells will attenuate more light (and produce poorer transparency) than an equal biovolume of colonial cells that tend to group together, such as the blue greens, Aphanizomenon and Microcystis. There is some indication that blue greens became more common in Middle Pelican Horn after pumping and diversion began. The first summer after diversion (1984), there were two large blooms of primarily Anabaena. That was the first time such blooms of Anabaena were noted in Pelican Horn. Blue greens averaged 50% of the biovolume that summer compared to 21% the summer before (Weiher, 1986). Comparison with pretreatment years is unfortunately not possible, because algal biovolume was not determined until 1982, the year when pumping began, which also apparently

Table 7. Mean (SD) of NO_3-N, SRP and TP in Crab Creek (3), East Low Canal (Columbia River water) and Lower Crab Creek (4-R) during 1977-1979, before the ashfall and BMPs and after during 1986-1988 (n = 53-84 for 1977-1979 and 30-36 for 1986-1988). Rocky Ford Creek nutrient concentration means are given for 1987 and 1988 upstream and downstream from the retention pond.

	NO_3-N		SRP		TP	
	'77-'79	'86-'89	'77-'79	'86-'88	'77-'79	'86-'88
Crab Creek (n = 58 - 71)	693 ± 469	678 ± 387	18 ± 12	7 ± 7	92 ± 56	47 ± 21
East Low Canal (n = 53 - 61)	38 ± 76	30 ± 104	11 ± 12	4 ± 8	24 ± 12	19 ± 16
Lower Crab Creek (n = 74 -84)	488 ± 431	912 ± 819	14 ± 8	7 ± 4	57 ± 35	40 ± 33

	NO_3-N		SRP		TP	
Rocky Ford Creek	'87	'88	'87	'88	'87	'88
(n = 8 - 12) upstream	816 ± 400	794 ± 251	106 ± 8	107 ± 55	151 ± 11	176 ± 67
(n = 7 - 11) downstream	762 ± 483	576 ± 148	89 ± 11	108 ± 56	134 ± 16	170 ± 59

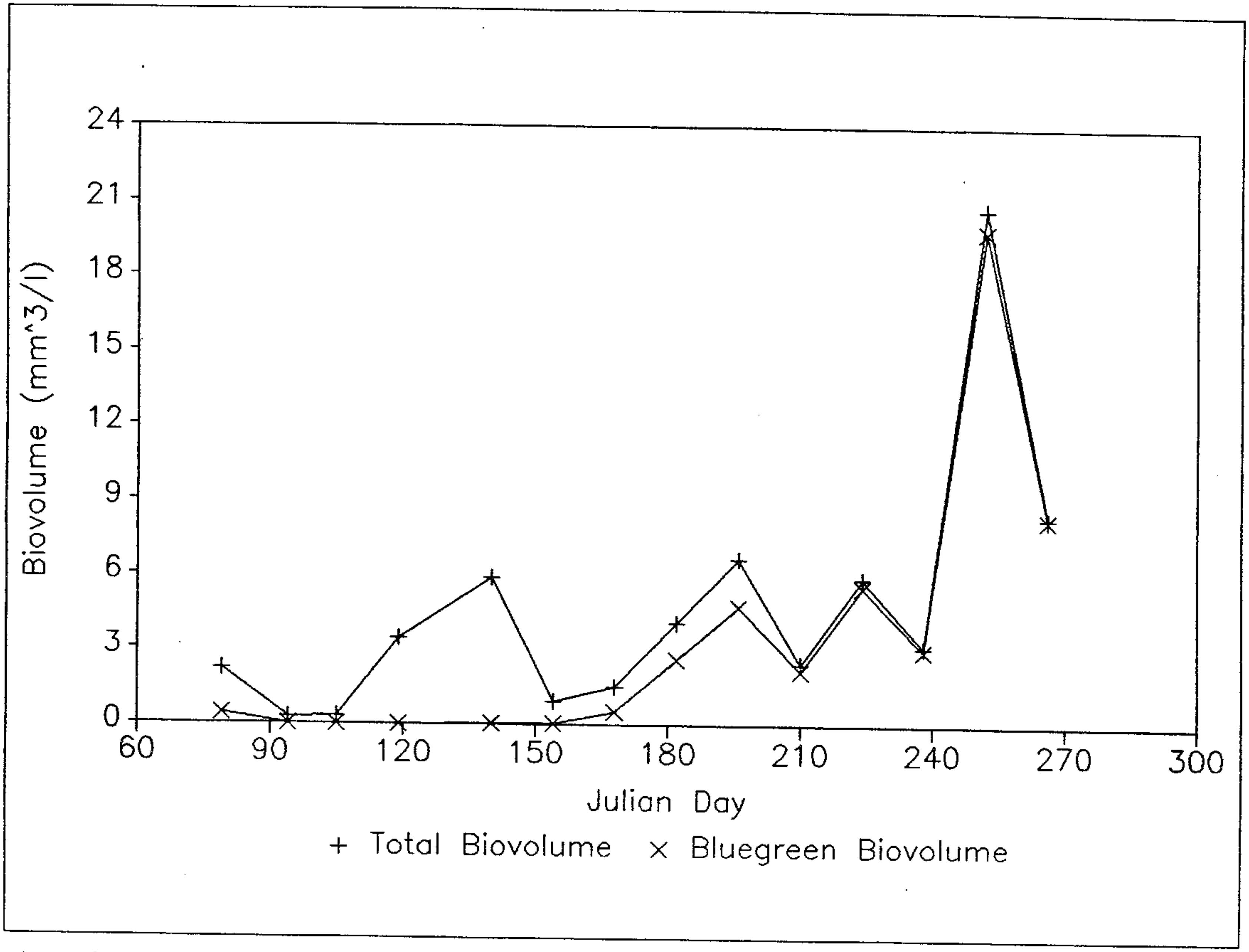


Figure 6. Total algal biovolume, and blue-green fraction, in South Lake (9) during March - September, 1986.

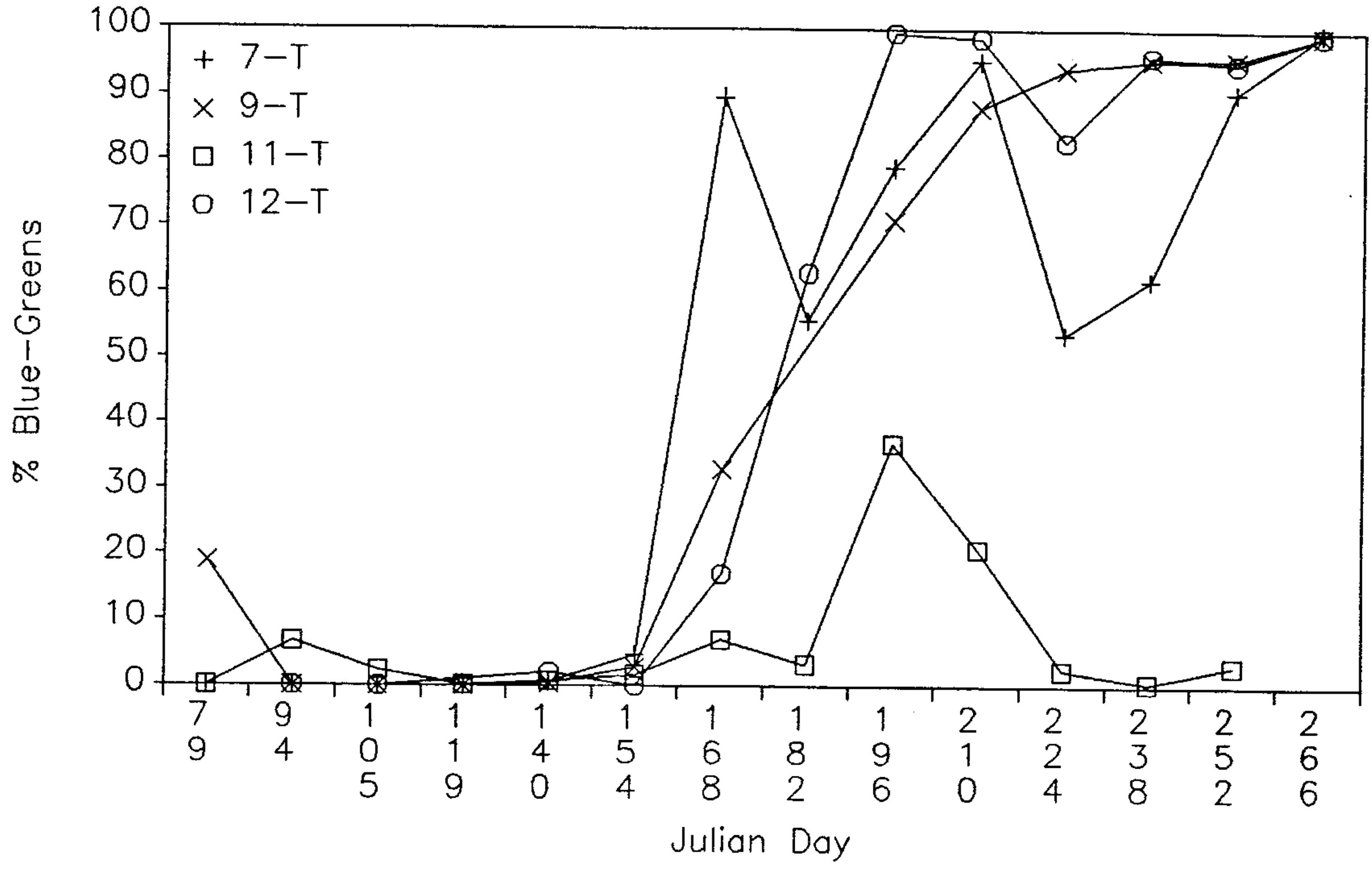


Figure 7. Percent blue-green algae in four sections of Moses Lake during March - September, 1986: Lower Parker Horn (7), South Lake (9), Pelican Horn (11) and Rocky Ford Arm (12).

promoted blue greens (24% of biovolume). A marked decrease in blue greens in Pelican Horn has occurred during the past three years (average about 5% of biovolume, Appendix A). Possible causes for seasonal dominance of blue-green algae will be discussed later.

Evaluating the effect of dilution on algal composition is also difficult because biovolume was not routinely determined in the pre-treatment years, 1969-1970. Instead, simple occurence and an index of similarity was determined (Bush, 1972). Some measurements of biovolume and percent blue greens were made in Upper Parker Horn (5) in 1970 as part of an in situ experiment (Buckley, 1971), which showed a summer average of 98% blue greens. Figure 8 shows that the mean summer fraction of blue greens in South Lake was much less than that (on the order of two thirds) in the first few years after dilution began (Welch and Patmont, 1980). However, the fraction seems to have increased in subsequent years.

Macrophytes: As a result of increased clarity, the rooted macrophytes *Potamogeton crispus* and *P. pectinatus* became relatively abundant in the shallow areas of Upper Parker Horn. Macrophytes were not noticeable during pretreatment years due to poor water transparency (mean 0.5 m) caused by high algal content, but roughly doubling transparency apparently was enough to trigger the development of relatively dense stands of those species.

In 1983 macrophytes were present to moderately abundant along the east shore of Upper Parker from the Alder Street Bridge about two thirds of the way to Lewis Horn. They were moderate to densely populated about half way from the bridge to Lewis Horn on the west side, but only present to moderately abundant the remaining distance to Lewis Horn.

Their density or distribution did not noticeably increase during the 1980s in spite of continued and even improved transparency (Table 5). Furthermore, macrophytes did not become abundant in other shallow waters of the lake. The reason is thought to be due to sediment type. Most of the shallow sediments along the shore are very course, even gravelly and probably not conducive to macrophyte development. If that were not the case, macrophytes should have developed in Lower Parker Horn and South Lake, because transparency has been greater there than in Upper Parker Horn (Table 5). Much of the shallow water sediments in Upper Parker Horn are organic, probably due to the downwind concentration of algae by prevailing southwesterly winds (Figure 1).

Nutrient Limitation Trends

Phosphorus and/or nitrogen: Through 1985, there was a trend of increasing N:P ratios (NO_3:SRP) in the lake and in the Crab Creek inflow (upstream from dilution water input), with little change noted in Rocky Ford Creek (Figure 9). This trend was ascribed to; 1) increased use of spray irrigation, which has progressively increased through the 1970s and 1980s to be 80% of the irrigation in the watershed and could have increased P retention in the soil, 2) diversion of sewage with its low N:P ratio, and 3) the Mount St. Helens' ashfall, which momentarily reduced internal loading of P and may also have increased P retention in the watershed soil (Welch and Weiher, 1987). The increases in the lake (7 and 9) appear to have been more related to the

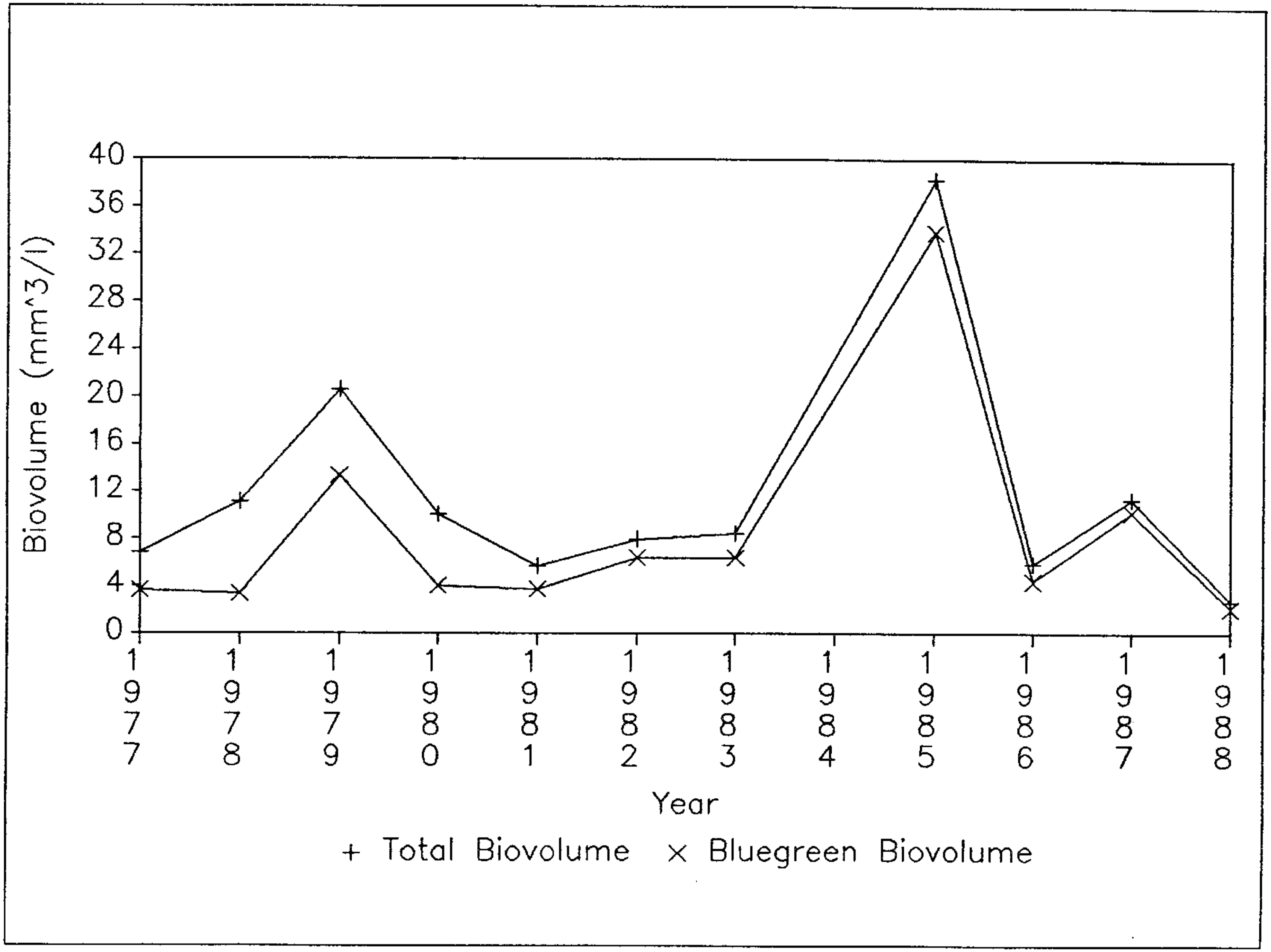


Figure 8. The mean May - September total algal biovolume, and blue-green fraction, in South Lake (9) during 1977-1988.

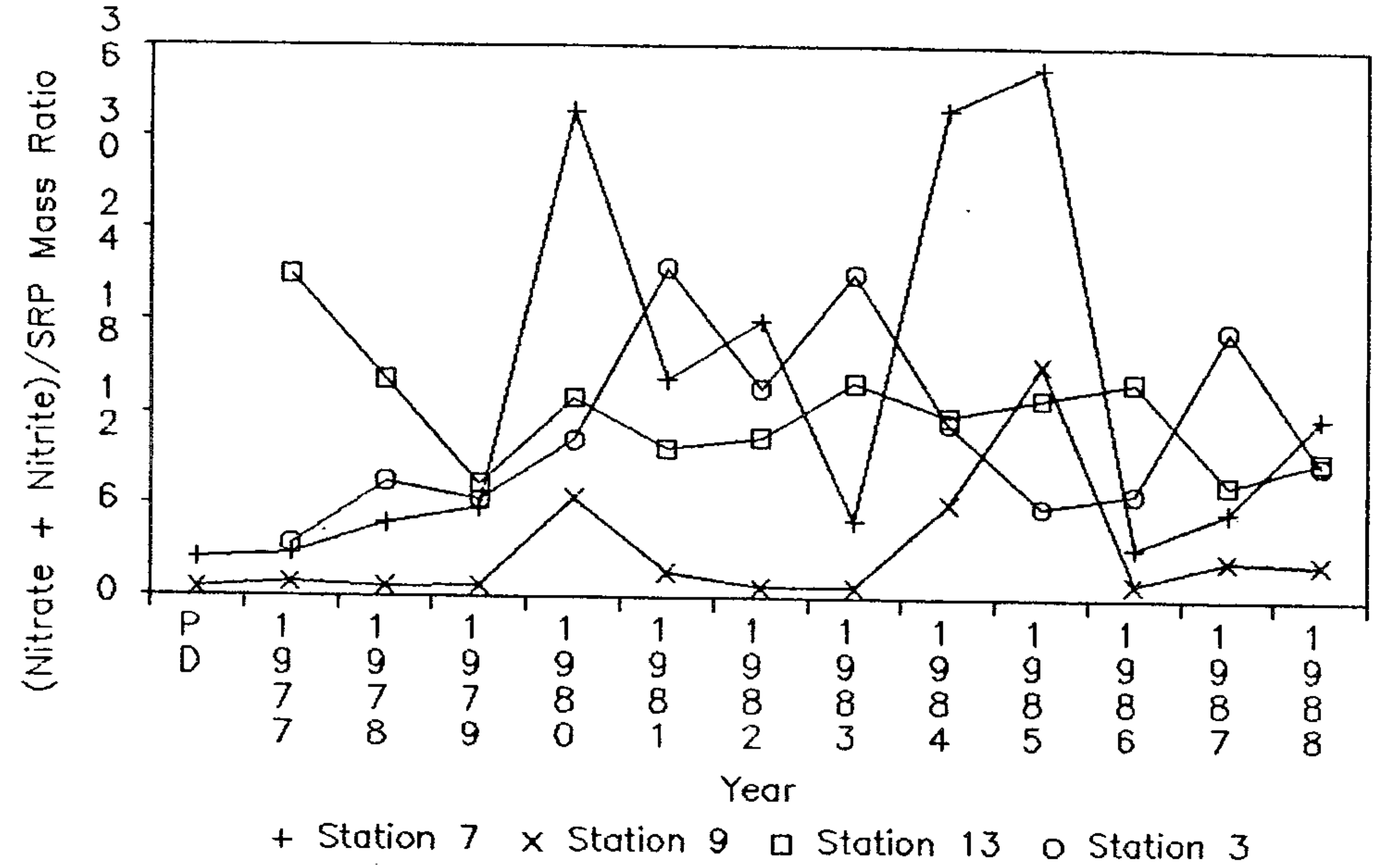


Figure 9. Soluble N:P ratios for Lower Parker Horn (7), South Lake (9), Rocky Ford Creek (13) and Crab Creek (3) for pre-dilution years (1969-1970; PD) and post-dilution years (1977-1988). *Values for stations 13 and 3 x 1/5*

ashfall and sewage diversion than to N:P ratios in the inflow. However, ratios in Crab Creek in 1986-1988 are not as high as following the ashfall. Nevertheless they are two to three times those in 1977-1979 (Figure 9 and Table 7). Although ratios in the lake during 1986-1988 had declined they were also higher than during 1977-1979. It seems clear that there has been a trend of increasing N:P ratios in the Crab Creek inflow and in the lake, due more to a reduction in P than an increase in N (Tables 6 and 7).

The TN:TP ratio shows a similar trend although its increase seems to be correlated more closely to sewage diversion. The mean TN:TP ratio in South Lake and Lower Parker Horn during 1977-1983 was 7.2 $\pm$ 1.0, while it was 10.9 $\pm$ 2.8 during 1984-1987. It decreased to 5.6 in 1988.

The trend of increasing N:P ratios in the lake has tended to cause a shift from N to P limitation of algal growth. In earlier post-dilution years, algal biomass was predictable from the inflow NO_3 concentration (Brenner, 1983; Welch et al., 1984). After 1980, however, that relationship no longer held (Welch and Weiher, 1987). As indicated above, the cause has been more related to decreasing P content in Crab Creek and to sewage diversion. The effect of the ashfall on sediment P release was shortlived. To illustrate the change to P limitation in the lake, observed chl <u>a</u> was compared with that predicted using the Jones and Backmann (1976) equation, which assumes P limitation (Table 8). During 1977-1983, the equation greatly overpredicted chl <u>a</u> at all stations. That indicates that N was largely controlling growth. A comparison between observed and predicted chl <u>a</u> for 1969-1985, using the Smith (1982) equation that includes TN, showed good agreement in South Lake (23 vs 21, respectively), but it underpredicted in Lower Parker Horn (34 vs 20). Since 1986, however, observed and predicted chl <u>a</u> concentrations using the Jones and Bachmann equation have been much more similar than in earlier years. N is still limiting in Pelican Horn, and some years in South Lake, but P appears to be consistently controlling growth in Lower Parker Horn.

<u>CO_2/pH</u>: The differences in CO_2 were much greater between sites and with time than with depth. As expected, CO_2 was higher in Pelican Horn (mean = 21.4 μM/L) than in Parker Horn (13.1 μM/L). Concentrations at both sites were highest on July 4, the day after strong winds gusting from 15 to 20 knots, and during heavy motor boat traffic, and lowest July 6-8 as calm conditions prevailed from the 4th to the 9th. The "critical" concentration of 7.5 μM/L, proposed by King (1970, 1972) to promote the dominance by blue-green algae, was surpassed on July 8 with values ranging from 7.9 μM/L to 6.7 μM/L (Figure 10).

A marked increase in the fraction of blue-green algae followed this decrease in CO_2 at both sites, especially at the surface in Parker Horn where a large mass of <u>Aphanizomenon</u> accumulated (Figures 11-12). Note that biomass was relatively low at both sites on July 6 when CO_2 had already declined, suggesting that increased dominance and buoyancy response observed on July 8 may have been an effect of rather than a cause for CO_2, while the CO_2 decline may have been more related to calm conditions (Figures 10 and 12). The surface accumulation of <u>Aphanizomenon</u> in Parker Horn, as well as the <u>Anabaena</u> bloom in Pelican Horn, markedly declined by July 10 following strong winds of 10-15 knots, gusting to 20 knots, which also probably caused the increase in CO_2, especially in Pelican Horn.

Table 8. Mean transect TP and corresponding predicted versus measured mean chl *a* for May through September at four representative stations (all values in µg/L).

		TP	Chlorophyll *a* PREDICTED[1]	Chlorophyll *a* OBSERVED
LOWER PARKER HORN (Sta. 7)	1977-83	67	38	23
	1985	116	84	72
	1986[2]	41	18	18
	1987	62	34	32
	1988	43	20	13
SOUTH LAKE (Sta. 9)	1977-83	85	53	18
	1985	88	56	41
	1986[2]	35	15	9
	1987	47	22	20
	1988	40	18	7
MIDDLE PELICAN HORN (Sta. 11)	1977-83	597	918	41
	1985	94	62	19
	1986[2]	81	50	14
	1987	68	39	11
	1988	84	52	11
ROCKY FORD ARM (Sta. 12)	1977-83	74	44	19
	1985	98	66	27
	1986[2]	49	24	23
	1987	50	25	20
	1988	61	33	15

[1]Chl *a* predicted by regression:
log chl *a* = 1.46 log TP - 1.09 (Jones & Bachmann, 1976).

[2]Data for 9/8/86 omitted because no chl *a* data for date.

NO_3 content was high at 4 m in Parker Horn reaching concentrations over 200 $\mu g/L$ on July 8 and 10, while it was barely detectable at the surface. That was expected with the very high accumulation of algae at the surface. The pattern was nearly the reverse in Pelican Horn, where levels at the surface went from near undetectable to near and over 100 $\mu g/L$ on those dates, but reached undetectable levels at 4 m. At some point in the water column there was much more NO_3 available on July 8 at both sites than previously.

Zooplankton

The crustacean zooplankton are very abundent in Moses Lake with Daphnia pulicaria being the dominant species (Carey, 1981). D. pulicaria represented 60% of the total biomass of crustacean zooplankton during the spring-summer periods of 1978 and 1979, and was the largest of the crustaceans with more than 10% of its population exceeding 2 mm in length. There are seven principle species of Daphnia and five copepod species in the lake (Carey, 1981).

May to September means for Daphnia and total crustacean zooplankton are given in Table 9 (Appendix B) for 1978-1979, 1983 and 1985-1988. Data in No/L for 1984 are not available because depths of 0.5 m net-hauls were inconsistently recorded. Daphnia were less abundant in Lower Parker Horn during 1985 when dilution water input continued throughout the summer. Yet abundance was relatively high in 1979 when summer dilution input rates were higher than in 1985 (see Table 4). Abundance was very low during April, 1988 (3/L) when flushing rate was the highest yet for any period (38%/day), but averaged 39/L during May to mid-June when flushing rate was 18%/day (Table 4). Carey (1981) observed a slight negative effect on abundance in Lower Parker Horn when flushing rate was only 14%/day. Flushing rates that high could significantly lower the abundance of zooplankton because the observed growth rate of D. pulicaria was only 23%/day. Abundance of Daphnia was also low in 1987, but dilution flows were low that year (Table 4).

Zooplankton were much more abundant in Middle Pelican Horn than in other areas before diversion (Table 9). The algal community is composed of more unicellular green algae and less filamentous and colonial blue greens in Pelican Horn than in other parts of the lake (see Figure 7). Thus, the greater pre-diversion abundance of zooplankton, particularly the smaller species, may have been a response to more algae of edible size. Since diversion, especially in 1987-1988, zooplankton abundance and composition have been similar to that in other lake areas.

Water and Phosphorus Budgets

A summary of the May-September water and phosphorus budgets for 1977-1988 is shown in Table 10. The May-September period was emphasized because that is when dilution and irrigation return flows occur, as well as being the period of interest with respect to lake quality. As will be seen in the modeling section, all segments except Rocky Ford Arm would be expected to have TP reach 95% of equilibrium with inputs during that period.

Table 9. May-September average abundance throughout the water column of all species of *Daphnia* and total crustacean zooplankton in No./L.

		Lower Parker Horn (7)	South Lake* (9)	Middle Pelican (11)	Rocky Ford Arm (12)
1978	*Daphnia*	35	31	--	44
1979	*Daphnia*	30	22	--	36
1983	*Daphnia*	38	31	88	56
	Total	92	96	644	126
1985	*Daphnia*	12	24	53	31
	Total	63	71	372	85
1986	*Daphnia*	20	31	145	32
	Total	100	149	509	124
1987	*Daphnia*	12	15	11	29
	Total	77	110	135	77
1988	*Daphnia*	38	31	48	28
	Total	100	91	125	80

*surface to 6 m

The flow in Rocky Ford Creek was rather constant, while Crab Creek (plus East Low Canal and Rocky Coulee Wasteway) flow was highly variable, due largely to year-to-year fluctuations in irrigation water use and dilution water addition. Ground water was also rather constant averaging 29 ± 9% of the total input for the 12 treatment years.

TP loading has been very similar in the two surface water inflows over the past 12 years (Table 10). Ground water averaged 15 ± 6% of the total input before sewage diversion and 30 ± 7% of the total after diversion. Sewage contributed an average of 27 ± 6% of the total external load.

Net internal loading is significant in Moses Lake, averaging 33 ± 15% of the total (including internal) for the 10 years when that source was positive (Table 11). There was no net internal loading in 1980, due to the ashlayer that blocked release of P for sediments (Welch et al., 1985). There is no explanation for the lack of internal loading occuring in 1978. The largest contribution from internal loading occurred in 1985 when it exceeded the external loading (Table 11). That was also the year of the largest algal blooms (Table 5).

The contribution from ground water becomes less significant if internal loading is included. Thus, ground water averaged only 13 ± 4% of the total before sewage diversion and 19 ± 6% after diversion.

The large internal load in 1985 was probably caused by increased wind mixing of the lake water column during August and September. Cool, windy weather promoted increased mixing, which in turn apparently caused increased transport of P accross the sediment-water interface. The increased surface TP was not simply a redistribution of TP already in the water column, because water column TPs during June-July, compared to August-September, were 58 versus 154 µg/L in South Lake and 45 versus 111 µg/L in Lower Parker Horn. The process (σ) that accounted for that high internal loading will be discussed later.

To determine if the stratification/mixing conditions influenced internal loading in the lake generally, RTR was calculated for the water columns at Stations 7, 9, 10 and 12 for each sampling day during May-September, with the average determined by dividing each season's total by the number of sampling days (Table 10). Stations 5 and 11 are shallow and consistently mixed so were not included in the calculations. A volume-weighted mean for the whole lake was determined from the Station means (Table 12). Flushing rate was also considered important so ρ and RTR were both related to internal loading for the period 1980-1984 (Jones, 1988):

$$J_{int} = 2.92\ \rho - 0.30\ \text{vwRTR} + 19.8 \quad (r^2 = 0.78)$$

where J_{int} is net internal TP loading in metric tons.

The above equation was then tested with 1985-1988 data to determine how well those two variables accounted for internal TP loading generally (Figure 13). Internal TP loading was underpredicted in 1987, otherwise these two factors (RTR and ρ) account for most of the year-to-year variation in that source of TP. A much stronger relationship exists if 1980-1981 data following the ashfall are eliminated and recent data (1982-1987) are used:

Table 12. May through September relative thermal resistence (RTR) for 1980 to 1988.

YEAR	STATION 7	STATION 9	STATION 10	STATION 12	VOLUME WEIGHTED
1980	62.2	116.9	84.4	66.5	76.0
1981	65.3	109.7		69.9	77.2
1982	64.4	135.8	60.0	84.4	91.4
1983	35.7	74.4	29.0	42.2	47.6
1984	42.0	110.8	55.7	79.4	79.5
1985	38.4	72.7	50.8	57.3	57.2
1986	56.3	172.3	51.6	68.8	87.7
1987	89.2	113.4		75.2	85.3
1988					60.9
VOLUME WEIGHT(%)	17	20		63	

Station 10 was not included in the volume-weighted RTR (vwRTR) because it was not sampled during two of the study years. It represented less than 6% of the lake volume.

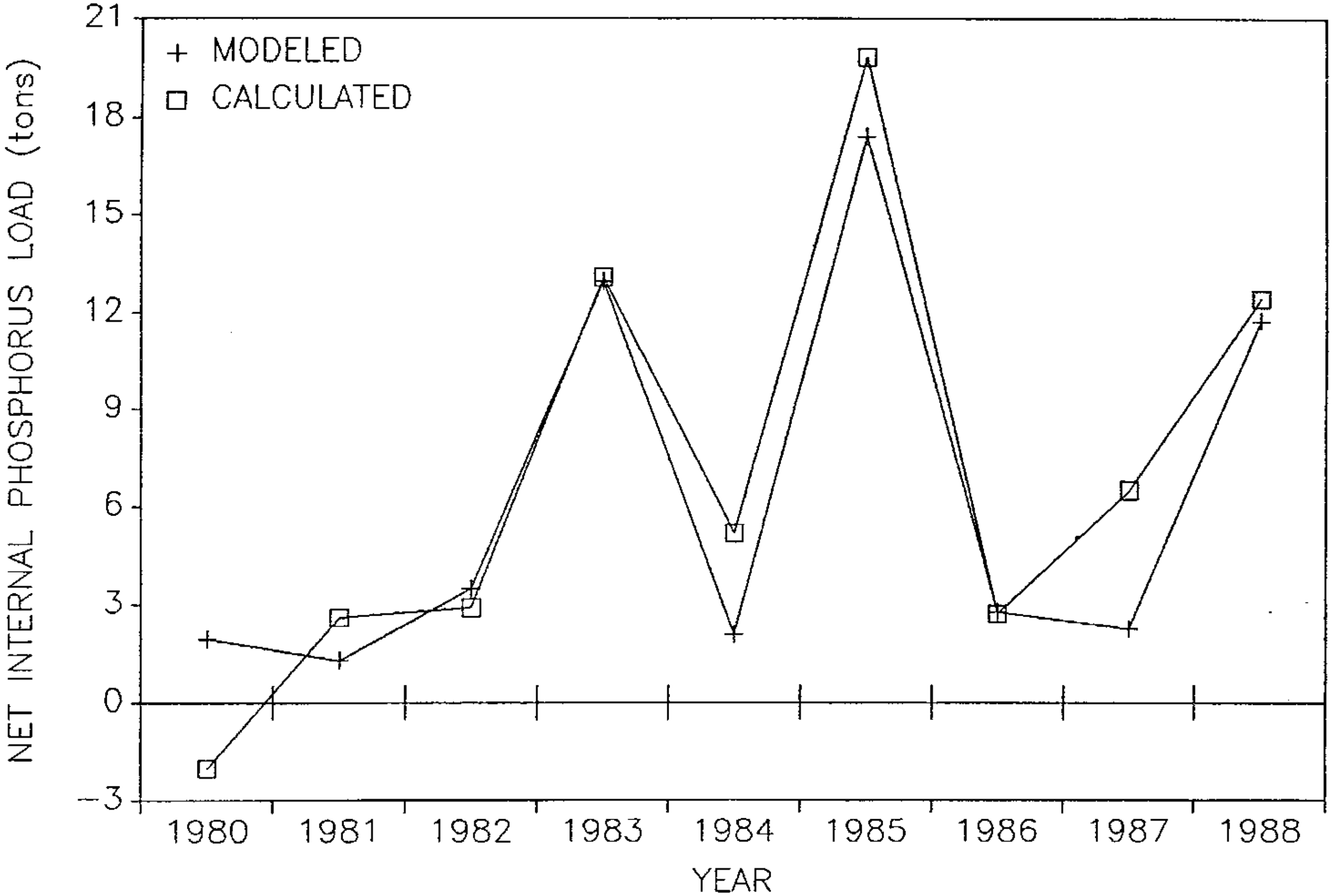


Figure 13. Calibration and verification of net internal TP loading model for Moses Lake using May through September totals. Calibrated with 1980-1984 data and verified with 1985-1988 data.

$$J_{int} = 2.63\ \rho - 0.30\ \text{vwRTR} + 22.32 \quad (r^2 = 0.92)$$

Mount St. Helens' ashlayer inhibited P transport from the sediment (note overprediction in 1980, Figure 13). Sinking of the ashlayer by 1982 should have greatly reduced that effect (Welch et al., 1985). Using the 1982-1987 equation to predict internal loading for 1980 and 1981 gave values of 3.9 and 3.3 metric tons, which were, respectively, 6.0 and 0.6 tons greater than the observed loadings. Thus, the ashlayer had its greatest effect in 1980 and has been negligible since.

The effect of RTR on internal loading was considerably more important than that of ρ. Using the mean RTR and mean ρ for 1980-1987, showed that RTR had a 2.8 times greater effect on internal loading than did ρ. A ten percent difference in RTR or ρ from their eight-year means would change the prediced net internal TP loading by 2.2 and 0.8 metric tons, respectively. A ten percent change from their eight-year maximums would alter the internal loading by 2.7 and 1.5 tons.

Phosphorus Model Predictions

Modelling Process: The column weighted TP concentrations in Lower Parker Horn, Main Arm (actually Cascade) and Lower Pelican Horn were predicted first in order to estimate the TP entering the South Lake. The transect [TP] in both the South Lake and Lower Pelican Horn were then predicted for the purpose of predicting chl *a* and transparency with different management scenarios. However, only Parker Horn transect TP will be included (see Jones, 1988). Each [TP] prediction was determined from the following procedure:

1) Calculate σ from the following rearrangement of the steady-state TP:

$$\sigma = J_{ext}/(V[TP]) - \rho \qquad (3)$$

2) Regress ρ and RTR (Table 12) on σ using 1980 through 1984 data for calibration. J_{ext} in Table 10 was corrected to an annual rate (x 12/5) to conform to units for ρ and σ.

3) Predict TP from the steady-state model using the σ predicted above. Verify the results with 1985 and 1986 data for Cascade and Lower Pelican Horn and also with 1985-1988 data for Lower Parker Horn.

Column-Weighted TP Models: Table 13 illustrates the large differences in σ and ρ among lake areas. The Main Arm, which includes middle Rocky Ford Arm and the Cascade region, behaves like many deep lakes in that much of the incoming TP settles out. This results in the relatively large sedimentation coefficients in the Main Arm. Pelican Horn also had positive sedimentation coefficients until low nutrient Upper Parker Horn water began in 1982. Since then, σs were negative. If internal TP loading were increased from adding low-TP water, thereby making the increases in ρ and $J_{ext}/[V(TP)]$ less proportional, then σs would become negative. This appears to be the case in Lower Parker Horn as well. Note the smallest (and most negative) σs occurred in 1985 in both Pelican and Parker Horns, when internal loading was greatest (Table 11).

Table 13. Observed and calculated data[1] for predicting TP in Parker Horn, the Main Arm, and Pelican Horn during May through September. Flushing and sedimentation rates were corrected to annual values (x 12/5).

	YEAR	TP(μg/l)	J_{ext}(kg)	ρ(yr^{-1})	σ(yr^{-1})
	1980	85.0	4526	13.8	-3.4
	1981	72.0	3049	11.3	-3.0
	1982	51.2	6202	37.4	-13.8
PARKER[2] HORN	1983	99.9	5811	20.1	-8.7
	1984	75.8	2713	8.1	-1.1
	1985	59.1	6200	42.1	-21.6
	1986	67.9	3800	21.4	-10.5
	1987	73.4	3000	19.2	-8.6
	1988	54.9	4693	27.8	-11.2
	1980	87.5	5390	0.48	1.02
	1981	80.0	3586	0.32	0.78
	1982	70.7	4439	0.39	1.15
MAIN[3] ARM	1983	101.3	4937	0.50	0.70
	1984	64.8	6816	0.68	1.89
	1985	87.6	3810	0.44	0.63
	1986	68.3	5065	0.55	1.27
	1987		3604	0.39	
	1977	107.5	5491	6.8	5.7
	1978	106.5	6211	5.6	8.6
	1979	119.5	8687	16.6	1.1
	1980	137.7	5478	5.6	4.1
PELICAN[4] HORN	1981		5750	10.7	
	1982	109.4	6102	14.2	-0.6
	1983	138.1	6500	14.4	-2.9
	1984	89.8	4590	19.4	-6.9
	1985	85.6	5160	25.2	-10.5
	1986	56.8	3840	20.0	-3.5
	1987		3350	17.8	

[1]Assumes 80% of groundwater flow enters Pelican Horn and 10% enters each the Main Arm and Parker Horn.
[2]Lower Parker Horn (Sta. 7) column-weighted mean.
[3]Cascade (Sta. 8) composite.
[4]Lower Pelican Horn (Sta. 10) column-weighted mean.

The sedimentation coefficients in Table 13 were determined from the following linear regression equations:

For the Main Arm:

$$\sigma = 3.02\ \rho + 0.02\ \mathrm{RTR} - 1.68$$

For Parker Horn:

$$\sigma = -0.46\ \rho + 0.07\ \mathrm{RTR} - 2.62$$

For Pelican Horn:

$$\sigma = -0.79\ \rho + 0.04\ \mathrm{RTR} + 5.68$$

As expected, RTR was found to be positively correlated with σ in all three arms. Flushing rate in the slow moving Main Arm was also positively related to σ which is the situation in most lakes (Jones and Bachmann, 1976). However, the two rates (σ and ρ) were inversely related in Parker Horn and Pelican Horn. The probable reason for this was indicated above. Average values from 1980 through 1987 were equally sensitive to changes in either ρ or RTR in the Main Arm, but were more sensitive to ρ in Parker Horn and Pelican Horn by factors of 2 and 1.2, respectively.

The column-weighted TP model for Lower Parker Horn calibrated precisely and predictions were within the confidence range for observed TP for two of the four verification years (Figure 14). The model for Lower Pelican Horn also calibrated well and was accurate in predictions for 1985-1986 (Figure 15). The Main Arm model calibrated rather easily and showed acceptable agreement with observed values during 1985-1986, the verification years (Figure 16). Again, this may have been due to the more expected behavior of σ with respect to ρ in the Main Arm. Cascade (8) and Lower Pelican Horn (10) were not sampled during 1987-1988, so those years could not be verified.

Transect TP Model: Epilimnetic TP concentrations in Lower Parker Horn were modelled in hopes of predicting chl a in transect samples. The linear regression of both RTR and o on s determined that:

$$\sigma = -0.45\ \rho - 0.11\ \mathrm{RTR} + 11.77$$

This indicates that RTR was negatively related to σ, instead of positively related, as was the case when the entire water column was considered.

Although the predicted TP for the calibration years again agreed well with observed TP in Lower Parker Horn ($r^2 = 0.98$), the TP in 1985 was underpredicted by a factor of 3 (Figure 17). Nevertheless, predictions for 1986-1988 (although 1988 not shown) were acceptably close to confidence intervals for observed TP.

Steady-State: The assumption of steady-state was tested for each arm by using the following relationship:

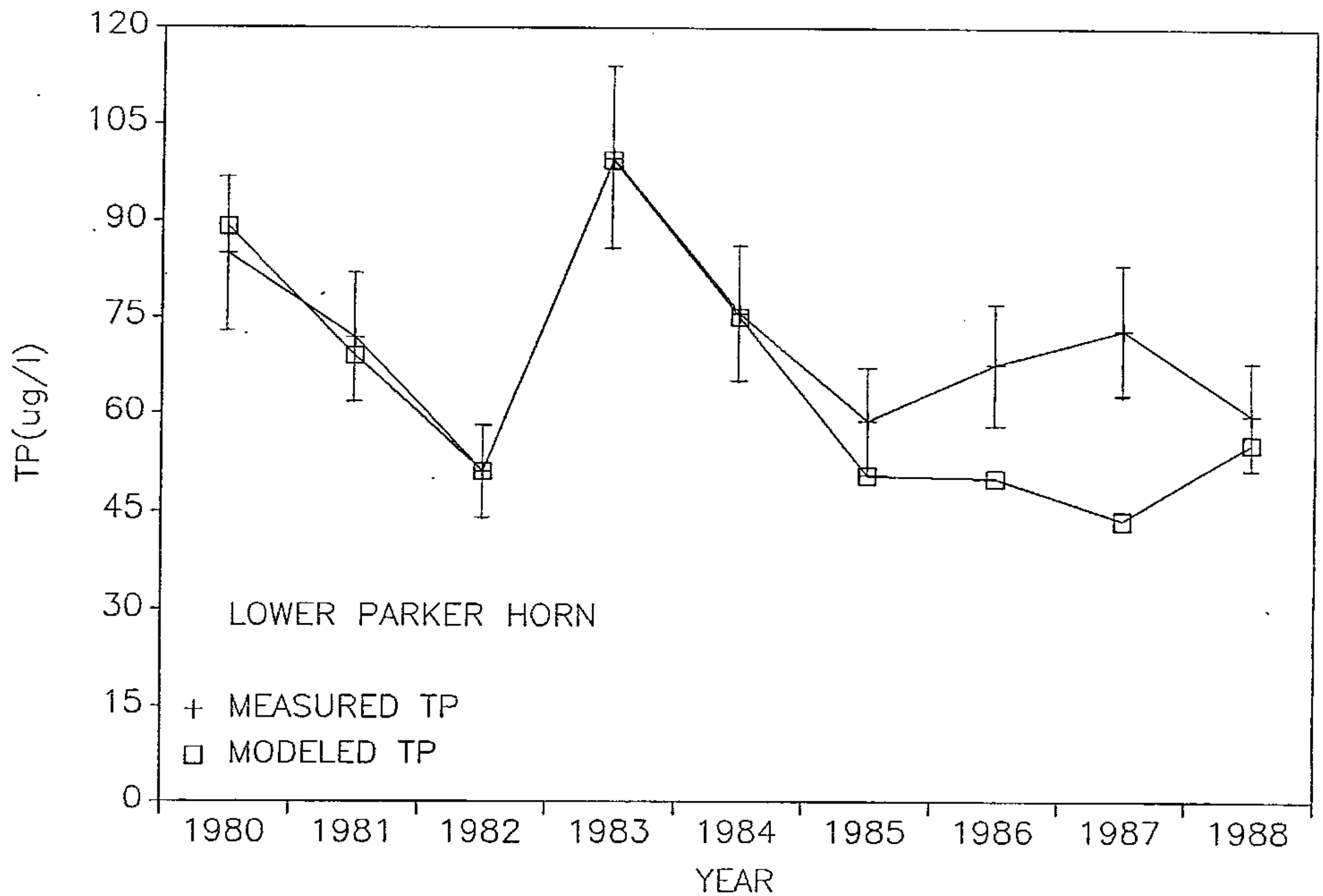
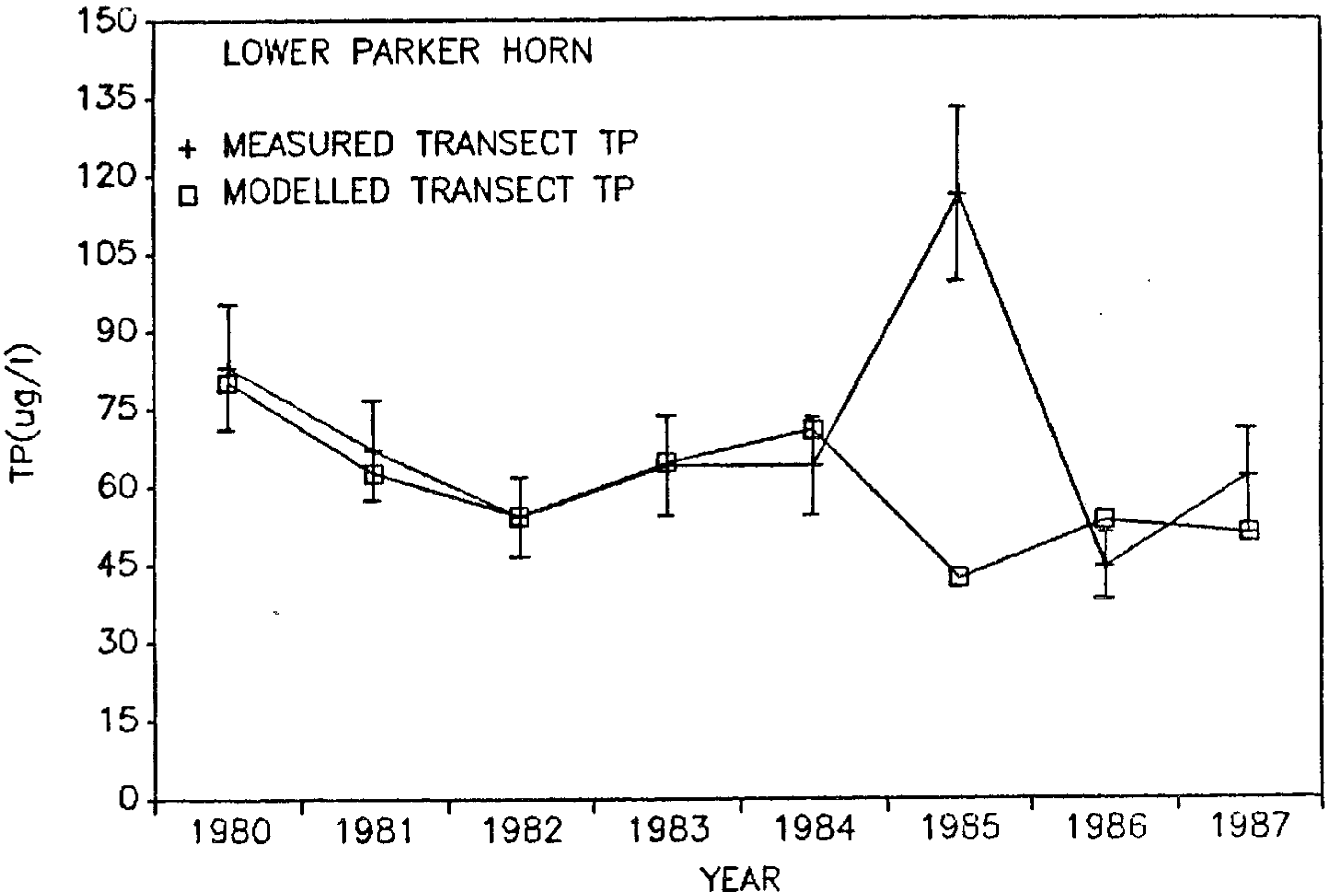


Figure 14. Calibration and verification of column weighted mean TP model at Lower Parker Horn (Station 7) for May through September period. Calibrated with 1980-1984 data and verified with 1985-1988 data. Error bars represent estimated 95% confidence intervals for the measured TP.

Figure 17. Calibration and verification of transect TP model at Lower Parker Horn (Sta. 7) for May through September period. Calibrated with 1980-1984 data and verified with 1985-1987 data. Error bars represent estimated 95% confidence intervals for the measured TP.



$$F = 1 - \exp[t(\sigma+\rho)]$$

where F = fraction of steady-state reached

This relationship is derived from the original mass balance equation for TP. Using 1980 through 1987 means, this approach showed that all but the Main Arm should have reached at least 95% of steady-state during the May through September period. The Main Arm was expected to reach only 50% of steady-state.

Discussion

The quality of Moses Lake has changed dramatically since an annual pattern of dilution water inputs started in 1977. Dilution water was distributed to Pelican Horn in 1982 and sewage was diverted from that part of the lake in 1984. There has also been a trend of decreasing P concentrations in Crab Creek, the inflow to Parker Horn, apparently related to the increasing use of spray irrigation and possibly to the Mount St. Helens' ashfall. BMPs in effect in the Crab Creek drainage since 1987 have apparently not affected the Creek's N content. There has also been no change in the nutrient content in Rocky Ford Creek, resutling from a retention pond constructed in 1985.

The first three above mentioned, thus far successful, controls on nutrients have all contributed, unfortunately in unknown proportions, to the improvement in lake quality, which in 1988 was at the highest level ever observed. Although the lake is still eutrophic, as indicated by the overall means during 1986-1988 for TP, chl a and transparency in Lower Parker Horn and South Lake of 45 mg/L, 17 mg/L and 1.6 m, respectively, it is no longer hypereutrophic as it was before treatment began in 1969-1970 (TP, chl a and transparency of, respectively, 154 mg/L, 57 mg/L and 0.8 m).

As an indication of the relative importance of dilution water and diversion, no dilution water was added in 1984, the spring that sewage was diverted. TP and chl a surprisingly did not indicate poorer quality in Lower Parker Horn that year (although transparency was less), but showed actual improvements in South Lake as a direct result of diversion. Parker Horn had remained unchanged either because it had previously been affected by wind-blown, sewage-enriched water from South Lake (with diversion offsetting the lack of dilution), or because the lowered P concentrations in Crab Creek were then controlling algal growth, in spite of dilution. Dilution has always affected inflow NO_3 (ratio Crab Creek:dilution water of 20:1) more than TP (ratio of 3.8). That ratio has not changed with respect to NO_3, but has declined to 2.4 for TP (Table 7). Thus, since the N:P ratio has increased in the lake and Crab Creek inflow, and P has become the more important limiting nutrient in recent years, dilution may be having less effect. That is also suggested by the TP model (see below).

Now that the lake no longer receives sewage effluent with its high P content (relative to N), and has become P limited, the principal control on spring-summer lake quality may be internal loading of P from sediments, controlled largely by the wind. Internal loading has averaged one third of the total load of TP to the lake during the ten of twelve years that net internal loading occurred, and it has become relatively more important since sewage effluent was diverted. In fact, it was greater than external loading in 1985 (and 1988). The high internal loading in 1985 was inversely related to RTR, meaning that when thermal stratification declined during the summer, primarily through wind-driven mixing of the water column, transport of P from sediments increased. Moreover, vwRTR ranged from 47 to 91 during 1980-1988 resulting in an internal loading difference of 13,000 kg TP, which is larger than external loading in 1986, 1987 and 1988. Overall, internal loading varied by nearly 100% of the mean, while external loading varied by less than 20%. Although RTR is a function of temperature as well as wind, the lack of air temperature differences between years of widely differing RTRs indicates that wind is the primary determinant. Also, Marquis (1985) simulated water-column stability in the lake as a function of wind speed.

There may be several processes involved to account for increased internal loading from increased wind mixing (low RTR). Particles containing P and sediment porewater rich in P may be entrained with mixing (Ahlgren, 1980). However, particles effectively sorb P, unless pH is high, and remove it from the water column through settling (Holdren and Armstrong, 1980), so entrainment of water with high soluble P may be more important. Also, mixing of low-P surface water to the sediment surface would increase the gradient across the sediment-water interface and increase the rate of diffusion. Mixing high-pH surface water to the sediment surface would also enhance the release of sediment P (Bostrom et al., 1982). Also, temperature at the sediment-water interface would be raised, increasing microbial activity and remineralization. While mixing restores or maintains aerobic conditions at the sediment-water interface, allowing iron to bind phosphate, aerobic release does occur in Moses Lake (Okereke, 1987). Although the aerobic release rate is much less than the anaerobic rate, the aerobic lake area is greater than the anaerobic area.

Internal loading reduces the effectiveness of dilution in controlling lake quality, especially now that P is limiting. The expected range in lake TP was calculated in response to dilution water addition using the model for Lower Parker Horn and the nine year range in vwRTR (Figure 18). The predicted limits for TP demonstrate the significance of wind-induced internal loading, especially when dilution addition is minimal. For example, TP in 1984, the no-dilution year, was 76 ug/L, but could have been much higher had RTR been near the nine-year minimum. Also, TP in 1985 would have probably been much higher if dilution-water input had not been so great (192 X 10^6 m^3). However, the gradual slope of the curve in Figure 14 illustrates the moderating influence of dilution water in Moses Lake because of the magnitude of internal loading. Without such high internal loading, there would be much greater benefits (lower lake TP) from such large amounts of dilution water, containing such low TP (16 μg/L) added to a relatively low Crab Creek inflow (28 X 10^6 m^3). The higher observed TP than the range of predictions during some years is because average flow and loading in Crab Creek were used.

The model indicates that internal TP loading was increased by increasing ρ, which is effectively increasing the dilution water addition, although the effect of increasing ρ was considerably less than that of decreasing RTR. An increase in sediment P release from increased flushing has, to our knowledge, been documented only once before, although it has been suggested as a logical possibility (Goldman, 1968). Using microcosms containing sediments from Narragansett Bay, Rhode Island, and ten liters of water, Poon (1977) found that TP was higher than expected in the microcosm outflow if flushing were

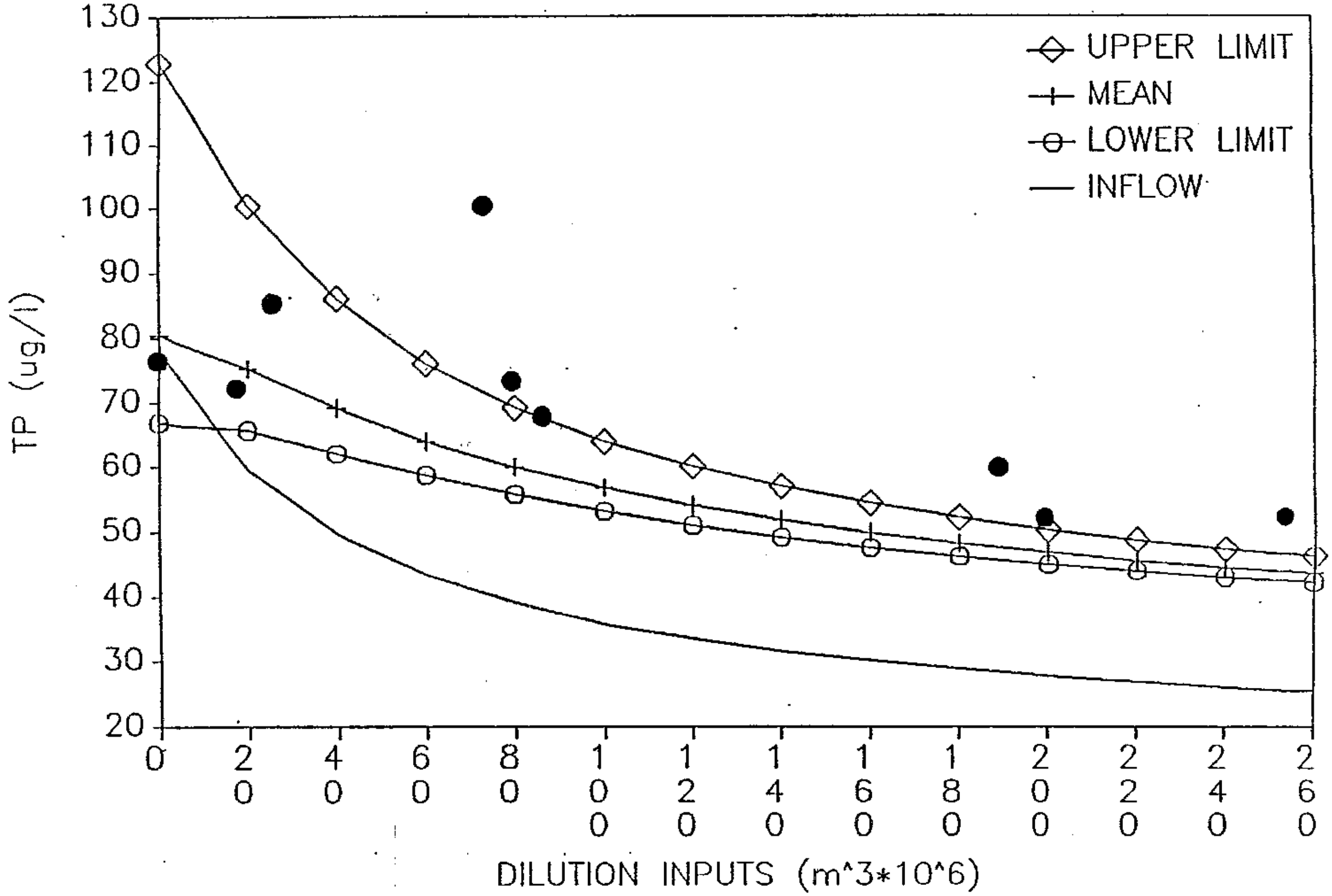


Figure 18. Predicted and observed (solid circles) column-weighted mean TP as a function of dilution water input in Lower Parker Horn (7) during May-September for the nine-year average TP loading, flushing rate and range of RTR values observed.

increased. When ρ was increased from 2.5%/day to 10%/day, steady-state P flux out of the sediment increased by 33%. The cause was attributed to high flushing maintaining lower sediment-water P concentrations and therefore higher diffusive flux of P out of the sediment. In Moses Lake, higher May-September flushing rates were due to higher inputs of dilution water with low P content, which would have resulted in higher P gradients between sediment and water and thereby higher diffusive flux rates of P. For example, the May-September mean whole-lake inflow concentrations of TP were 87 and 41 μg/L for 1984 and 1985 when flushing rates were 0.55 and 1.4%/day, respectively.

Because net internal loading is calculated by difference in the mass balance, increased net internal loading could also be caused by decreased sedimentation in response to increased flushing rate (shorter water residence time). However, the increased net internal loading that resulted from increased dilution water input in Moses Lake could not have been due simply to decreased water detention time and decreased sedimentation. The lower curve in Figure 18 is the calculated inflow TP concentration to Parker Horn as a function of dilution. That represents the maximum TP concentration possible in Parker Horn (nine-year mean) with no sedimentation. Therefore, decreased sedimentation due to decreased detention time could not have accounted for the much higher observed and predicted TP in the lake.

Algal composition had apparently shifted away from a nearly exclusive dominance by bloom forming blue green algae (e.g. Aphanizomeon and Microcystis) during the first four summers following the start of dilution. Patmont (1980) suggested that higher observed CO_2 concentrations resulting from reduced productivity were a likely cause for that shift away from blue greens. However, the shift did not persist. Although algal abundance is much less in recent years than before, or even during the earlier years following dilution, the percent composition by blue greens is high (75-90%). Furthermore, the composition in Pelican Horn shifted from exclusively green algae before diversion to a sizable fraction of blue greens the first two years following diversion. But that shift has not persisted either and blue greens there are again of minor importance. Bush (1972) had suggested that the dominance of green algae in Pelican Horn was due to its high total carbon content (therefore high CO_2).

A prominent hypothesis to explain the blue green dominance, or lack thereof, involves the content of free CO_2 (King, 1970, 1972; Shapiro, 1973, 1984). High productivity that raises pH and lowers free CO_2 apparently favors blue greens over greens. The effect is even more pronounced in low alkalinity than high alkalinity waters because for a given pH increase, CO_2 would reach lower concentrations where alkalinity is lower. Low CO_2 may also promote buoyancy in blue greens assisting their dominance.

The experiments conducted in Lower Pelican and Parker Horns in 1986 tended to verify the CO_2 hypothesis. Blue greens increased markedly and some to the surface in Parker Horn after CO_2 had declined to less than King's suggested critical level (7.5 μM/L). Blue greens were not as dominant and did not rise to the surface in Pelican Horn where CO_2 did not reach such low levels. Apparently, enrichment and algal productivity have not yet reached a low enough level in Parker Horn and South Lake to raise CO_2 sufficiently above the critical level that promotes blue green dominance. And CO_2 may have remained high enough in Pelican Horn to continue to favor green algae.

Further improvements in Moses Lake quality should be directed at continued efforts to reduce P loading through best management practices in the watershed or through in-lake control of sediment P release. Now that P is limiting algal growth in most of the lake, except Pelican Horn, effort should be directed toward P control instead of N. Unfortunately, dilution water has become less effective since P has become limiting (see explanation above). Currently, the use of BMPs is too low to have been effective in reducing N (or P) in the inflow to the lake. Originally, the decrease in N in the undiluted Parker Horn inflow that could be expected from BMPs was estimated at 25%. Such a decrease could switch limitation back to N and further reduce algal biomass, even now when P is usually limiting, although P might be expected to decline as well with BMPs. However, only one fifth of the landowners initially undertook BMPs, which reduced the expected N decrease to 5% and with dilution water addition would amount to less than a 2% decrease in Parker Horn inflow N. Thus, it is not surprising that a nutrient concentration change attributed to BMPs has not yet been observed. With greater compliance in the future water quality improvements from current conditions are still possible.

The detention basin constructed on Rocky Ford Creek was expected to reduce TP by 37% (Bain et al., 1985). Using the TP model for the Main Arm, TP should decrease by 35%, from 55 μg/L in 1985-1986 to 36 ug/L, which should in turn reduce chl a by 46%, from 28 to 15 μg/L. The expected effect of the Rocky Ford Creek detention pond in South Lake would be only a 4% reduction in TP, because the Main Arm contributes only 12% of the TP load to South Lake. Although a decrease in TP loading from Rocky Ford Creek of 22% was observed since 1985, the cause was due more to a decrease in water flow than to a decrease in TP concentration. Moreover, there was no observed change in Rocky Ford Creek TP downstream of the detention basin, compared to upstream, during 1987-1988. P may have been difficult to remove in the detention basin because such a large fraction of Rocky Ford Creek TP is soluble (65 $\pm$ 18%, n=21, 1987-1988). The success of the detention basin in removing nutrients was based on carp being eliminated upstream from the dam and the development of aquatic plants to absorb the soluble nutrients. Carp were removed in 1988 and dense stands of plants did develop upstream in 1989, thus an assessment of the nutrient removal capability of the pond/wetland using the 1986-1988 data is premature (Bain, personal communication).

Although the effect of dilution water addition on nutrient reduction and algal control is diminished because dilution increases internal loading of P, the lake would experience much poorer quality without continued dilution. As shown in Figure 18, the range in TP due to wind mixing is minimized by dilution water addition up to 80 - 100 X 10^6 m^3. Furthermore, dilution water improves water clarity more than would be expected from the resulting decrease in algae. This is because of the relatively high concentration of non-algal particulate matter in the lake. However, because internal loading is increased by dilution water addition, it may be advisable to optimize dilution water input to: 1) minimize internal loading, 2) maximize clarity and 3) minimize lake TP content. An input of about 100 X 10^6 m^3, distributed throughout the summer, would appear to represent such an optimum.

Summary and Conclusions

1. The restoration measurers of dilution and diversion to improve the quality of Moses Lake have been highly successful. The trophic state indicators, total phosphorus, chlorophyll a and Secchi disc transparency have improved markedly, especially in recent years (1986-1988). Compared to pre-treatment years (1969-1970), phosphorus and chlorophyll have decreased on the average by 70% and transparency has increased by 200% in Lower Parker Horn and South Lake. Transparency has increased more than could be expected from the decrease in algae (chl a), due to the high clarity of Columbia River dilution water. Increased clarity has promoted the development of dense stands of rooted macrophytes (P. crispus and pectinatus) in Upper Parker Horn. The lake is still eutrophic, but not hypereutrophic as before treatment.

2. The degree of improvement was even greater in Pelican Horn, especially following diversion. Although algal concentrations have declined to levels similar to other segments of the lake, total phosphorus content (77 μg/L) and transparency (0.8 m) indicate that a hypereutrophic state still exists. Pelican Horn's shallowness, and susceptability to the wind, and the dense carp population, may preclude further improvements in quality.

3. A change to fewer blue-green algae, that occurred in the first couple years after dilution began (30% in South Lake), has not continued and the average blue-green fraction now ranges from 75-90% in Lower Parker Horn and South Lake. The increase in blue-greens in Middle Pelican Horn to 50% in 1984, immediately after diversion, did not persist as well. In fact, the blue-green fraction is now only 5% in Middle Pelican Horn. Experimental evidence was developed to substantiate the hypothesis that buoyancy and dominance of blue-green algae is affected by the concentration of free carbon dioxide.

4. Dilution had more effect in controlling algal growth in earlier years (1977-1979) when nitrogen was limiting. Diversion of sewage effluent from Pelican Horn in 1984 and a trend of decreasing phosphorus content in Crab Creek, due probably to the increase in spray irrigation, as well as the ashfall, have led to increasing N:P ratios in the inflow and in the lake and a switch to phosphorus limitation. Dilution was more effective at reducing algae when nitrogen was limiting due to the more favorable ratio between East Low Canal water and Crab Creek for nitrogen than for phosphorus. The recent reduced effect of dilution was apparent in 1984 when little change in nutrients or algae was observed in spite of no dilution that year.

5. Internal loading of phosphorus, largely through release from bottom sediments, has become increasingly important since diversion and the decrease in Crab Creek phosphorus content. Internal loading has exceeded external loading two of the last four years. The year-to-year variation in internal loading was nearly 100% of the mean and was controlled largely by wind mixing, and to a lesser extent by flushing, as indicated by a strong correlation between loading calculated by mass balance and relative thermal resistance to mixing (RTR) and flushing rate (ρ). The positive relation between flushing and internal loading probably resulted

from reduced concentration of phosphorus due to dilution and the consequent larger diffusive gradient between sediment and water.

6. Steady-state models to predict total phosphorus, including internal loading and the factors controlling it, were developed for three segments in the lake; Lower Parker Horn, Rocky Ford Arm and Lower Pelican Horn. There was good agreement between observed and predicted total phosphorus in the Lower Parker Horn water column, except during 1986 and 1987. However, except for 1985 (the year of large surface accumulations), agreement was good for transect concentrations, so that models could be used for reliable future predictions of the combined effect of dilution and further decreases in phosphorus loading in Crab Creek, using the range in RTR observed over the last nine years as a measure of uncertainty. The Jones and Bachmann equation is reliable to predict average chl a.

7. The planned Best Management Practices, including the detention basin constructed on Rocky Ford Creek, had not yet contributed to the recorded improvement in Moses Lake quality by 1988. This is largely because only 20% of the anticipated land area had thus far been included for irrigation water and fertilizer use controls. Theoretically, changes in nutrient concentrations from that small a land area would probably not be detectable in the lake. The decrease in Crab Creek phosphorus content over the past nine years, noted in no. 3 above, is probably partly related to the gradual shift to spray irrigation and, thus, improved irrigation techniques. Further reduction in lake nutrient content may occur, however, if a greater BMP compliance is attained. While the detention basin on Rocky Ford Creek had not shown a detectable removal of nutrients through 1988, carp were removed upstream from the dam in 1988 and aquatic plants developed extensively during 1989. Therefore, nutrient removal may be expected from that facility in the future.

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