MAY 1996

SNAKE RIVER SOCKEYE SALMON HABITAT AND LIMNOLOGICAL RESEARCH

Annual Report 1995

BONNEVILLE
POWER ADMINISTRATION

This report was funded by the Bonneville Power Administration (BPA), U.S. Department of Energy, as part of BPA's program to protect, mitigate, and enhance fish and wildlife affected by the development and operation of hydroelectric facilities on the Columbia River and its tributaries. The views in this report are the author's and do not necessarily represent the views of BPA.

For additional copies of this report, write to:

Bonneville Power Administration
Public Information Center - CKPS-1
P.O. Box 3621
Portland, OR 97208

Please include title, author, and DOE/BP number from the back cover in the request.

SNAKE RIVER SOCKEYE SALMON HABITAT AND LIMNOLOGICAL RESEARCH

ANNUAL REPORT 1995

Prepared by:

David Teuscher
Doug Taki

Shoshone-Bannock Tribes
Fisheries Department

Prepared for:

U. S. Department of Energy
Bonneville Power Administration
Environment, Fish and Wildlife
P.O. Box 3621
Portland, OR 97208-3621

Project Number 91-71
Contract Number DE-BI79-91BP22548

May 1996

TABLE OF CONTENTS

CHAPTER 1

ONCORHYNCHUS NERKA POPULATION MONITORING IN THE SAWTOOTH VALLEY LAKES

David M. Teuscher, Doug Taki, and Kenneth Ariwite

Abstract - Critical habitat for endangered Snake River sockeye salmon includes five rearing lakes located in the Sawtooth Valley of central Idaho. Most of the lakes contain either introduced or endemic kokanee populations. Snake River sockeye occur naturally in Redfish Lake, and are being stocked in Redfish and Pettit Lakes. Because kokanee compete with sockeye for limited food resources, understanding population characteristics of both species such as spawn timing, egg-to-fry survival, distribution and abundance are important components of sockeye recovery. This chapter describes some of those characteristics.

In 1995, hydroacoustic estimates of *O. nerka* densities in the Sawtooth Valley Lakes ranged from 57 to 465 fish / ha. Densities were greatest in Pettit followed by Redfish (167), Alturas (95), and Stanley Lakes. *O. nerka* numbers increased from 1994 values in Pettit and Alturas Lakes, but declined in Redfish and Stanley. Despite a decline in total lake abundance, *O. nerka* biomass estimates in Redfish Lake increased.

Approximately 144,000 kokanee fry recruited to Redfish Lake from Fishhook Creek. *O. nerka* fry recruitment to Stanley and Alturas lake was 5,000 and 30,000 fry, respectively. Egg-to-fry survival was 14% in Fishhook and 7% in Stanley Lake Creek.

In Fishhook Creek, kokanee spawning escapement was estimated using stream surveys and a weir. Escapement estimates were 4,860 from weir counts, and 7,000 from stream surveys. As part of the kokanee reduction program, 385 of the spawning female kokanee were culled. Escapement for Stanley Lake Creek was only 60 fish, a ten fold decrease from 1994. In Alturas Lake, kokanee spawners dropped by 50% to 1,600.

INTRODUCTION

Historically, thousands of Snake River Sockeye salmon returned to the Sawtooth Valley to spawn. Evermann (1896) reported that the Sawtooth Valley Lakes were teeming with red fish. Bjornn (1968) estimated that 4,360 sockeye returned to Redfish Lake in 1955. These numbers no longer exist. In the 1980's, less than 50 Snake River sockeye salmon survived to spawn (Bowler 1990). Since 1990, only 14 sockeye have returned. Because of recent declines, the Shoshone-Bannock Tribes (SBT) petitioned the National Marine Fisheries Service (NMFS) to list Snake River sockeye as endangered. As a result, Snake River sockeye were listed and the Bonneville Power Administration (BPA) began funding recovery efforts.

Initial recovery efforts began in 1991 and focused on issues specific to the Sawtooth Valley Lakes, in central Idaho. Historically, these lakes (Redfish, Alturas, Pettit, Stanley, and Yellow Belly) provided critical spawning and nursery habitat for Snake River sockeye salmon. Exotic species introductions, low productivity, altered spawning conditions, and abundant kokanee populations may impact the ability of these systems to produce sockeye smolts. Some of our 1995 objectives that focused on the above questions included: completing hydroacoustic surveys to estimate total *O. nerka* abundance in Redfish, Stanley, Pettit and Alturas Lakes, completing kokanee and residual sockeye spawning surveys, and estimating kokanee fry recruitment from Fishhook and Stanley Lake Creeks. Also in 1995, 8,570 juvenile sockeye produced from the captive broodstock program were introduced to Pettit Lake. To facilitate migration of those sockeye, construction began to remove a non-game fish barrier and build a juvenile and adult collection facility on the outlet stream. These projects will help prevent extinction of Snake River sockeye salmon while solutions to poor smolt-to-adult survival are sought.

STUDY AREA

Four lakes in the Sawtooth Valley are currently the focus of our habitat and limnology studies. The lakes were glacially formed, range in elevation from 1985 to 2157 m, and are located in central Idaho (Figure 1). Specific features of the rearing lakes are shown in Table 1.

All of the Sawtooth Valley Lakes are oligotrophic. Mean summer total phosphorous concentration in the epilimnion range from 5.9 to 8.3 μg/L. Chlorophyll a concentrations range from 0.4 to 1.1 μg/L. Mean summer secchi disk transparencies range from 8 - 16 m.

Table 1. Morphological features of the Sawtooth Valley Lakes.

Lake	Area (km^2)	Volume (m^3x10^6)	Mean Depth (m)	Drainage Area (km^2)
Redfish	6.15	269.9	44	108.1
Alturas	3.38	108.2	32	75.7
Pettit	1.62	45.0	28	27.4
Stanley	0.81	10.4	13	39.4
Yellow Belly	0.73	10.3	14	30.4

Native fish species found in the nursery lake system include sockeye/kokanee salmon *O. nerka*, rainbow trout *O. mykiss*, chinook salmon *O. tshawytscha*, cutthroat trout *O. clarki*, bull trout *Salvelinus confluentus*, sucker *Catastomus sp.*, redside shiner *Richardsonius balteatus*, dace *Rhinichthys sp.*, northern squawfish *Pytchocheilus oregonensis*, and sculpin *Cottus sp.*. Non-native species include brook trout *S. fontinalis*, and lake trout *S. namaycush*.

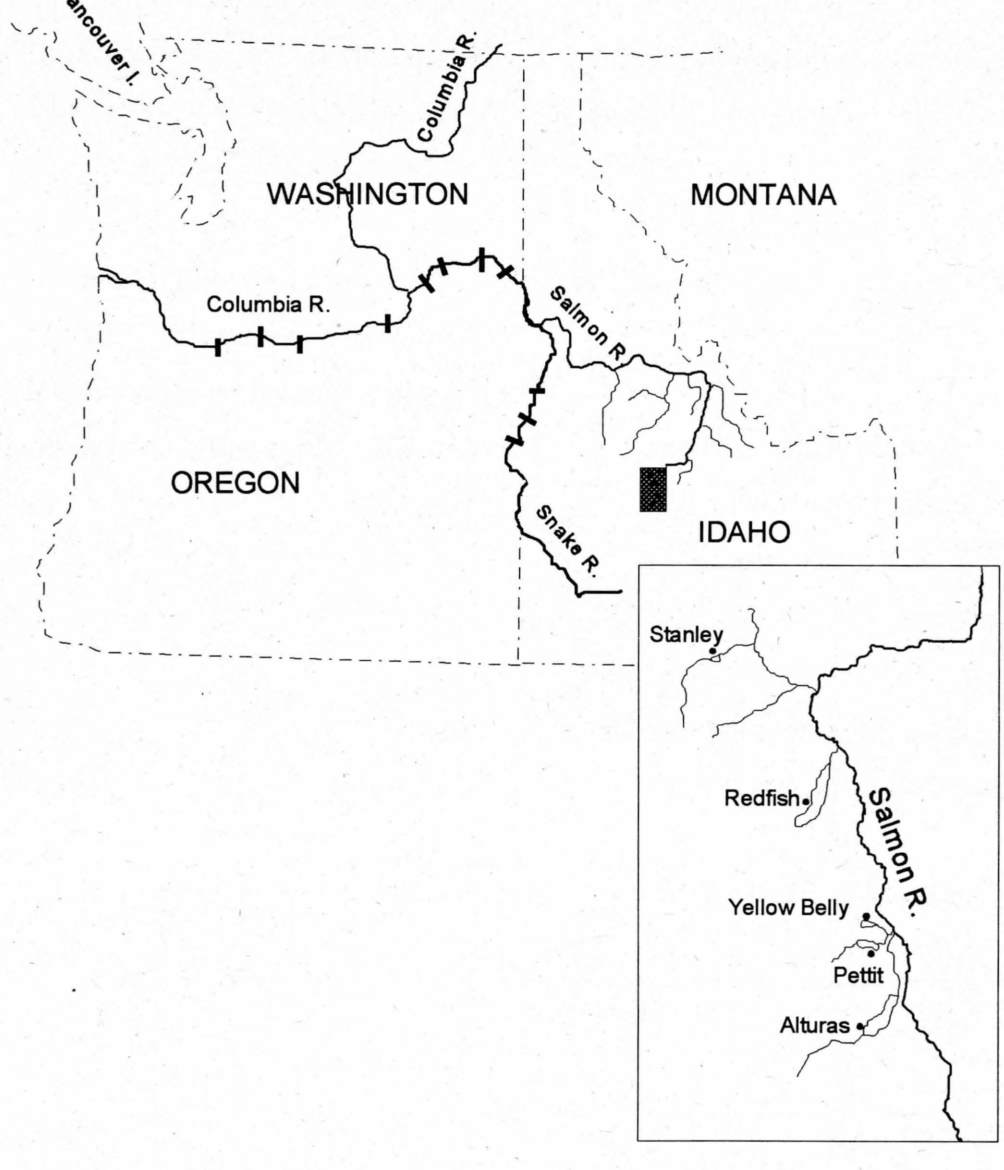


Figure 1. Sockeye rearing lakes in the Sawtooth Valley of central Idaho. Lines across the rivers indicate hydroelectric dams.

METHODS

Hydroacoustics

Data acquisition. - Echo sounding data were collected with a Hydroacoustic Technology, Inc. Model 240 split-beam system. We used a 15 degree transducer, and the echosounder criteria were set to a pulse width of 0.4 milliseconds, a time varied gain of 40 log(R) + 2αr, and four pings per second for Alturas and Redfish lakes, and five pings per second for Pettit and Stanley lakes. A minimum of four pings/target was necessary to qualify as a fish target. Data were recorded on a Panasonic SV-3700 digital audio tape recorder.

We followed identical transects as were set with a global positioning system (GPS) during 1994 (Teuscher and Taki 1995). Waypoints were set to allow for sampling transects to run zigzag across all lakes except Pettit Lake, where we used five parallel and one diagonal transects (Teuscher and Taki 1995). We sampled twelve, fourteen, and eight transects at Alturas, Redfish, and Stanley lakes, respectively. Sampling was conducted on two nights during the new moon phase in September, and one night in October on Redfish Lake. Sampling was started approximately one and a half hours after sunset. Boat speed during data collection ranged from 1-1.5 m/s.

Vertical gill netting and trawling (by IDFG) were done concurrently with hydroacoustic sampling to assist in partitioning targets. Past trawling efforts have shown a selectivity for *O. nerka*. Therefore, we employed vertical gillnets in Pettit Lake to determine if other fish species were found in the pelagic areas during sampling. Previous gill net sampling conducted in Alturas Lake has not yielded sufficient numbers for partitioning targets and therefore were not used. Due to permit requirements we were unable to set vertical gillnets in Redfish Lake.

Data analysis. - Target strengths and fish densities were processed using a Model 340 Digital Echo Processor and plotted with a Model 402 Digital Chart Recorder.

Target strengths were used to estimate fish length by the equation

$$TS = 19.1 \cdot \mathrm{Log}(L) - 0.9 \cdot \mathrm{Log}(F) - 62.0 \qquad (1)$$

developed by Love (1977) where TS = target strength in decibels, L = fork length in centimeters, and F = frequency of transmitted sound (kHz). Fish density estimates were calculated for different size classes for each lake to approximate year class densities based on previous years length frequency distributions and age analyses. In Alturas Lake only fish within 100-200mm were used to estimate *O. nerka* densities. That range encompasses the variance observed in trawl catches. Five different size classes were used for Pettit and Stanley lakes, and four for Redfish Lake. Total lake abundance and vertical distribution was also estimated.

Individual fish detections were weighted by the ratio of the designated area width to the diameter of the acoustic beam at the range of the detected targets. An effective beamwidth was calculated for each tracked target for the fish weighting algorithm.

The effective beamwidth equation

$$X\,(\mathrm{ABS} \cdot (M^{TS} - F^{TS})^{Y} \qquad (2)$$

was used where X = 8.6, M^{TS} = minimum system detection (-60), F^{TS} = mean target strength, and Y = 0.47.

Fish densities were computed by using adjacent transects as replicates within a stratum (lake). Population estimates for individual size classes were obtained with the equation

$$\overline{D}_i = \frac{\sum_{j=1}^{T_i} L_j \overline{D}_{ij}}{\sum_{j=1}^{T_i} L_j} \qquad (3)$$

and variance was estimated by

$$Var\,\overline{D}_i = \frac{T_i}{T_i - 1} \sum_{j=1}^{Ti} L_j^2 (\overline{D}_{ij} - \overline{D}_i)^2 / \left(\sum_{j=1}^{Ti} L_i \right)^2 \qquad (4)$$

where D_i = mean density (number/stratum) in stratum I, D_{ij} = mean density for the jth transect in stratum I, L_i = length of transect j, and T_i = number of transects surveyed in stratum I.

We used *FISHPROC* software to compile acoustic target information for each lake. This allowed us to select targets based on acoustic size, depth or various other parameters. We could process single or multiple transects and fish were sorted into one decibel bins. Vertical distribution was estimated by

$$\overline{D}_i = \sum_{i=1}^{h} D_{vi}(R_{iu} - R_{il}) \qquad (5)$$

where D_{vi} = number of fish/m^3 in depth stratum I, R_{iu} = upper range limit for depth stratum I, R_{il} = lower range limit for depth stratum I, and h = number of depth strata. These values were then multiplied by the percentage of each depth stratum surveyed within the conical beam.

Tributary Fry Recruitment

Kokanee fry recruitment was estimated for Fishhook and Stanley Lake Creeks. Fry are collected with steal frame drift nets that are 30cm wide and 60 cm deep. Each frame was fitted with a tapered nylon net. The nylon nets direct fry to live boxes made with plywood and wire-mesh screen with longitudinal partitions used to create backwater refuge for captured fish. The frames, nets, and live boxes are anchored to the substrate with rebar. In Fishhook Creek, between five and three fry traps were set under the bridge located 50 m above the confluence with the lake. In Stanley Lake Creek two fry traps were deployed about 200 m above the confluence with the lake. Nets were generally fished between 1800 and 0800 hrs. After enumeration, all the fry captured in Stanley Lake Creek traps were released. To meet kokanee management goals, a portion of the Fishhook Creek fry were transported to the Sawtooth Hatchery, for eventual introduction to a lake outside the Sawtooth Valley. Fry traps were operated at least four days each week from 7 April to 28 June 1995.

Daily fry recruitment was estimated by summing the number of fish caught in each trap, and applying a correction factor for trap efficiencies. Trap efficiencies were estimated using a radish release method. To estimate the proportion of the stream sampled, about 100 radishes were released 100 m above the traps. The number of radishes captured in the traps provided an estimate of the proportion of total recruitment sampled. In 1993, we compared the radish release method to traditional fry mark-recapture techniques. The radish release estimates of trap efficiencies were within 2% of mark-recapture results (Teuscher et al. 1994).

Smolt Monitoring

A rotary screw trap was used to monitor smolt production from Alturas Lake Creek. The screw trap was placed upstream of the confluence of Pettit and Alturas Lake Creeks. The placement of the screw trap allowed us to estimate sockeye smolt migration from Alturas Lake as well as contributions from Pettit and Yellow Belly Lakes. Beginning in May, initial plans were to operate the trap several nights a week until smolts began outmigrating and then operate the trap every night. The trap was set between 1800 and 2000 hrs and pulled by 0800 hrs. To protect juvenile salmonids from being consumed by squawfish or salmonids, sage brush bundles were placed in the live box. Periodic sampling continued through June without catching a single *O. nerka*.

O. nerka Spawning Surveys

Stream Spawning -We conducted stream surveys to estimate tributary spawning abundance in Fishhook, Stanley, and Alturas Lake Creeks. Counts were completed from the bank by one or two observers equipped with polarized sunglasses. Surveys were conducted at three day intervals. On days when counts were missed, the number of fish in the stream was estimated by averaging the counts preceding and following the period with missed counts. Total escapement estimates were made by summing daily counts of kokanee and dividing by average stream life as described by English et al. (1992).

In addition to the English et al. method, a picket weir was deployed on Fishhook Creek. The weir served two purposes. First, the weir was used as an alternative method for estimating kokanee escapement. Secondly, the weir was used to cull a portion of the female kokanee spawners. Culling was suggested as part of a kokanee management program. The objective of the program is to replace 50% of the kokanee population with juvenile sockeye salmon. Escapement goals for kokanee in Fishhook

Creek were set at 2,000 female spawners. Egg deposition and subsequent fry production from 2,000 females should be about 460,000 eggs and 50,000 fry.

Beach Spawning -Sockeye Beach and a small section of the south east corner of Redfish Lake are spawning grounds for residual sockeye. In 1995, snorkel surveys were completed to estimate the relative abundance of spawners in both locations. Snorkel surveys were completed by at least two observers swimming parallel to shore at depths ranging from 0.5 to 5 m. For Sockeye Beach, we estimated residual spawner abundance within the boundaries of Sockeye Beach as delineated by U.S.D.A. Forest Service signs. The total length of beach snorkeled was about 600 m. Based on past observation, spawning in the south end of the lake is restricted to the shoal area near the two south-east inlet streams (about a 200 m section). Surveys began the first week in October and were completed every other week through the end of November.

RESULTS AND DISCUSSION

Hydroacoustics

Hydroacoustic estimates of *O. nerka* densities ranged from 57 to 465 fish / hectare. Densities were highest in Pettit followed by Redfish (167), Alturas (95) and Stanley Lakes (Figure 2). Volumetric densities were .30, .44, .61, and 1.95 *O. nerka* / 1000 m^3 for Alturas, Stanley, Redfish, and Pettit lakes, respectively. Lakes that experienced the greatest changes in *O. nerka* abundance from 1994 were Stanley (-83% decline) and Pettit (510% increase; Table 2).

Biomass estimates (kg / ha) increased from 1994 estimates in all of the lakes except for Stanley (Table 3). Surprisingly, *O. nerka* biomass increased in Redfish Lake despite a decline in overall abundance. The contradiction between less fish but greater biomass occurred because age-1 and age-2 *O. nerka* made up a larger percentage of the overall population and were slightly larger in size (Table 3).

Since 1994, hydroacoustic and trawl estimates of *O. nerka* biomass and densities have shown similar patterns (Figure 2 & 3). For example, in Pettit Lake, both trawl and hydroacoustic estimates of *O. nerka* densities increased significantly from 1994 (Figure 3). The only trend that was different between the two sampling techniques was in Redfish Lake where trawling showed an increase in the population of *O. nerka* and hydroacoustic estimates were lower (Table 2).

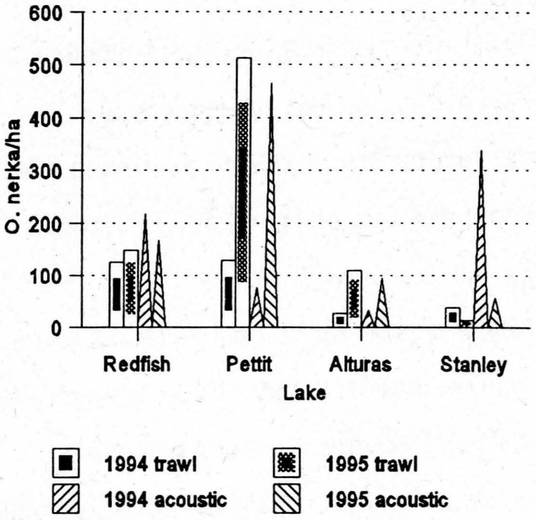


Figure 3. A comparison of *O. nerka* densities between trawl and hydroacoustic estimates for 1994 and 1995 (trawl data from Paul Kline, IDFG)

Table 2. Trawl and acoustic estimates of *O. nerka* abundance in the Sawtooth Valley Lakes. Trawl data is from Kline and Younk (1995).

Lake	Year	Acoustic	Trawl	Ratio A/T
Redfish	1995	102,600	61,600	1.7
Redfish	1994	133,400	51,500	2.6
Redfish	1993	203,500	49,600	4.1
Redfish	1992	188,000	39,500	4.8
Alturas	1995	32,300	23,000	1.4
Alturas	1994	11,000	5,800	1.9
Alturas	1993	200,700	49,000	4.1
Alturas	1992	144,000	47,000	3.1
Alturas	1991	230,000	125,000	1.8
Pettit	1995	75,400	59,000	1.3
Pettit	1994	12,400	14,700	0.8
Pettit	1993	20,400	10,500	1.9
Pettit	1992	19,000	3,000	6.3
Stanley	1995	4,600	1,000	4.6
Stanley	1994	27,500	2,700	10.2
Stanley	1993	32,400	1,300	24.9
Stanley	1992	33,000	2,100	15.7

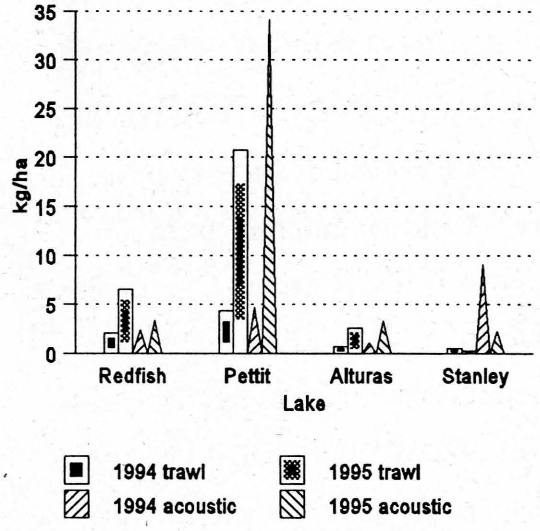


Figure 2. A comparison of *O. nerka* biomass between trawl and hydroacoustic estimates for 1994 and 1995 (trawl data from Paul Kline, IDFG).

Table 3. Hydroacoustic results for four Sawtooth Valley Lakes during September 1995.

Size Class (mm)	Population Estimate	% change from 1994	Mean wt (g)	% change from 1994	Biomass Estimates (kg)	% change from 1994	kg / ha	% change from 1994
Redfish								
30-70	22,400 ± 6,400	- 71	1.5	+ 25	33.5 ± 9.6	- 65		
71 - 130	49,100 ± 12,400	+36	10.7	+ 4	526.6 ± 132.7	+ 41		
131 - 210	31,100 ± 12,300	+50	49.3	+2	1,531.8 ± 608.4	+ 53	3.4	+ 42
all targets	146,950 ± 33,100	- 18						
Pettit								
25 - 60	2,900 ± 1,300	- 37	1.4	+ 55	4.1 ± 1.8	0		
61 - 120	15,600 ± 9,300	+687	10.5	+25	164.4 ± 98.3	+ 890		
121 - 200	37,300 ± 23,600	+1,061	59.1	0	2,203.1 ± 1,393.4	+ 1,061		
201 -250	19,700 ± 13,900	+ 656	161.0	- 24	3,162.5 ± 2,238.0	+ 479	34.2	+ 631
> 300	10,100 ± 9,300	+107						
all targets	100,500 ± 61,200	+483						
Stanley								
50 - 100	1,200 ± 900	- 91	4.7	+18	5.5 ± 1.2	- 89		
101 - 170	1,900 ± 1,300	- 80	24.0	- 6	45.2 ± 31.8	- 82		
171 - 240	1,600 ± 800	- 70	86.0	+ 2	134.2 ± 68.2	- 69	2.3	- 75
260 - 340	1,800 ± 1,000	- 30						
> 360	1,000 ± 800	- 77						
all targets	8,700 ± 2,800	- 76						
Alturas								
100 - 200	32,300 ± 5,100	+ 194	34.7	+6	1,119.4 ± 176.6	+ 210	3.3	+ 209
all targets	80,900 ± 11,200	+ 55						

Alturas Lake - We estimated total fish abundance in Alturas Lake at 80,900 ± 11,200 during September of 1995 (Table 3). By partitioning targets into a conservative size range of 100 - 200 mm, *O. nerka* abundance was 32,300 ± 5,100, an increase of 194% from the previous year. The size range selected for *O. nerka* was based on trawl and gillnet catches. Rainbow trout, squawfish, and suckers may have all contributed to non-*O. nerka* fish targets (48,600 total).

Pettit Lake - We estimated fish abundance in Pettit Lake during September at 100,500 ± 61,200 (Table 3) of which 75,400 were partitioned as *O. nerka* (25 - 250 mm). The estimate is a 510% increase from 1994. One factor that may have contributed to the increase was a change in survey design. The 1995 survey of Pettit Lake included an additional transect that was not used in 1994. While this complicates the comparison, the IDFG's trawling survey also showed an increase of 300%. Also, we captured redside shiners in the vertical gillnets that fell within the size range used for estimating *O. nerka* abundance, but they were all caught less than five meters deep and were not associated with other tracked targets.

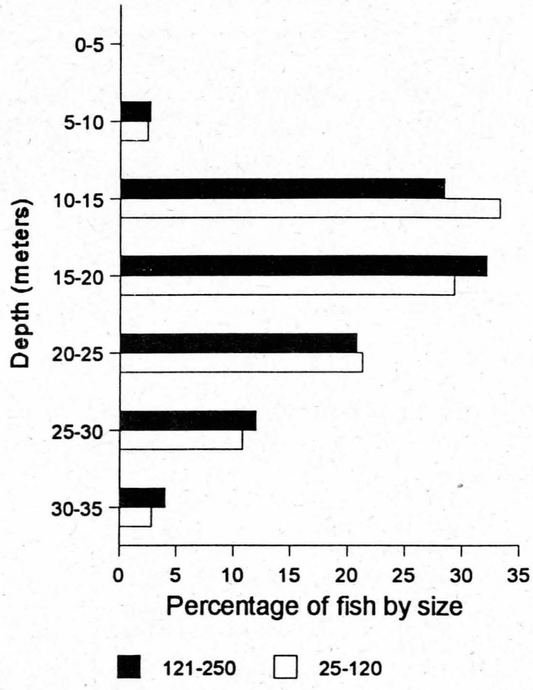


Figure 4. Vertical distribution of *O. nerka* in Pettit Lake during September of 1995.

Vertical distribution of *O. nerka* in Pettit Lake was concentrated in the ten to twenty-five meter strata, with small numbers found in adjacent depths (Figure 4). Unlike 1994, no hatchery rainbow trout were captured in vertical gillnets. *O. nerka* captured in vertical gillnets were found at similar depths as shown in hydroacoustics but the sample size was very small (n = 8).

Redfish Lake - In September, total lake abundance was 146,900 ± 33,100 fish (Table 3). In lake *O. nerka* (targets with lengths 25 - 210 mm) abundance was 102,600. That estimate is a 23% decline from the previous year. An additional 5,000 age-3 kokanee were spawning in Fishhook Creek during the hydroacoustic survey that were not included in the estimate.

Vertical distribution of fish in Redfish Lake was similar to Pettit Lake, with most of the targets distributed in ten to twenty meters of water (Figure 5). The fifteen to twenty meter strata contained the highest proportion of all three size classes. No fish were located above five meters, and few fish were found below thirty meters

Survival of age-0 kokanee in Redfish Lake in 1995 was poor. Fry survival from May to September was only 16% (22,400 fall fry / 144,000 spring recruits). Survival of older year classes was much better. From September 1994 to September 1995, age-1 and age-2 survival was 39% and 46%, respectively.

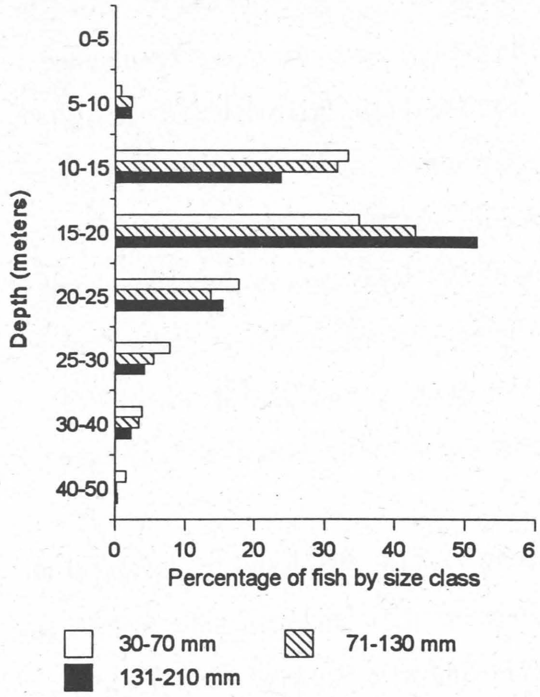


Figure 5. Vertical distribution of *O. nerka* in Redfish Lake during September of 1995.

In addition to the September monitoring program, we completed a hydroacoustic survey in October. The purpose of the survey was to determine movement and survival of sockeye released from net pens and direct lake releases. The sockeye (about 60,000 total) were released two weeks prior to completing the survey.

Unfortunately, the October population estimate dropped 28% from September. Abundance declines were 5.8, 10.9, and 51.8% for age-0, 1, and 2 *O. nerka*, respectively. If the sockeye recruited to the pelagic zone, the age-1 abundance estimate should have doubled. Snorkeling observation indicated that many of the 30,000 sockeye released from the boat ramp did not recruit to the pelagic zone. The net-pen fish, however, were released in the pelagic zone, and should have contributed to the population estimate.

Except for the smallest size class, spatial distribution of fish targets in Redfish Lake were similar in September and October (Figure 6). Fish densities were highest in the south end of the Lake (Figure 6; right side). In September, most of the small targets were sampled mid-lake (transects 6 -9; Figure 6). In October, small targets were evenly distributed.

In 1995, pelagic fish populations were not controlling the zooplankton population of Redfish Lake. If fish were controlling zooplankton densities, the two parameters should be negatively correlated. Conversely, in 1995, fish and zooplankton densities were positively correlated (Figure 6). It appears that lake fertilization increased zooplankton production enough to dampen or eliminate predation impacts by fish. Therefore, the major objective of lake fertilization was accomplished. Specific lake fertilzation results are reported in a later chapter of this report.

Stanley Lake - *O. nerka* abundance in Stanley Lake was 4,610, a 600% decline from 1994. Trawl estimates and spawning populations also dropped significantly. The decline was proportional for all size classes (Table 3).

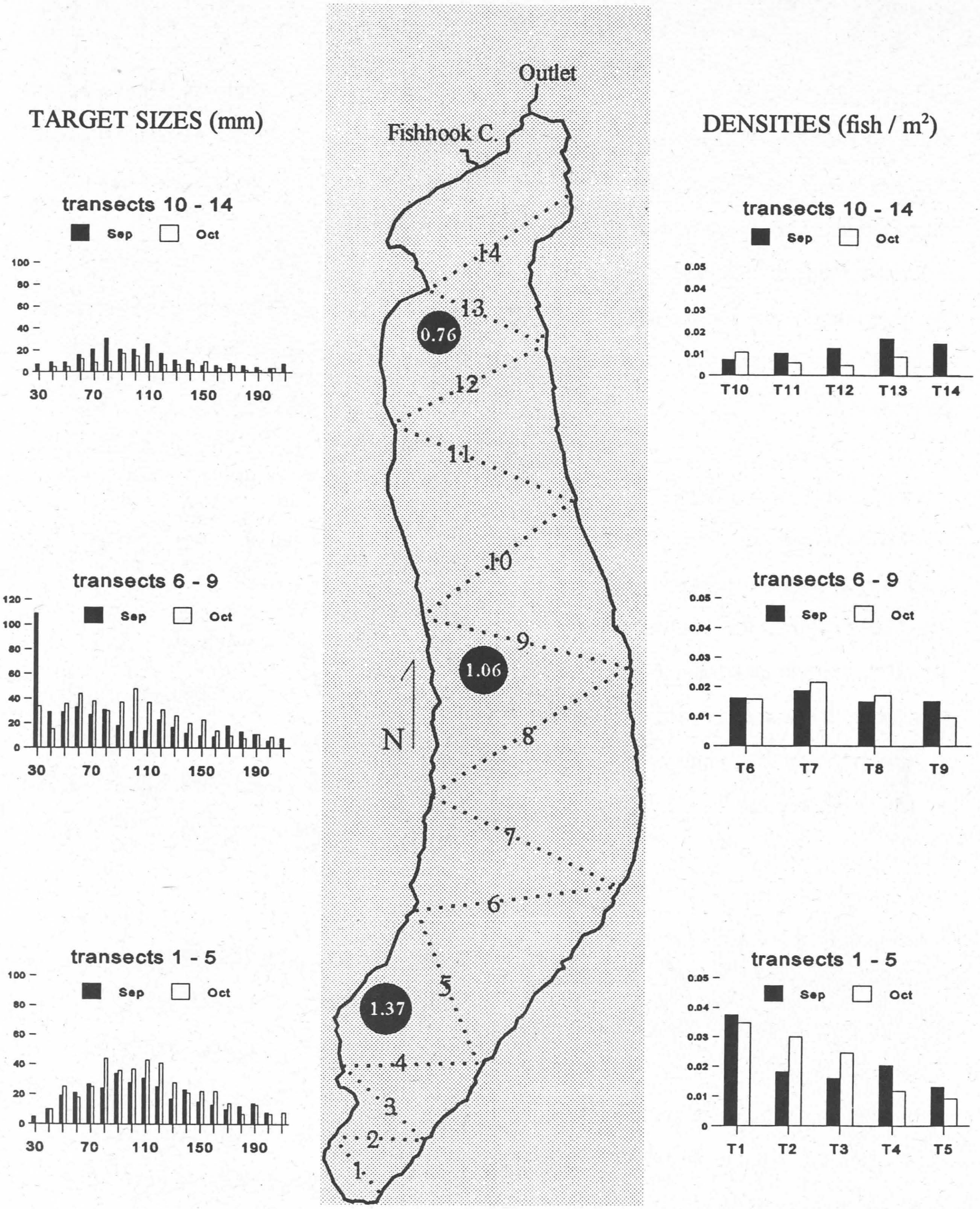


Figure 6. Length frequency distributions (left side) and fish densities in fish / m^2 (right side) from hydroacoustic surveys completed in September and October in Redfish Lake (center). Fish densities were highest in the South end of the lake, which followed trends in forage abundance (see black circles for zooplankton numbers per liter collected in October).

Tributary Fry Recruitment

An estimated 144,000 kokanee fry entered Redfish Lake from Fishhook Creek. The recruitment numbers were similar to estimates made in 1993 and 1994 (Table 4). Assuming equal sex ratios and 230 eggs per female, egg-to-fry survival for 1995 was 14%.

Stanley Lake Creek produced an estimated 5,000 kokanee fry in 1995. The 1995 recruitment was down from 19,000 reported in 1994. The reduction in fry numbers was expected due to a similar rate of decline in spawning densities. Egg-to-fry survival was 7%, which was the same as the 1994 estimate

In Alturas Lake, high water conditions washed out the fry traps for most of the recruitment period. Therefore, we did not attempt to estimate recruitment based on daily counts and trap efficiencies. A rough estimate can be made using the 13% egg-to-fry survival estimate from 1994. About 240,000 eggs were deposited in Alturas Lake Creek in 1994. At 13% survival, 30,000 fry would have recruited to the lake. The 1995 estimate is a fifteen fold increase over 1994 (Table 4). Unlike current trends in Stanley Lake, fry numbers suggest a modest rebound in the Alturas Lake *O. nerka* populations.

Table 4. Fry recruitment and egg-to-fry survival in Fishhook, Alturas, and Stanley Lake Creeks.

creek	year	recruits	egg/fry survival
Fishhook	1995	144,000	14%
Fishhook	1994	142,000	11%
Fishhook	1993	166,000	12%
Fishhook	1992	36,000	3%
Alturas	1995	30,000	-
Alturas	1994	2,000	13%
Alturas	1992	7,000	-
Stanley	1995	5,000	7%
Stanley	1994	19,000	7%

Emergence timing has been consistent among systems and between years. Emergence begins the first week of April, peaks the end of May and is complete by the first week of July. In Fishhook Creek, fry recruitment is also correlated to discharge. The recruitment peaks in 1995 mirrored increases in discharge (Figure 7). Similar patterns may occur in Stanley and Alturas, but discharge data were not available.

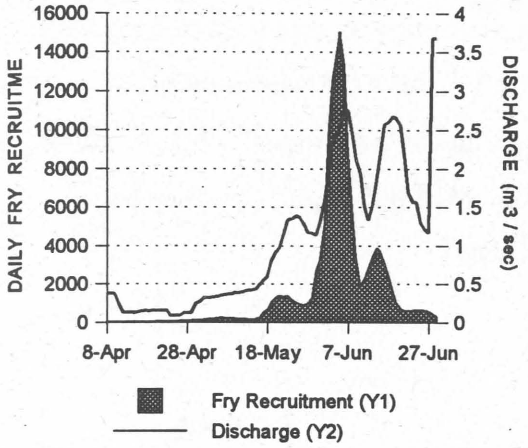


Figure 7. Fry recruitment and discharge for Fishhook Creek.

Spawning Surveys

Stream Spawners: Using the area under the curve (AUC) method, kokanee escapement in Fishhook, Alturas, and Stanley Lake Creeks was 7,000, 1,600, and 90, respectively. All three of the population experienced declines (Figure 8; Table 5). Since 1992, spawning populations have been most variable in Alturas Lake Creek ranging from 60 to 3,200. The Stanley Lake populations has experiences a constant rate of decline (Figure 8). Conversely, the Fishhook Creek populations appears to be more stable.

Table 5. Escapement estimates for Fishhook, Alturas, and Stanley Creeks.

Creek	Year	Escapement
Fishhook	1995	7,000
Fishhook	1994	9,200
Fishhook	1993	10,800
Fishhook	1992	9,600
Fishhook	1991	7,200
Alturas	1995	1600
Alturas	1994	3,200
Alturas	1993	200
Alturas	1992	60
Stanley	1995	90
Stanley	1994	600
Stanley	1993	1,900

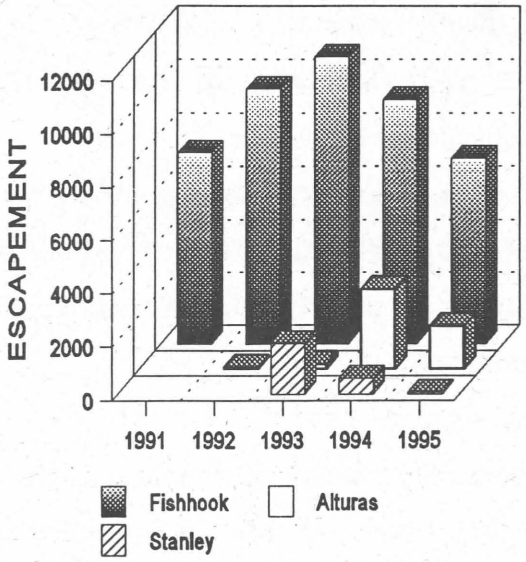


Figure 8. Escapement estimates for Fishhook, Alturas, and Stanley Creeks.

Using weir counts, an estimated 4,860 kokanee entered Fishhook Creek to spawn. The weir estimate is a 30% reduction in spawning abundance compared to the AUC method. The differences may be due to interpolation error (i.e., the weir was pulled on several occasion; Table 6).

The male-to-female ratio of kokanee spawners collected at the weir was 1.2. Males dominated the first two weeks of the run (8-Aug to 22-Aug), but were replaced by females later in the run (Table 6). Not all of the fish that reached the weir were passed. Approximately, 99% of the males and 82% of the females were released to spawn above the weir. Fish removed from the creek were used in age and fecundity estimates and (or) culled from the population as part of kokanee management programs. A total of 385 females (18%) and 33 males (1%) were removed from the creek.

Spawning size and fecundity was similar to results from 1994. Mean fork length was 222 mm for females and 220 mm for males. Fecundity was 230 eggs per female (n = 68; Figure 9). Each female carried about 18.4 g of eggs, which made up 15% of total body weight. Females passed over the weir deposition an estimated at 416,000 eggs (1809 females * 230 eggs / female). Resulting fry recruitment should be about 51,000 (416,000 eggs * 0.123 egg/fry survival).

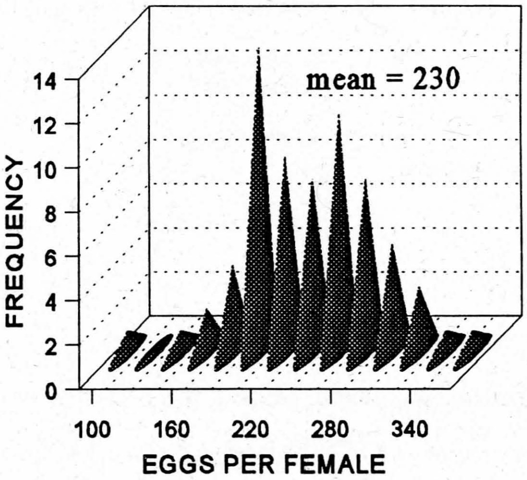


Figure 9. Frequency distribution of fecundity for Fishhook Creek kokanee in 1995. Mean fecundity is 230 eggs per female (n = 68).

Similar to fry recruitment, run timing patterns have been consistent in Fishhook Creek. Adult kokanee entered the creek the first week of August, counts peak the last week of August, and the kokanee drop out of the creek by the middle of September (Table 7). Spawn timing for all the creek populations in the Sawtooth Valley are reported in Table 7.

Table 6. Weir and stream counts for kokanee spawning in Fishhook Creek. The weir was pulled periodically to allow bull trout passage. On days when the weir was pulled, the number of fish passing the weir was estimated by averaging the counts preceding and following the period with missed counts. A stream survey count was used to estimate weir passage on 21 August, when the weir was pulled to release a large number of kokanee stacked up below the weir. After 21 August, the weir was modified to reduce short stopping. The last column of data is the total number of kokanee observed during stream surveys.

Dates and notes	♂	♀	total	♂/♀	♂ released	♀ released	stream counts
8/04							41
8/08							57
8/16 deployed weir	1,000	706	1,706		1,000	706	1,706
8/17	97	68	165	1.4	97	0	
8/18	154	73	227	2.1	154	0	
8/19	165	109	274	1.5	165	0	2,100
8/20	128	91	219	1.4	128	91	
8/21 pulled weir	690	489	1,179		690	489	3,498
8/22	62	44	106	1.4	50	32	
8/23 trap failure	40	51	93		40	51	
8/24 trap failure	40	51	93		40	51	3,241
8/25	18	58	76	0.3	18	58	
8/26	36	61	97	0.6	36	53	
8/27 pulled weir	40	53	93		40	53	
8/28 pulled weir	40	53	93		40	53	3,463
8/29	43	45	88	1.0	43	45	
8/30	45	88	133	0.5	24	38	3,597
8/31	22	60	82	0.4	22	30	
9/01	17	35	52	0.5	17	0	3,484
9/02 pulled weir	13	27	40		13	27	
9/03 pulled weir	9	18	27		9	18	
9/04 pulled weir	5	9	14		5	9	2,974
9/05	1	1	2	1.0	1	1	
9/06 pulled weir	1	1	2		1	1	2,308
9/07	0	2	2	0.0	0	2	
9/08 removed weir	0	1	1		0	1	2,008
9/12							1,499
9/19							654
9/26							0
totals	**2,666**	**2,194**	**4,864**	**1.2**	**2,633**	**1,809**	

Beach Spawners: In 1995, fewer residual sockeye were observed during snorkel surveys on Sockeye Beach and in the south end of Redfish Lake than in past years. No residual sockeye were observed in the south end of the lake and the peak count on Sockeye Beach was only 10. In 1994, peak counts were 22 for Sockeye Beach and 40 in south end. The reason for fewer residual observation may be related to a reduction in sample frequency, natural variation in the population, or the fact the no anadromous fish have spawned with the residual population since 1992.

Table 7. Escapement timing for Fishhook, Alturas, and Stanley Lake Creeks. Mean number of days past 1-Aug (range) that 20, 50, and 80% of the total spawning populations had entered each creek. The mean of four years of data was used (1992-1995), except for Stanley where observation began in 1993. The late (October) creek spawning population from Stanley was not included in calculations.

Creek	20%	50%	80%
Fishhook	21	29	38
	(20-23)	(28-32)	(33-41)
Alturas	18	26	33
	(16-20)	(23-32)	(27-42)
Stanley	12	18	27
	(12-12)	(16-20)	(18-33)

REFERENCES CITED

Bjornn, T. C., D. R. Craddock, and D. R. Corley. 1968. Migration and survival of Redfish Lake, Idaho, sockeye salmon, Oncorhychus nerka. Trans. Am. Fish. Soc. 97:360-373.

Bowler B. 1990. Additional information on the status of Snake River sockeye salmon. Idaho Department of Fish and Game, Boise, Idaho.

English, K. K., R. C. Bocking , and J. R. Irving. 1992. A robust procedure for estimating salmon escapement based on the area -under-the-curve method. Can. J. Fish. Aquat. Sci. 49: 1982-1989.

Evermann, B. W. 1896. A report upon salmon investigations in the headwaters of the Columbia River, in the state of Idaho, in 1895, together with notes upon the fishes observed in that state in 1894 and 1895. Bulletin of the United States Fisheries Commission. 16:151-202.

Teuscher, D., and D. Taki. 1995. Snake River sockeye salmon habitat and limnological research. *In*, D. Teuscher and D. Taki (eds.). Snake River sockeye salmon habitat and limnological research. U.S. Department of Energy, Bonneville Power Administration, Portland, OR. Project number 91-71. pp. 1-50.

CHAPTER II

REVIEW OF POTENTIAL INTERACTIONS BETWEEN STOCKED RAINBOW TROUT AND LISTED SNAKE RIVER SOCKEYE SALMON IN PETTIT LAKE, IDAHO

David Teuscher

Abstract- The objective of this study was to determine if hatchery rainbow trout compete with or prey on juvenile Snake River sockeye salmon *Oncorhynchus nerka* in Pettit Lake, Idaho. In 1995, a total of 8,570 age-0 sockeye and 4,000 hatchery rainbow trout were released in Pettit Lake. After releasing the fish, gillnets were set in the pelagic and littoral zones to collected diet and spatial distribution data. Interactions were assessed monthly from June 1995 through March 1996. Competition for food was discounted based on extremely low diet overlap results observed throughout the sample period. Conversely, predation interactions were more significant. A total of 119 rainbow trout stomachs were analyzed, two contained *O. nerka*. The predation was limited to one sample period, but when exptrapolated to the whole rainbow trout populations resulted in significant losses. Total consumtion of *O. nerka* by rainbow trout ranged from an estimated 10 to 23% of initial stocking numbers. Predation results contradict earlier findings that stocked rainbow trout do not prey on wild kokanee or sockeye in the Sawtooth Lakes. The contradiction may be explained by a combination of poorly adapted hatchery sockeye and a littoral release site that forced spatial overlap that was not occurring in the wild populations. Releasing sockeye in the pelagic zone may have reduced or eliminated predation losses to rainbow trout.

INTRODUCTION

The proposed recovery plan for Snake River salmon identified predation and competition as potential factors contributing to salmon declines (Schmitten et al. 1995). To date, most research has focused on interactions in the migration corridor (see review by Bevan et al. 1994). Marine mammals, birds and piscivorous fish contribute to salmon mortalities. In addition to a hostile migration corridor, juvenile salmon are subject to potential predators and competitors in their natal stream and lake environments. O'Neill and Hyatt (1987) concluded that threespine sticklebacks *Gasterosteus aculeatus* compete for food resources with juvenile sockeye *Oncorhynchus nerka* in coastal lakes of British Columbia. Beauchamp (1990) identified hatchery rainbow trout *Oncorhynchus mykiss* as a significant predator of sockeye salmon in Lake Washington. Similar processes may slow recovery efforts for Snake River sockeye in the Sawtooth Valley Lakes of Idaho.

In 1993, stocking catchable rainbow trout in Redfish Lake, a nursery lake for Snake River sockeye, was discontinued because of concerns that the trout may compete with or prey on listed sockeye. Since then, diet studies completed in other Sawtooth Valley Lakes have failed to detect negative interactions between hatchery rainbow trout and non-anadromous *O. nerka* (Teuscher et al. 1994). However, inferences made about predation and competition were limited to spring and fall sample periods. This paper addresses that concern by describing potential interactions on a monthly basis. The specific questions addressed in this study include; (1) do rainbow trout prey on *O. nerka* and if so what is the magnitude of predation, (2) do rainbow trout and *O. nerka* compete for food, and (3) what is the rate of overwinter survival for stocked rainbow trout?

STUDY SITE

Pettit Lake is one of five historic Snake River sockeye salmon nursery lakes in the Sawtooth Valley. Pettit is a small (163 ha) oligotrophic lake at an elevation of 2,132 m. Fish species include redside shiners *Richardsonius balteatus*, brook trout *Salvelinus fontinalis*, kokanee

Oncorhynchus nerka, and hatchery rainbow trout. All species, except redside shiners, were introduced after a fish eradication project completed in the 1960's. On 27 July 1995, 8,570 age-0 sockeye were introduced to the littoral zone. The sockeye release was the first attempt to reestablish sockeye to Pettit Lake. Prior to the sockeye release, 4,000 rainbow trout (25 trout / ha) were stocked from the boat ramp of Pettit Lake (Figure 1; also note sockeye release site). The trout were age-1 with a mean weight of 155 g (fork length range ≈ 200 - 260 mm). Stocking procedures were similar in 1994 and 1993 when 4,000 and 3,500 rainbow trout were stocked in Pettit Lake, respectively.

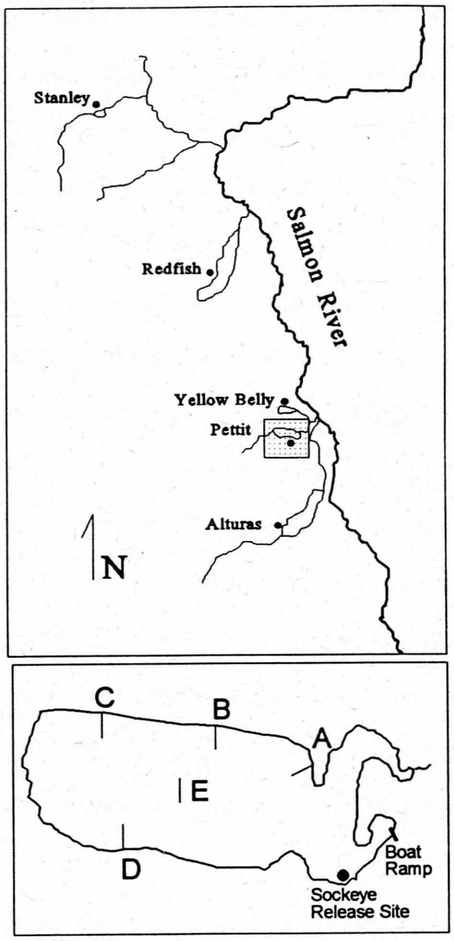


Figure 1. Sawtooth Valley lakes in central Idaho. Gillnet locations and sockeye release site in Pettit Lake. A - D were horizontal gillnet sets and E was the vertical gillnet set.

METHODS

Gillnets were used to estimate population size, distribution, and diet of rainbow trout and *O. nerka*. Nets were set in the littoral and pelagic zones of Pettit Lake (Figure 1). For sampling the littoral zone, four 30 m horizontal nets with square mesh sizes ranging from 2.5 to 7.6 cm were used. To sample pelagic habitat, five vertical gillnets of similar mesh sizes were set mid-lake. To prevent lethal take of listed sockeye, the smaller meshes were not deployed after 27 July. Littoral nets were set bimonthly and vertical nets monthly. Fish were identified, measured for weight and length, otoliths removed for aging, and stomach contents preserved for diet analysis. Prey from rainbow and *O. nerka* were categorized as zooplankton, aquatic insect, mollusks, redside shiners, etc. Diet overlap indices were calculated using equations described in Koenings et al. (1987).

Because *O. nerka* were found in rainbow trout stomachs, total consumption estimates were made using bioenergetics model II (Hewett and Johnson 1992). Model inputs include rainbow trout diet, abundance, growth, and water temperature. Rainbow trout growth was estimated using the mean differences in weight between age-1 and age-2 individuals. Diet results for the month of August were pooled (n = 35) and recorded in the model as aggregate percent by weight. It is important to note that diet results for rainbow trout were collected from four sample locations in Pettit Lake and should account for spatial overlap, or the lack there of, between predators and prey. Temperature inputs were based on average rainbow trout depth in gillnets and the associated water temperatures. The model ran for 20 days; beginning the day that sockeye were stocked (27 July) and continuing through 16 August. The end date was determined by taking the mid-point between observed predation (8 August) and when *O. nerka* were no longer present in the diet (25 August).

Rainbow Trout Population Estimate

To estimate the total rainbow trout population in Pettit Lake, we used age information in combination with a Petersen mark-recapture method. Otoliths from a total of 71 rainbow trout were evaluated for age. Approximately 61% of the trout were age-1 (Figure 2). The age-1 rainbow trout were considered recaptures in the Petersen mark-recapture equation. The initial number of marked (age-1) rainbow trout in the population was 4,000, but that number was corrected to account for natural mortality and harvest. Recent return-to-creel data generated for Redfish, Alturas, and Stanley Lakes indicates anglers harvested 17%, 39%, and 33% of the catchable rainbow trout planted in 1987, respectively (Reingold and Davis 1987). Approximately 85% of the 1987 harvest occurred during the months of May, June, and July. No data was available for Pettit Lake, but personal observation of fishing pressure suggests that Pettit Lake is probably at the low end of the harvest range (15% was used; 4,000 * 0.15 = 600). Secondly, Miller (1953) observed a range in natural mortality of pond-reared hatchery rainbow trout from 35 to 66.2% over the first summer. In a later report, Miller (1958) noted that most of the mortality occurred in the first two weeks after release. Natural mortality and angler harvest were used to create two modeling scenarios; a high predator density model that assumed 30% natural mortality and 15% harvest, and a low predator density model that assumed 60% natural mortality and 15% harvest. These data were used in the population estimate as follows:

$N = mc / r$

95% confidence intervals $= N \pm 1.96$ SE

$$SE=\sqrt{m^2 * c(c-r)/r^3}$$

where;

N = total rainbow trout population

m_{high} = number marked [(4,000 * 0.70) - 600] = 2,200)

m_{low} = number marked [(4,000 * 0.40) - 600] =1,000)

c = total number of trout caught in gillnets

r =number of age-0 recaptures

therefore;

N_{high} = 2,200 * 131 / 80 = **3,603 ± 320**

N_{low} = 1,000 * 131 / 80 = **1,638 ± 114**

Additionally, rainbow trout size is an important factor to consider in the predator population estimate. Beauchamp (1990) and Parkinson et al. (1989) described significant ontogenetic shifts in rainbow trout feeding behavior. In Pettit Lake, roughly 90 % of the piscivorous rainbow trout were 270 mm or larger. Therefore, the predator abundance variable added to the model was the proportion of the rainbow trout population 270 mm or larger (52%). The high predator population estimate was **1,874** (3,603 * 0.52) and the low was **852** (1,638 * 0.52).

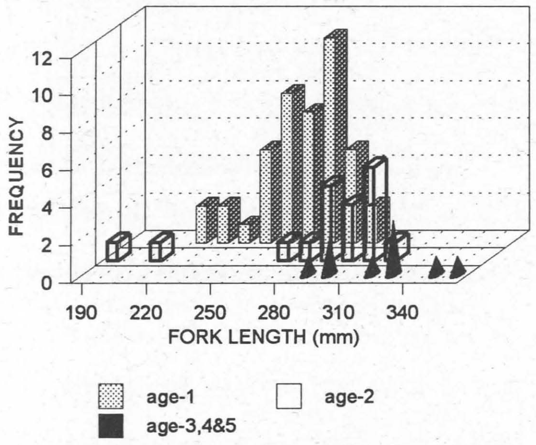


Figure 2. Length frequency distributions for age-1 through age-5 rainbow trout caught in Pettit Lake gillnets.

RESULTS

Competition For Food

A total of 119 rainbow trout and 75 *O. nerka* stomachs were analyzed. In general, rainbow trout diet was dominated by dragon fly larvae, snails and miscellaneous terrestrial insects (Table 1). Spring diet for *O. nerka* consisted primarily of chironomid pupae. Later in the year, *O. nerka* selected zooplankton prey (Figure 3). Diet overlap between rainbow trout and *O. nerka* was generally very low. Overlap ranged from 0 to 22% (Table 1). Even when *O. nerka* and rainbow trout occupied similar habitat (spring littoral zone), they foraged on primarily different prey. Winter diet results are not yet complete, but it is unlikely that competition will occur under the ice when *O. nerka* feed exclusively on small zooplankton prey (Steinhart and Wurtsbaugh 1995; Teuscher et al. 1995), which make poor prey items for rainbow trout.

Predation

Approximately 13% of all the rainbow trout stomachs contained fish prey. Redside shiners dominated the fish fraction of the rainbow trout diet (Table 1; Figure 3). Only two rainbow trout stomachs contained *O. nerka*. Both rainbow trout containing *O. nerka* prey were collected on 8 August, and in the same net. Interestingly, that net was located along the same shore as the sockeye release site (net D; Figure 1). Flesh color and length of the *O. nerka* pery matched hatchery sockeye released 12 days earlier.

Although *O. nerka* were observed in only 2 of 119 (1.7%) stomach samples, extrapolation to the whole predator population resulted in surprisingly high consumption estimates. Over a 20 d model period, rainbow trout consumed between 5,690 and 12,510 g of *O. nerka* prey (Table 2). Because kokanee were spatially separated from rainbow trout (Appendix A) and the color of flesh and fish size matched released sockeye, it is not unreasonable to assume that all of the predation was on listed fish. Therefore, the equivalent consumption of hatchery sockeye (mean weight of 7.4 g) ranged from 770 to 1,691 fish, or a total loss of 9 to 20% of the sockeye population.

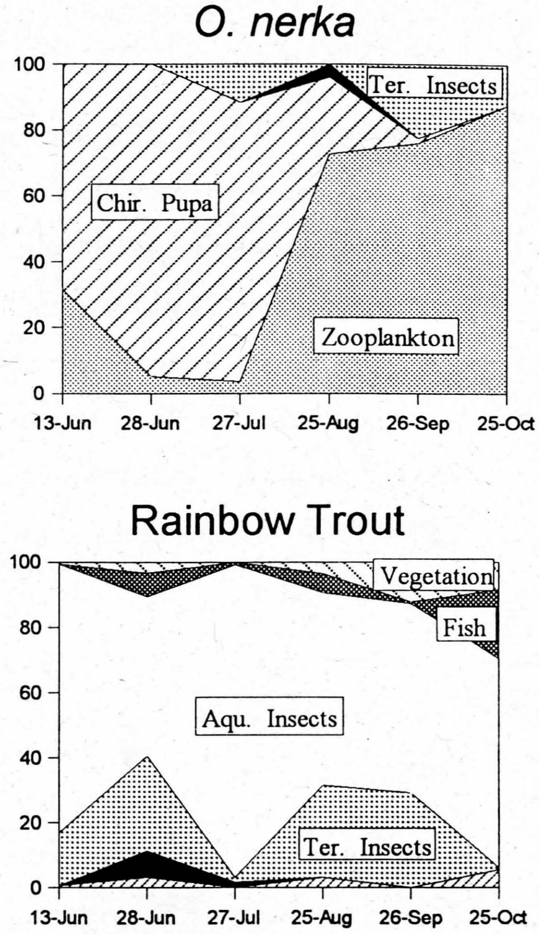


Figure ?. Seasonal diet results from *O. nerka* and hatchery rainbow trout in Pettit Lake. Diet overlap rainged from 0 to 22% (seeAppendix B).

Table 2. Consumption of *O. nerka* by rainbow trout as predicted by the bioenergetics model. Start day in the model was 27 July 1995. Consumption results are cumulative.

Day	Temp. (°C)	Hrb wt (g)	Hrb pop$_{low}$	Hrb pop$_{high}$	*O. nerka* cons$_{low}$ (g)	*O. nerka* cons$_{high}$(g)
0	12.9	203.0	852	1,874	0.0	0.0
1	12.9	203.0	850	1,869	281	618
4	13.0	203.1	843	1,855	1,125	2,475
8	13.3	203.3	834	1,836	2,257	4,965
12	13.6	203.4	826	1,817	3,396	7,470
16	13.8	203.3	817	1,799	4,541	9,988
20	14.1	203.1	809	1,780	**5,690**	**12,510**

Overwinter Survival

Approximately 39% of the rainbow trout caught in gillnets had survived at least one winter in Pettit Lake (Appendix B). Age-2 fish that survived one winter, comprised 64% of the total number of holdovers. Those fish had an estimated overwinter survival of between 10.2 to 22.5%. Age-3 trout that had survived two winters, comprised 25% of the total number of holdovers. Those fish had an estimated two year overwinter survival of between 4.6 and 10.0%. Age-4 and older rainbow trout accounted for 4% of the total rainbow trout population and 11% of the holdovers (Figure 2). The oldest rainbow trout caught in 1995 was age-5, with a fork length of 445 mm and weighed about 1,200 g.

CONCLUSIONS AND RECOMMENDATIONS

- Hatchery rainbow trout do not compete for food with *O. nerka* in the Sawtooth Valley Lakes. Monthly results from Pettit Lake corroborate earlier findings of no significant diet overlap between rainbow trout and *O. nerka* (Schrader 1991; Teuscher et al. 1995).

- Hatchery rainbow trout successfully overwinter in Pettit Lake. Estimated overwinter survival of age-2 rainbow trout was between 10 and 23%. Holdovers made up about 39% of the total rainbow trout population. These estimates, however, depend on age analysis, which is not 100% accurate. To eliminate errors associated with age analysis, hatchery rainbow trout should be fin clipped prior to release.

- Rainbow trout predation on *O. nerka* contradicts findings from Schrader (1991) and Teuscher et al. (1994). Not a single *O. nerka* was found in the diet of 255 rainbow stomachs previously analyzed (Teuscher et al. 1994; Schrader 1991). The contradicting results can be explained by the inclusion of hatchery sockeye. In the past, hatchery rainbow trout had access to only wild *O. nerka* which are likely better at avoiding predators and exhibit spatial segregation for most of the year (Appendix A). Conversely, hatchery reared sockeye may become disoriented and fail to seek cover. Additionally, releasing sockeye in the littoral zone forced spatial overlap that was not occurring in the wild populations. In conclusion, releasing hatchery sockeye to the pelagic zone may have reduced or eliminated losses to rainbow trout predators.

REFERENCES

Beauchamp, D. A. 1990. Seasonal and diet food habits of rainbow trout stocked as juveniles in Lake Washington. Transaction of the American Fisheries Society 119:475482.

Bevan D., and six coauthors. 1994. Snake River Salmon Recovery Team: Final Recommendations to National Marine Fisheries Service.

Hewett, S. W., and B. L. Johnson. 1992. Fish bioenergetics model 2: an upgrade of a generalized bioenergetics model of fish growth for microcomputers. University of Wisconsin, Sea Grant Institute, Technical Report WIS-SG-92-250, Madison.

Miller, R.B. 1953. Comparative survival of wild and hatchery-reared cutthroat trout in a stream. Transactions of the American Fisheries Society 83:120-130.

Miller, R.B. 1958. The role of competition in the mortality of hatchery trout. Journal of the fisheries Research Board of Canada 15:27-45.

O'Neill, S. M., and K. D. Hyatt. 1987. An experimental study of competition for food between sockeye salmon and threespine sticklebacks in a British Columbia coastal lake. Canadian Special Publication of Fisheries and Aquatic Sciences 96. 142-160.

Parkinson, E.A., J.M.B. Hume and Dolighan. 1989. Size selective predation by rainbow trout on two lacustrine (Oncorhynchus nerka) populations. Fisheries Management Report No. 94, British Columbia Fisheries Branch, Canada.

Reingold, M., and J. Davis. 1987. Regional Fishery Management Investigations. Job Performance Report. Project F-71-R11, Job No. 6, Salmon Subregion mountain lake investigations. Idaho Department of Fish and Game.

Schmitten, R., W., and Stelle, R.P. Jones. 1995. Proposed Recovery Plan for Snake River Salmon.

Schrader W. C. 1991. Assessment of potential interactions between catchable rainbow trout and sockeye salmon relative to the federal endangered species act. Idaho Department of Fish and Game. Fisheries Report, Boise.

Steinhart G. B., and W.A. Wurtsbaugh. 1995. Winter Limnology and Ecology of Kokanee Salmon, Oncorhynchus nerka, in the Sawtooth Valley Lakes. In Teuscher et al. 1995. Snake River salmon habitat and limnological research.

Teuscher, D., and six coauthors. 1994. Snake River sockeye salmon habitat and limnological research. Report (Contract DE-BI79-91BP22548) to Bonneville Power Administration, Portland, Oregon.

APPENDIX - A

Gillnet results by number and catch rate (# / gillnet hr.).

	rainbow		*O. nerka*	
Date	littoral	pelagic	littoral	pelagic
13 June	8 (0.133)	-	7 (0.116)	-
28 June	28 (0.528)	0 (0.000)	24 (0.452)	11 (0.209)
27 July	4 (0.088)	0 (0.000)	3 (0.066)	7 (0.121)
8 Aug	19 (0.371)	-	0 (0.000)	-
25 Aug	17 (0.369)	0 (0.000)	0 (0.000)	6 (0.166)
13 Sep	19 (0.395)	-	0 (0.000)	-
27 Sep	8 (0.137)	0 (0.000)	0 (0.000)	6 (0.107)
12 Oct	8 (0.137)	-	4 (0.068)	-
25 Oct	14 (0.223)	0 (0.000)	8 (0.133)	9 (0.147)
7 Nov	4 (0.062)	0 (0.000)	4 (0.062)	2 (0.125)
10 Jan	2 (0.028)	-	50 (0.694)	-
22 Feb	0 (0.000)	0 (0.000)	5 (0.313)	12 (0.250)
21 Mar	0 (0.000)	0 (0.000)	2 (0.026)	22 (0.386)

Key for abbreviations in Appendix B on the following page.

ZOO = zooplankton
CP = chironomid pupa
CL = chironomid larva
TI = terrestrial insects
AI = aquatic insects
MOL = mollusks
PLT = aquatic vegetation
RSS = redside shiners
On = *O. nerka*

APPENDIX B

O. Nerka and rainbow trout diet analysis in Pettit Lake. Diet overlap (from Koenings et al. 1984) is shown in the far right column. Diet overlap was calculated only on days when both species were collected. Diet overlap can range from 0 to 1, with 1 being complete overlap. See Appendix A for prey abbreviations.

Date	N	FL (mm)	ZOO	CP	CL	TI	AI	MOL	PLT	RSS	On	DietOverlap
rainbow												
13 June	8	215-310	0.0	0.7	0.0	16.2	82.2	0.3	0.6	0.0	0.0	**0.01**
28 June	15	220-303	0.0	5.9	12.5	30.5	36.1	4.1	3.7	7.2	0.0	**0.10**
27 July	4	160-310	0.0	0.0	1.9	1.4	71.1	24.8	0.0	0.8	0.0	**0.00**
8 Aug	20	219-310	0.0	15.3	7.6	0.1	37.9	15.4	12.6	0.0	11.1	
25 Aug	17	255-315	0.0	3.3	0.0	28.2	50.2	8.9	3.7	5.7	0.0	**0.02**
13 Sep	19	181-328	0.0	5.4	0.0	63.2	21.8	9.2	0.0	0.4	0.0	
26 Sep	8	247-290	0.0	0.0	0.0	29.2	46.3	11.7	12.3	0.5	0.0	**0.22**
12 Oct	8	242-329	0.0	28.5	0.0	10.5	43.3	0.0	17.7	0.0	0.0	
25 Oct	14	270-365	0.0	5.6	0.0	0.6	52.3	11.8	8.5	21.1	0.0	**0.00**
7 Nov	4	270-445	0.0	5.6	0.0	22.9	0.0	0.0	43.7	27.8	0.0	
10 Jan	2	286-299	0.0	0.0	0.0	0.0	58.2	26.1	15.7	0.0	0.0	
O. nerka												
13 June	7	159-182	31.7	68.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	
28 June	18	95-275	5.1	94.9	0.0	0.0	0.0	0.0	0.0	0.0	0.0	
27 July	7	-	3.7	84.7	0.0	11.6	0.0	0.0	0.0	0.0	0.0	
25 Aug	6	152-202	72.7	23.2	4.1	0.0	0.0	0.0	0.0	0.0	0.0	
26 Sep	19	105-217	75.9	1.5	0.0	22.6	0.0	0.0	0.0	0.0	0.0	
25 Oct	8	165-295	87.5	0.0	0.0	12.5	0.0	0.0	0.0	0.0	0.0	
22 Feb	10	113-180	100	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	

PART II

LIMNOLOGY OF SAWTOOTH LAKES - 1995: EFFECTS OF WINTER LIMNOLOGY AND LAKE FERTILIZATION ON POTENTIAL PRODUCTION OF SNAKE RIVER SOCKEYE SALMON

Chris Luecke

Wayne A. Wurtsbaugh

Phaedra Budy

Geoffrey B. Steinhart

Michael Slater

DEPARTMENT OF FISHERIES AND WILDLIFE
UTAH STATE UNIVERSITY

EXECUTIVE SUMMARY

In this section we describe the results of limnological sampling conducted on Redfish, Alturas, Pettit and Stanley Lakes from October 1994 through October 1995. We also describe winter limnological conditions and kokanee growth characteristics from 1993 to 1995 in Chapter 1. In Chapter 2 we describe limnological conditions during the spring, summer and fall of 1995, and compared characteristics among the four lakes. We also describe the fertilization of Redfish Lake in 1995. In Chapter 3 we compare the limnological characteristics of the four lakes from 1992-1995, emphasizing effects of fertilization of Redfish Lake in 1995 and effects of annual variation in planktivorous fish abundance among the lakes.

Winter limnological conditions varied among the lakes. In both the winters of 1993-94 and 1994-95 zooplankton biomass was reduced during winter months in Redfish, Pettit and Stanley Lakes compared to summer conditions. Low summer abundances of zooplankton were maintained during the winter in Alturas Lake. In all four lakes zooplankton species composition and kokanee diets shifted from a predominance of cladocerans in summer to copepods during the winter. The decline in the quantity and quality of zooplankton food resources had negative consequences on kokanee energy reserves in Alturas and Stanley Lakes. During both winters in both lakes, kokanee lipid levels decreased throughout the winter to levels of approximately 5% of dry body weight by May. These low lipid levels have been shown to decrease the survival of salmonids migrating through the Columbia River system to reach the Pacific Ocean. Winter limnological conditions in Sawtooth Valley Lakes may have similar consequences for the growth and survival of brood stock progeny of Snake River sockeye salmon, and suggest that carrying capacity of the lakes may be controlled by food availability in the winter.

Redfish Lake received weekly fertilizer additions of approximately 14 kg phosphorus and 259 kg nitrogen from early June through mid-October 1995. Despite the fertilization, limnological analyses of the four Sawtooth Valley Lakes indicated that water transparency and epilimnetic chlorophyll concentrations were similar in Redfish, Alturas, and Pettit Lakes. Stanley Lake had lower transparency and higher chlorophyll concentrations compared to the other lakes. Rates of primary production and algal biomass were higher in Redfish Lake compared to the other lakes. Zooplankton biomass was low in all the lakes in June and July. In August the zooplankton biomass of Stanley and Redfish Lakes increase substantially, whereas crustacean zooplankton biomass in Alturas and Pettit Lakes remained low. The increased zooplankton biomass in Redfish and Stanley Lakes was due to increases in the biomass of *Daphnia*.

Comparison of limnological conditions in the four lakes from 1992-1995 revealed several patterns. The cool, wet years of 1993 and 1995 tended to increase concentrations of chlorophyll and decrease abundance of crustacean zooplankton in all the lakes. The fertilization of Redfish Lake in 1995 appeared to increase rates of primary production and biomass of zooplankton compared to previous years. These increases occurred in spite of similar levels of chlorophyll among years. The emergence of *Daphnia pulex* in Redfish Lake in 1995 was unusual and occurred in conjunction with the lake fertilization. Multivariate assessments of limnological conditions among the lakes and among years indicated that lake differences were greater than year differences during the period of our study. The one exception to this statement was the response of Pettit Lake in 1995. Pettit Lake in previous years was grouped with Stanley Lake based on similarities in zooplankton abundance and species composition. Reductions in zooplankton and especially daphnid abundance in Pettit Lake in 1995 caused it to be grouped with Alturas Lake. Analysis of our data suggested that the decrease in Pettit zooplankton was due to increased abundance of planktivorous fish.

In summary, results from our analyses indicated that winter conditions in the Sawtooth Valley Lakes were severely limiting for juvenile salmonids. Lake fertilization slightly enhanced the growth potential of these lakes for juvenile salmonids. The lack of deleterious effects on water quality suggested that increased levels of fertilization could be attempted. Limnological conditions in these lakes appears to vary more in response to planktivorous fish abundance than to variation in nutrient income, suggesting that efforts to reduce kokanee and to maintain densities of stocked sockeye salmon below critical levels are paramount.

CHAPTER 3

OVER-WINTER ECOLOGY OF ONCORHYNCHUS NERKA IN THE SAWTOOTH VALLEY LAKES

Geoffrey B. Steinhart
Wayne A. Wurtsbaugh

INTRODUCTION

In 1991 the Snake River sockeye salmon (*Oncorhynchus nerka*) were listed as endangered species. This unique race of salmon represent the southernmost sockeye population and undergo one of the longest migrations of any salmon. In the 1800's, perhaps as many as 20,000 adult sockeye returned to spawn in Redfish Lake, located in the Sawtooth Valley of central Idaho (Evermann 1896). Since 1990, however, only 14 sockeye have returned to Redfish Lake. In an effort to restore this stock to historic populations, research began in 1991 to investigate the rearing lakes located in central Idaho. Most of this work has focused on the ice-free period in these lakes, but the lakes are frozen for 5-6 months of the year.

Winter has been shown to be an extremely harsh period for many salmonids (Cunjak and Power 1986, Cunjak and Power 1987, Metcalfe and Thorpe 1992, Prevost et al. 1992, Ruggerone 1992), yet little is known about the over-winter ecology of sockeye salmon. Unlike many warm-water species, sockeye are active at temperatures below 4° C (Brett 1983, Burgner 1992), so winter cannot be considered a period of dormancy. At 4° C, sockeye still must consume approximately 0.4% of their body weight per day in order to meet basic metabolic costs (Brett 1983, Brett et al. 1969), however, only 1.5 % of sockeye salmon captured during winter in Black Lake, Alaska, had consumed food (Ruggerone 1992). Ruggerone suggested that the lack of prey in sockeye stomachs was not due to inactivity of the sockeye, but was due to low zooplankton abundance during winter. However, in coastal Lake Washington, which does not freeze, juvenile sockeye were more active in late winter than in December, presumably due to declining energy reserves (Eggers 1978). The Sawtooth Valley Lakes also have much reduced winter zooplankton densities (Steinhart and Wurtsbaugh 1995), which could limit sockeye growth and survival.

After the long winter, Snake River sockeye salmon leave their nursery lakes in May, and migrate over 1400 Km to the ocean (Bjornn et al. 1968). Rondorf et al. (1985) found that migrating chinook salmon (*O. tshawytscha*) smolts lost all their fat reserves in only 370 river Km. The Snake River sockeye not only have a longer migration, but they often out-migrate prior to ice-out, so there is little time to build up fat reserves prior to smoltification. Traditionally, the rearing lakes produced smolts that were capable of surviving the migration, but the addition of eight hydroelectric dams along the Snake and Columbia Rivers has more than doubled the duration this journey. In addition, introduced predators now stalk the reservoirs behind the dams and prey heavily on juvenile salmon. As a result, smolt to adult survival has plummeted (Teuscher and Taki 1995). If sockeye are to survive their seaward migration, they must acquire or maintain adequate energy reserves during winter.

The purpose of this study was to examine the winter limnology of four historic rearing lakes in the Sawtooth Valley and to describe the winter ecology of *O. nerka*. Temperature, dissolved oxygen, and light, all of which can limit sockeye distribution or feeding (Ali 1959, Burgner 1992, Davis 1975, Ruggerone et al. 1991), were measured. Zooplankton populations were sampled throughout the winter to estimate food availability for juvenile *O. nerka*. Finally, changes in diet, growth, and lipid content of kokanee from two very different lakes were measured. Kokanee were used as a surrogate for endangered sockeye in this study, since juveniles are functionally similar in this system.

METHODS

Limnological Sampling

On each sampling date, we conducted limnological sampling near the deep section of the lake (See chapter 2 of this report for a complete description of limnological methods). During periods of ice-cover, a hole was cut with an ice auger or a chainsaw. Snow and ice thickness was measured to the nearest centimeter. Temperature and dissolved oxygen were measured using either a YSI Model 58 Dissolved Oxygen Meter® or a Hydrolab Surveyor III Meter®. Measurements were taken at approximately 1-m intervals to 10 m, and less frequently below 10 m.

Light intensity was measured using a submersible Li-Cor Model LI-188B® radiometer through hole approximately 0.75 m^2 cut by chainsaw. To avoid the increased light intensity around the hole, the spherical light sensor (measuring light between 400-700 nm) was extended approximately 3 m away from the hole using a plastic pipe with floatation (Figure 1). In addition, snow was piled back into the hole to reduce light penetration. The probe was lowered at 1 m intervals to 36 m, or until light was undetectable (<0.1 μE m^2 s^{-1}). A deck cell was used to correct for changes in surface intensity during the sampling period. Extinction coefficients were calculated as the slope of the regression of the natural log of % surface intensity against depth (Wetzel and Likens 1991).

Samples for chlorophyll *a* analysis were collected using a length of Tygon® tubing to obtain an integrated water sample from the surface to 6 m. In addition, a 4-L Van Dorn bottle was used to collect water from discrete depths throughout the lake for chlorophyll analysis. Two replicate 50-ml aliquots per sample were filtered through 0.45-μm cellulose acetate filters. They were subsequently frozen prior to being placed in 6 ml of 100% buffered methanol and extracted in the dark for 24 hr. The extracted samples were analyzed with a Turner model 111 fluorometer (Holm-Hansen and Riemann 1978).

Zooplankton samples were collected with a 80-μm mesh closing conical net from discrete depth strata: 10 m to surface, 30 m to 10 m (or bottom to 10 m in Stanley Lake), and bottom to 30 m. Samples were preserved in 10% neutralized formalin solution. Zooplankton were identified, enumerated and measured in the laboratory. Dry biomass was calculated by length-weight regressions from McCauley (1984) and Koenings et al. (1987).

Fish Sampling

Before the lakes were ice-covered and soon after ice-off, kokanee salmon were collected from Alturas and Stanley Lakes using a beam midwater trawl (3-m wide by 7-m deep). Trawling occurred during November in 1993 and 1994, and in May of 1994 and 1995. Sampling was done at or near the dark phase of the moon to standardize light conditions and trawling efficiency. In May of 1994 and 1995, kokanee in Stanley Lake were concentrated near the surface and mid-water trawling proved ineffective. In order to collect sufficient numbers of fish, horizontal and vertical gillnets were used.

Kokanee were collected under-ice with gillnets. Stanley Lake was sampled monthly from December through April, with the exception of February 1994 and 1995, and April 1995. Alturas Lake was sampled with gillnets in February 1994 and 1995. Both horizontal and vertical gillnets were used for sampling. During the winter of 1993-94, three sinking gillnets (three panels each, 22, 27, and 35 mm stretch mesh) and one floating gillnet (25, 32, 38, and 44 mm stretch mesh) were set under-ice by cutting a series of holes and passing the net from hole to hole with a long plastic pipe equipped with floatation. Beginning in March 1994, three single-paneled vertical gillnets 25 m deep by 3 m wide (19, 25, and 38 mm square mesh) were used instead of the horizontal gillnets. Vertical gillnets were set in a rectangular hole cut by chainsaw. In Stanley Lake, the vertical nets always touched the bottom of the lake, but in Alturas Lake, nets were sometimes set in water deeper than 25 m. Initially, gillnets were checked twice daily: once in the morning and once in the evening. However, since no fish were captured during the day, nets were subsequently checked only in the morning.

After capture, non-target fish species were weighed and measured, and returned to the lake whenever possible, whereas kokanee were kept in a cooler filled with water or ice until returning to the laboratory. Total lengths were recorded to the nearest millimeter. Fish were then weighed to the nearest 0.01 g on an Ohaus electric balance. During 1993-94, otoliths were carefully removed with a sharp scalpel for aging of individual fish. The fish were re-weighed to account for any lost tissue. Aging was done by the Shoshone-Bannock and Idaho Department of Fish and Game personnel.

Stomach lavage was used to collected the contents of the stomach, which were then preserved in 95% ethanol. Stomach contents were identified, enumerated and measured in the laboratory using a compound dissecting scope. The total weight of the stomach contents was calculated using zooplankton length-weight regressions (McCauley 1984, Koenings et al. 1987) and chironomid length-weight regressions (Wurtsbaugh and Hawkins 1990). Prey selection was examined using an electivity index that compares the proportions of zooplankton prey found in the diets with the proportions of zooplankton found in the lake (Ivlev

1961, Lechowicz 1982).

$$\mathbf{E} = r_i - p_i / r_i + p_i$$

where r_i is the proportion of prey found in the digestive tract and p_i is the proportion of that prey group in the lake. An electivity value of 0 means the prey type was eaten at the same proportion as found in the environment, while a positive value denotes selection for a prey item and a negative value means the prey are avoided. Electivities were calculated using frequencies of the different taxa in the whole water column. Since copepod nauplii were not found in kokanee diets, they were excluded from the electivity analyses.

Kokanee growth

To facilitate drying and fat extraction, fish were thawed and cut into several pieces using a scissors . Fish were dried for 24 hours at 60°C to determine dry weight. Drying tissue at 60°C will not oxidize fats (Kerr et al. 1982). Percent water was determined by subtracting the dry weight from the wet weight. The relative condition factor (Kn) was calculated using the equation:

$$Kn = W / aL^{b}$$

where the coefficients a and b are from the length-weight regression for this population ($a=5.9 \times 10^{-6}$, $b=3.03$).

Lipid content was measured using a Soxhlet lipid extraction (Bligh and Dyer 1959), using petroleum-ether as a solvent, since it does not dissolve structural fats (Dobush et al. 1985). After drying, each fish was put in a thimble which was then placed in the extraction apparatus. Dobush et al. (1985), found that 8 hr was sufficient to extract all the lipids from homogenized goose tissue; however, since these fish were not homogenized, they were extracted for 24 hr to insure a complete fat extraction. The thimble and extracted fish were then dried an additional 24 hr and re-weighed. The difference in weight before and after extraction was the weight of the extracted lipids.

RESULTS

Limonology

The lakes were inversely stratified during winter. Most of the water column was slightly below 4 °C, with colder temperatures above 10 m (Figure 2). Dissolved oxygen levels were high enough to support salmon (> 5 mg/L; Davis 1975) throughout most of the water column. In Redfish and Stanley Lakes, however, oxygen was < 5 mg/L close to the bottom. In Pettit and Alturas Lakes, the bottom 5 to 10 m were too low in oxygen to support salmon.

Ice and snow blocked much of the light from penetrating into the lake: up to 99% of the surface intensity was prevented from reaching the lake. The depth of the photic zone (1 % surface intensity) was less in winter than summer (Figure 3, Table 1). In general, Pettit and Stanley Lakes had the lowest light penetration, followed by Alturas and Redfish Lakes (Table 1). Redfish Lake had higher light penetration than the other lakes, likely due to a large fetch which resulted in less snow accumulation than the other lakes (Figure 3).

Epilimnetic (0 - 6 m) chlorophyll *a* levels were usually higher under-ice than during ice-free periods (Figure 4). Redfish and Pettit Lakes had the highest winter epilimnetic chlorophyll, with Alturas and Stanley Lakes slightly lower (Table 1). Other than in the top 6 m, chlorophyll concentrations were generally below 0.5 mg/m^3 during winter.

Zooplankton community dynamics varied among lakes and between years (Figure 5, Table 1). In Redfish Lake, winter zooplankton biomass was approximately one-quarter of summer biomass. In summer, the zooplankton community was dominated by *Daphnia* and *Holopedium*. During winter, however, *Bosmina* dominated the community in 1993-94, while calanoid copepods dominated in 1994-95. In Alturas Lake, winter zooplankton biomass was double summer biomass, with the dominant zooplankton being *Bosmina*. Of all the lakes, Pettit Lake had the highest biomass of zooplankton during winter. Its zooplankton community was dominated by cyclopoids in winter and by *Daphnia* in summer. Stanley Lake also had a relatively high zooplankton biomass. In summer, *Daphnia*, *Holopedium*, and calanoids composed most of the zooplankton community. During the winter of 1993-94, zooplankton biomass was one-tenth the summer biomass and was composed largely of calanoids. In the winter of 1994-95, however, zooplankton biomass was four to five-times higher than the previous winter and dominated by *Bosmina* and cyclopoids.

Kokanee diet and growth

In Stanley Lake, fish under 150 mm TL were generally age-0, and were used for all the analyses described below. In Alturas Lake, there was more overlap in TL of different age *O. nerka*, but generally fish under 100 mm TL were age-0. It was sometimes difficult to obtain an adequate sample of age-0 kokanee in Alturas Lake (Teuscher and Taki 1995). Therefore, some older fish (up to 150 mm TL) were used in analyses of condition factor, percent lipid, and diet content, but were excluded from the wet weight and dry weight analyses.

Kokanee diet was dominated by large prey items in summer, but during winter smaller, less favorable prey items were also eaten (Figure 6, Table 2). When large bodied organisms such as *Daphnia* and chironomid pupae were present in the lakes, they were important in kokanee diets. In fact, kokanee tended to have high electivities for *Daphnia* on all dates, except when *Daphnia* were only present at very low densities (Figures 7 and 8). In Stanley Lake, when *Daphnia* were not abundant, kokanee diets were composed of up to 75% copepods (Figure 6). Copepods' escape behavior makes them difficult to catch, and not a preferred food choice. Small-bodied *Bosmina* were also common in winter diets and generally had higher electivities in winter, when zooplankton were not abundant (Figure 7). Unlike kokanee in Stanley Lake, kokanee in Alturas Lake ate some benthic zooplankton, including *Alona* and ostracods. (Table 2). Most of their diet, however, was composed of *Bosmina* and cyclopoid copepods, with many chironomid pupae eaten in spring.

The mean weight of stomach contents in Stanley Lake kokanee was extremely low during winter, but most fish were consuming some prey (Figure 6). Other than the May samples, stomach contents were less than 0.2% dry weight of the fish. In May of 1995 did the weight of prey in the stomach exceed the maintenance ration for *O. nerka* at 4°C (0.4% dry weight of fish, Brett 1983, Brett et al. 1969). This increase in stomach contents coincided with a spring chironomid hatch, when large prey items were abundant. Since our data are from kokanee captured at dawn, we do not know the total amount of food eaten during a 24-hr period. However, the frequency of empty stomachs was extremely low during the sampling period (2 empty stomachs out of 98 sampled), suggesting that kokanee were feeding nearly continuously.

During the winter of 1993-94, juvenile kokanee in Stanley Lake did not show a significant increase in either wet weight or dry weight, however, in 1994-95, the wet weight of sampled kokanee increased significantly (Figure 9, Appendix 1). The observed increase, however, may be confounded by size-selective over-winter mortality (Henderson et al. 1988, Miranda and Hubbard 1994, Oliver and Holeton 1979, Post and Evans 1989, Toneys and Coble 1979). In 1993-94, the total lengths of captured kokanee did not increase much, and possibly a few smaller fish were lost to mortality (Figure 10). During the winter of 1994-95, however, the lengths of kokanee appear to steadily increase from November to May (Figure 11).

Condition factor and percent lipid, declined over-winter in both years (Figure 9). Condition factor in May was significantly lower than in November during 1994-95, but not in 1993-94 (Appendix 1). Kokanee entered the winter of 1993-94 with less lipid than in 1994-95 (7.7% versus 10.6%). By May in each year, lipid levels were significantly lower than in fall: 4.3% in 1994 and 4.9% in 1995. The rate of lipid loss was significantly greater in 1994-95 than in 1993-94 ($df = 178$, $t = 4.41$, $p < 0.0005$).

In Alturas Lake, kokanee salmon weight did not increase during the winter of 1993-94, but weights did increase over-winter in 1994-95 (Figure 12, Appendix 1). The sample size of confirmed age-0 kokanee was small during 1993-94 (November $n = 3$, May $n = 5$), so the growth estimates may not be accurate and could be biased by size-selective mortality. The total lengths of age-0+ kokanee increased in 1994-95 (Figure 13), indicating growth occurred, not size-selective mortality of smaller size-classes.

Relative condition factors of Alturas Lake kokanee increased significantly during both winters (Figure 12 and Appendix 1). Although kokanee in Alturas Lake began the winter with much lower lipid content (6%) than the Stanley kokanee, by May, lipid levels declined only slightly to 4.8% in 1994 and 5.1% in 1995 (Appendix 1). In 1993-94 this loss was significant ($p<0.05$) but in 1994-95 it was not. There was, however, no significant difference between years in the rate of lipid loss ($df = 62$, $t = 0.51$, $p > 0.25$).

There was a strong relationship between total length, percent water and percent lipid for kokanee. As % water increased in the fish, there was nearly a comenserate decrease in the % lipid (Figure 14). Fish total length had a small, yet significant, effect on lipid content. Using kokanee from Stanley and Alturas lakes, as well as kokanee from a Utah hatchery, the following multiple regression was highly significant ($r^2=0.93$; % water: $t = 62.8$, $p < 0.0001$; TL: $t = 11.6$, $p < 0.0001$): % Lipid = -0.86 (% Water) + -0.03(TL) + 74.86, N = 317.

DISCUSSION

Dissolved oxygen does not appear to be limiting to sockeye distribution or survival in the Sawtooth Valley Lakes. The deep, oligotrophic rearing lakes contain adequate dissolved oxygen to support salmon throughout the water column. Although the bottom of some of the lakes are anoxic, this should not limit sockeye survival as seen in some shallow, eutrophic lakes (Ruggerone 1992, 1993). In the presence of low oxygen concentrations, some fish still enter anoxic water to feed (Rahel and Nutzman 1994). The relatively small volume of the Sawtooth Valley Lakes that are below 5 mg/L should not limit sockeye feeding.

Smolt production may be limited by low zooplankton populations during winter, however. Low under-ice temperatures and light intensities may limit phytoplankton production. Low phytoplankton abundance can lower zooplankton production and biomass, which can lead to lower fish growth, escapement, and survival (Carpenter et al. 1990, Hyatt and Stockner 1985, Kyle et al. 1988, Riemann and Meyers 1992). In addition, low under-ice light levels inhibit feeding during many nights, and may reduce capture success for much of the day (Chapter 2).

Kokanee tended to favor large bodied prey in their diets: chironomids in spring, *Daphnia* in summer, and copepods in winter. During winter, however, low zooplankton biomass led to higher electivities than in summer for the small bodied *Bosmina*. Furthermore, kokanee showed higher electivities for copepods during winter than in summer. Because copepods are effective at avoiding predators, they were generally avoided during summer when alternative prey such as *Daphnia* were more abundant.

Electivities for copepods were high in winter, when low zooplankton abundance favored the consumption of any prey item encountered.

Kokanee gut fullness was relatively low during winter, but the incidence of empty stomachs (2 fish out of 98) was much lower than reported in other *O. nerka* systems (Eggers 1978, Ruggerone 1991, 1992). The low incidence of empty stomachs suggests that fish must feed continually, unlike more productive systems where sockeye may feed for only a short time, and then fast for several days to avoid predation (Eggers 1978). If after feeding, kokanee remain inactive while digesting, our results may be biased by the use of passive sampling gear which would tend to capture the more active, hungry fish. However, these results still show low gut fullness, and a switch to alternative prey, as shown by changes in electivities. Therefore, *O. nerka* may be food limited in these lakes during winter, and they appear to feed on whatever prey are available in order to prevent excessive lipid loss..

Even with current *O. nerka* populations in the Sawtooth Valley Lakes, the young salmon lose energy and can experience size-selective over-winter mortality during the harsh winter. In Stanley Lake, low zooplankton biomass in the winter of 1993-94 may have resulted in mortality of the smaller fish, while larger kokanee lost over 40% of their lipid content. With the higher zooplankton biomass in 1994-95, fish entered the winter with higher lipid levels, experienced less size-selective mortality, but still lost over 50% of their lipids. In Alturas Lake, *O. nerka* entered the winter much smaller, and with less lipid, than Stanley Lake kokanee. While loss of lipids was not as great in Alturas Lake as in Stanley Lake, fish in Alturas had less energy reserves to carry them through the winter and size-selective mortality may have played a role in the demographics of the population.

The long migration of these smolts makes it essential that they leave the nursery lakes physiologically prepared to migrate over 1400 km to the ocean. Before the addition of eight hydroelectric dams along their migration route, the smolts reached the ocean in 2 weeks. Now, slowed by the dams, young fish may take over 4 weeks to reach the ocean, and are vulnerable to native and introduced predators in impoundments created by damming of the rivers. Migrating chinook smolts in the Columbia River lose all their lipid stores after traveling only 370 Km (Rondorf et al. 1985). Assuming this loss is linear until reserves are exhausted and that the rate of loss in sockeye is comparable to chinook, fish migrating from the Sawtooth Valley Lakes would need a lipid content of 11.5% to avoid losing all their lipid during migration. During this study, the highest mean lipid content of kokanee in spring was only 5..1%: an insufficient amount to last for the entire downstream migration. In addition, Rondorf et al. (1985) found that increased initial lipid content led to higher downstream survival of chinook smolts: a 1% increase in starting lipid resulted in nearly an 8% increase in survival. Given these numbers, *O. nerka* in the Stanley and Alturas Lakes appear ill-prepared to undergo the long downstream migration to the ocean. Unfortunately, we were unable to obtain samples of outmigrant O. nerka from Redfish Lake to determine their lipid content. It is possible that sockeye will not leave the lake until they reach a certain lipid content. Future efforts should be made to understand the lipid dynamics of sockeye smolts leaving the Sawtooth Valley Lakes.

The stocking of endangered sockeye fry into the Sawtooth Valley Lakes will likely result in even greater over-winter mortality and/or lipid losses. Even under current conditions, *O. nerka* present in the lakes do not fare well over-winter. Stocking sockeye fry into several lakes is preferable to overstocking any one lake, so future releases should be considered in Stanley, and possibly Alturas Lakes. For best survival and growth, however, the growing conditions can be improved in the lakes. Currently, there are two strategies being implemented to accomplish this goal: 1) reducing existing kokanee populations to decrease competition for limited food, and 2) using whole-lake fertilization to increase productivity of the lakes (Hyatt and Stockner 1988). Both these strategies may prove beneficial to increase production, growth, and survival of endangered sockeye salmon.

In conclusion, the winter appears to be a period of low production in the Sawtooth Valley Lakes. As a result, *O. nerka* in the lakes are losing energy over winter, which can decrease downstream survival of this endangered species. Recovery efforts should focus on improving over-winter food resources, since under current conditions there would likely be low downstream survival of smolts due to inadequate energy reserves. The addition of several dams along the Snake and Columbia Rivers, however, has likely increased the energetic demand on migrating smolts since the migration takes longer and the fish must swim through slack-water reservoirs. Under pre-dam conditions, *O. nerka* leaving the Sawtooth Valley Lakes did not need as much energy during their migration, but now, we must boost production in the lakes to aid the fish in preparing for their journey to the ocean.

REFERENCES CITED

Ali, M.A. 1959. The ocular structure, retinomotor and photo-behavioral responses of juvenile pacific salmon. Can. J. Zool. **37**: 965-998.

Bjornn, T.C., D.R. Craddock, and D.R. Corley. 1968. Migration and survival of Redfish Lake, Idaho, sockeye salmon, *Oncorhynchus nerka*. Trans. Am. Fish. Soc. **97**: 360-373.

Bligh, E.G., and W.J. Dyer. 1959. A rapid method of total lipid extraction and purification. J. Biochem. Physiol. **37**: 911-917.

Brett, J.R. 1979. Environmental factors and growth. in *Fish physiology volume VIII: bioenergetics and growth*. W.S. Hoar, D.J. Randall, and J.R. Brett, Eds. Academic Press, London. pp. 599-675.

Brett, J.R. 1983. Life energetics of sockeye salmon, *Oncorhynchus nerka*. in *Behavioral energetics: the cost of survival in vertebrates.* I. Lustic, Eds.

Brett, J.R., J.E. Shelbourn, and C.T. Shoop. 1969. Growth rate and body composition of fingerling sockeye salmon, *Oncorhynchus nerka*, in relation to temperature and ration size. J. Fish. Res. Bd. Can. **26**: 2363-2394.

Burgner, R.L. 1992. Life history of sockeye salmon (*Oncorhynchus nerka*). in *Pacific salmon life histories.* C. Groot, L. Margolis, Eds. University of British Columbia Press, Vancouver: 1-117.

Carpenter, S.R., J.F. Kitchell, and J.R. Hodgson. 1985. Cacading trophic interactions and lake productivity. Bioscience **35**: 634-639.

Cunjak, R.A., and G. Power. 1986. Seasonal changes in the physiology of brook trout, *Salvelinus fontinalis* (Mitchill), in a sub-Arctic river system. J. Fish Biol. **29**: 279-288.

Cunjak, R.A., and G. Power. 1987. The feeding and energetics of stream-resident trout in winter. J. Fish Biol. **31**: 493-511.

Davis, J.C. 1975. Minimal dissolved oxygen requirements of aquatic life with emphasis on Canadian species: a review. J. Fish. Res. Bd. Can. **32**: 2295-2332.

Dobush, G.R., C.D. Ankney, and D.G. Krementz. 1985. The effect of apparatus, extraction time, and solvent type on lipid extractions of snow geese. Can. J. Zool. **63**: 1917-1920.

Eggers, D.M. 1978. Limnetic feeding behavior of juvenile sockeye salmon in Lake Washington and predator avoidance. Limnol. Oceanogr. **23**: 1114-1125.

Evermann, B.W. 1896. A report upon salmon investigations in the headwaters of the Columbia River, in the state if Idaho, in 1895, together with notes upon the fihes observed in that same state in 1894 and 1895. Bulletin of the United States Fisheries Commission **16**: 151-202.

Henderson, P.A., R.H.A. Holmes, and R.N. Bamber. 1988. Size-selective overwintering mortality in the sand smelt, *Atherina boyeri* Risso, and its role in population regulation. J. Fish Biol. **33**: 221-233.

Holm-Hansen, O., and B. Rieman. 1978. Chlorophyll a determination: improvements in methodology. Oikos **30**: 438-447.

Hyatt, K.D., and J.G. Stockner. 1985. Responses of sockeye salmon (*Oncorhynchus nerka*) to fertilization of British Columbia coastal lakes. Can. J. Fish. Aquat. Sci. **42**: 320-331.

Ivlev, Y.S. 1961. Experimental ecology of feeding fishes. Yale University Press. New Haven, CT. 302 p.

Kerr, D.C., C.D. Ankney, and J.S. Millar. 1982. The effect of drying temperature on extraction of petroleum ether soluble fats of small birds and mammals. Can. J. Zool. **60**: 470-472.

Koenings, J.P. J.E. Edmundson, G.B. Kyle, and J.M. Edmundson. 1987. Limnological field and laboratory manual: methods for assessing aquatic production. Alaska Department of Fish and Game. FRED Div. Rep. Ser. No. 71. Juneau, AK. 212 p.

Kyle, G.B., J.P. Koenings, and B.M. Barrett. 1988. Density-dependent, trophic level responses to an introduced run of sockeye salmon (*Oncorhynchus nerka*) at Frazer Lake, Kodiak Island, Alaska. Can. J. Fish. Aquat. Sci. **45**: 856-867.

Lechowicz, M.J. 1981. The sampling characteristics of electivity indices. Oecologia. **52**: 22-30.

McCauley, E. 1984. The estimation of the abundance and biomass of zooplankton in samples. *In*, J.A. Downing and F Rigler (eds.). A Manual on methods of secondary productivity in freshwaters, Second edition. Blackwell Scientific Publishing, Oxford, MA.

Metcalfe, N.B., and J.E. Thorpe. 1992. Anorexia and defended energy levels in over-wintering juvenile salmon. J. Anim. Ecol. **61**: 175-181.

Miranda, L.E., and W.D. Hubbard. 1994. Length-dependent winter survival and lipid composition of age-0 largemouth bass in Bay Springs Reservoir, Mississippi. Trans. Am. Fish. Soc. **123**: 80-87.

Oliver, J.D., and G.F. Holeton. 1979. Overwinter mortality of fingerling smallmouth bass in relation to size, relative energy stores, and environtmental temperature. Trans. Am. Fish. Soc. **108**: 130-136.

Post, J.R., and D.O. Evans. 1989. Size-dependent overwinter mortality of young-of-the-year yellow perch (Perca flavescens): laboratory, in situ enclosure, and field experiments. Can. J. Fish. Aquat. Sci. **46**: 1958-1968.

Prevost, E., E.M.P. Chadwick, and R.R. Claytor. 1992. Influence of size, winter duration and density on sexual maturation of Atlantic salmon (*Salmo salar*) juveniles in Little Codroy River (southwest Newfoundland). J. Fish Biol. **41**: 1013-1019.

Rahel, F.J., and J.W. Nutzman. 1994. Foraging in a lethal environment: fish predation in hypoxic waters of a stratified lake. Ecology **75**: 1246-1253.

Rieman, B.E., and D.L. Meyers. 1992. Influence of fish density and relative productivity on growth of kokanee in ten oligotrophic lakes and reservoirs in Idaho. Trans. Am. Fish. Soc. **121**: 178-191.

Rondorf, D.W., M.S. Dutchuk, A.S. Kolok, and M.L. Gross. 1985. Bioenergetics of juvenile salmon during the spring outmigration. U.S. Fish and Wildlife Service.

Ruggerone, G.T., D. Helton, and D.E. Rogers. 1991. Potential factors influencing the large annual fluctuations of adult sockeye salmon returning to Black Lake, Alaska. Fisheries Research Institute, University of Washington. FRI-UW-9117.

Ruggerone, G.T. 1992. Winter ecology of sockeye salmon in the Chignik Lakes, Alaska. Fisheries Research Institute, University of Washington. FRI-UW-9214.

Ruggerone, G.T. 1993. Winter investigations of salmon in the Chignik Lakes, Alaska, during 1993. Natural Resources Consultants, Inc.

Steinhart, G.B., and W.A. Wurtsbaugh. 1995. Winter limnology and the ecology of kokanee salmon, *Oncorynchus nerka*, in the Sawtooth Valley Lakes. *In*, D. Teuscher and D. Taki (eds.). Snake River sockeye salmon habitat and limnological research. U.S. Department of Energy, Bonneville Power Administration, Portland, OR. Project number 91-71. pp. 77-95.

Teuscher, D., and D. Taki. 1995. Snake River sockeye salmon habitat and limnological research. *In*, D. Teuscher and D. Taki (eds.). Snake River sockeye salmon habitat and limnological research. U.S. Department of Energy, Bonneville Power Administration, Portland, OR. Project number 91-71. pp. 1-50.

Toneys, M.L., and D.W. Coble. 1979. Size-related, first winter mortality of freshwater fishes. Trans. Am. Fish. Soc. **108**: 415-419.

Wetzel, R.G., and G.E. Likens. 1991. *Limnological Analyses*. Springer-Verlag, New York. 391 p.

Wurtsbaugh, W.A. and C. Hawkins. 1990. Trophic interactions between fish and invertebrates in Bear Lake, Utah-Idaho. Special Publication of the Ecology Center. Utah State University, Logan, UT.

Table 1. Mean summer and winter light penetration, epilimnetic (0-6 m) chlorophyll *a* concentrations, and crustacean zooplankton biomass for the Sawtooth Valley Lakes during 1993-1995.

Lake	Mean Depth of 1% Surface Light Intensity (m)		Mean Epilimnetic Chlorophyll *a* (µg/L)		Mean Crustacean Zooplankton Biomass (µg/L)	
	Ice-free	Ice	Ice-free	Ice	Ice-free	Ice
Redfish	28.0	8.6	0.5	1.1	6.8	1.8
Alturas	21.9	5.4	0.7	1.0	1.8	3.5
Pettit	26.7	4.1	0.5	1.1	21.3	7.4
Stanley	14.4	4.5	0.9	0.8	18.9	6.1

Table 2. Diet compostion (percent by number) of kokanee captured in Alturas Lake from 1993-95. T= trace amount found in diet (<1%). Diet codes: DAP = *Daphnia*, HOL = *Holopedium*, BOS = *Bosmina*, ALO = *Alona*, CYC = Cyclopoid, OST = Ostracode, POL = *Polyphemus*, LAR = Chironomid larvae, PUP = Chirnomid pupae, TER = terrestrial insect. June 1994 data from Teuscher and Taki (1995).

Date	N	DAP	HOL	BOS	ALO	CYC	OST	POL	LAR	PUP	TER
Nov 93	10	2	T	51	T	42	4	0	T	0	T
May 94	9	T	T	34	T	65	0	T	0	0	0
Jun 94		2	0	35	0	0	0	0	2	58	5
Nov 94	10	T	0	T	4	83	0	0	9	0	4
Feb 95	2	0	0	28	0	70	0	0	2	0	0
May 95	6	0	0	13	0	1	0	0	19	68	0

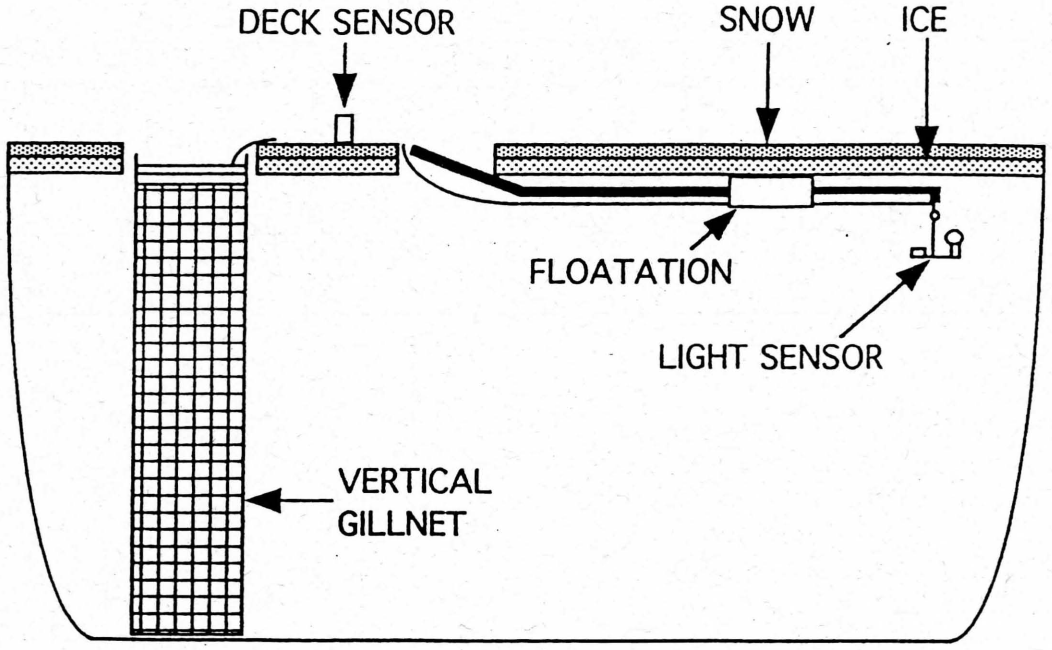


Figure 1. Diagram of methods used during periods of ice-cover on the Sawtooth Valley Lakes. Holes were cut using a chainsaw or an auger. Note the construction of the flexible, plastic pole used to extend the underwater sensor away from the light surrounding the cut hole, but keeping the sensor near the surface of the ice.

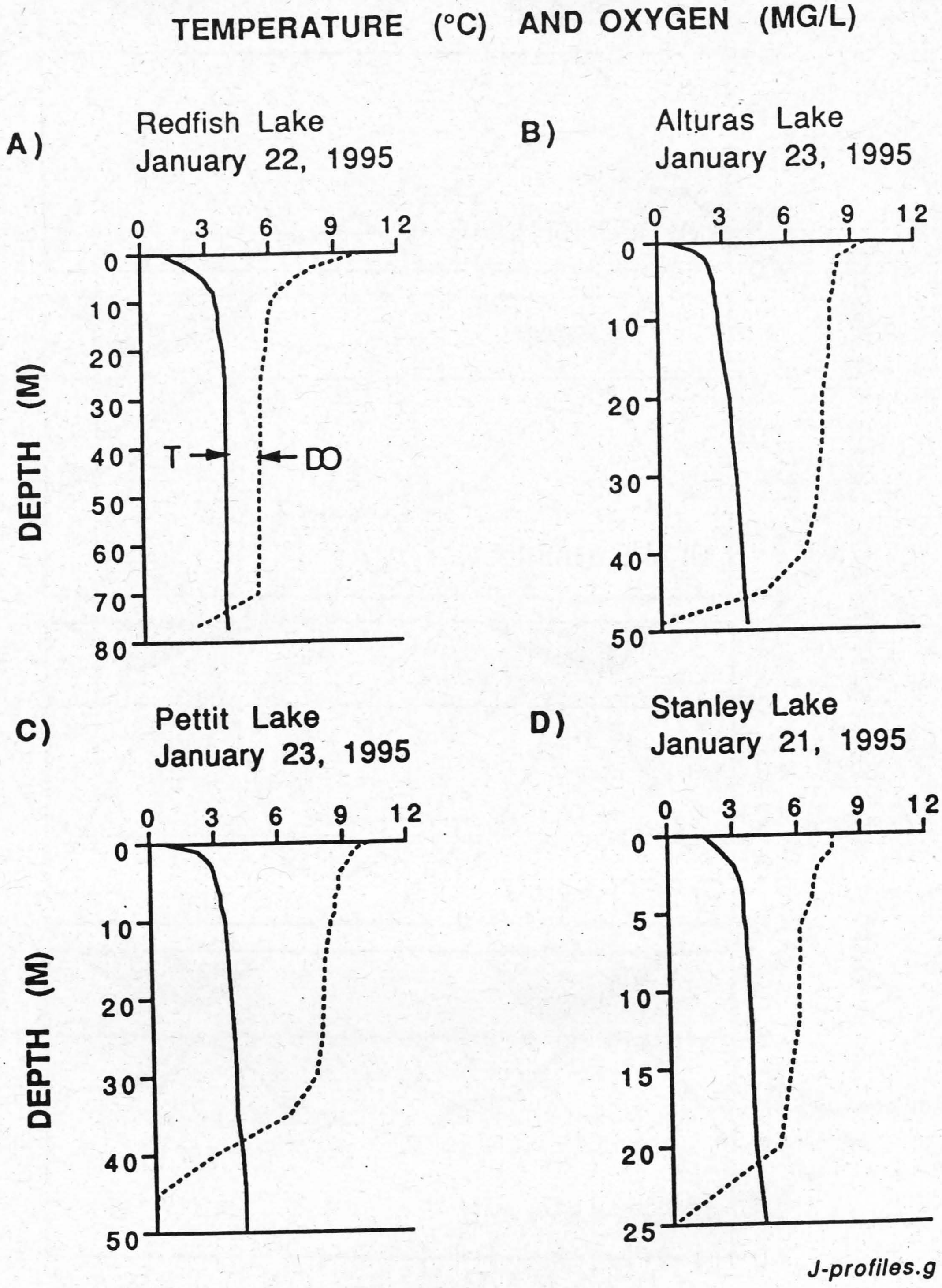


Figure 2. Temperature (T) and oxygen concentration (DO) vertical profiles of four Sawtooth Valley Lakes: A) Redfish, B) Alturas, C) Pettit, and D) Stanley Lakes. Data were collected through the ice in January 1995.

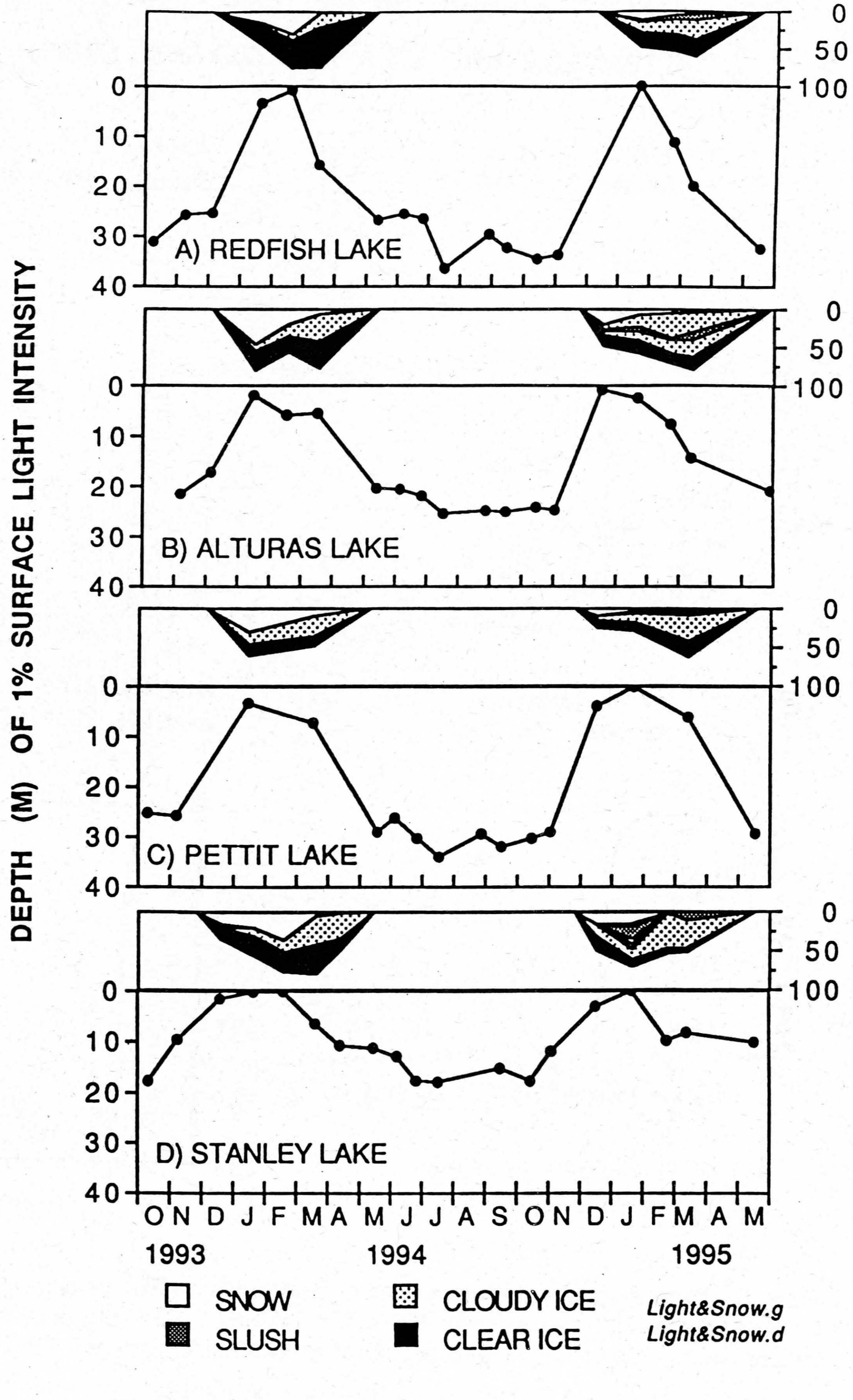
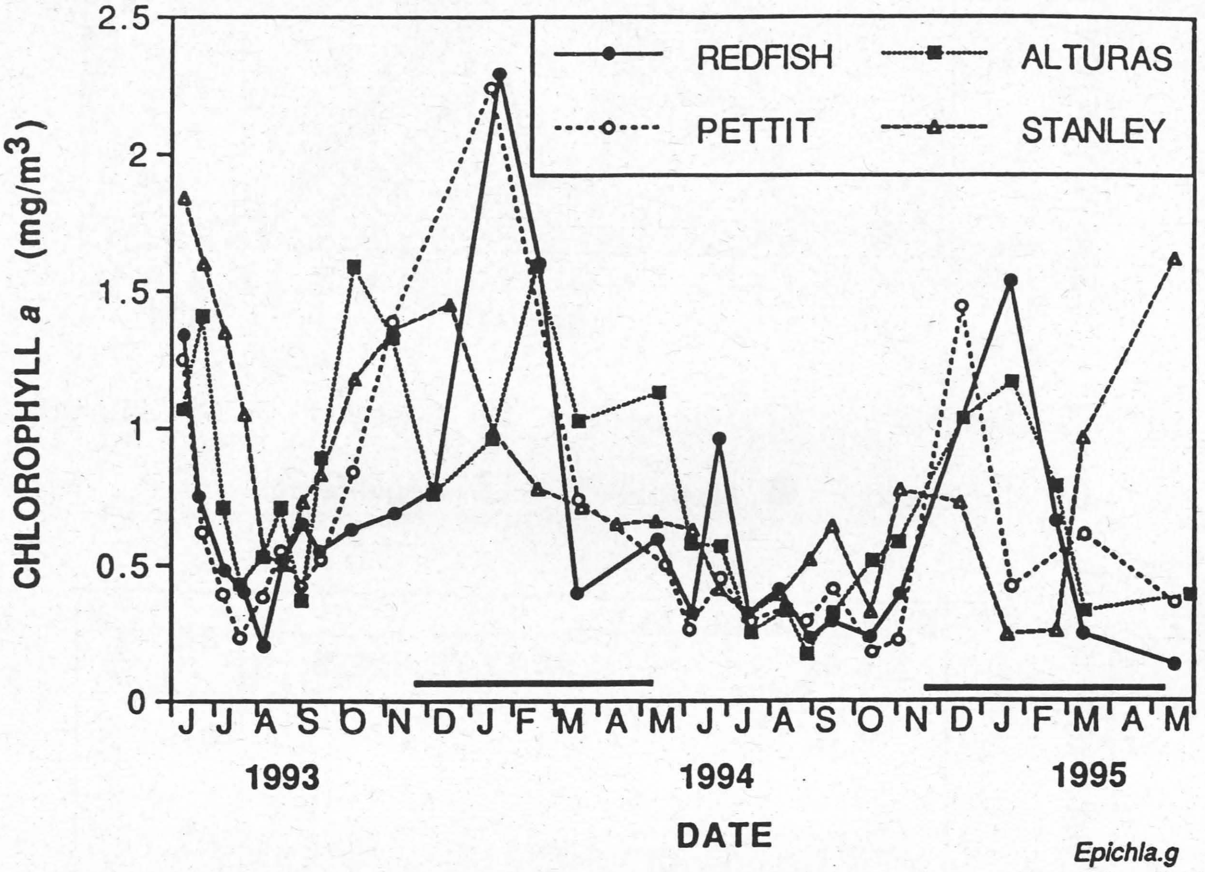
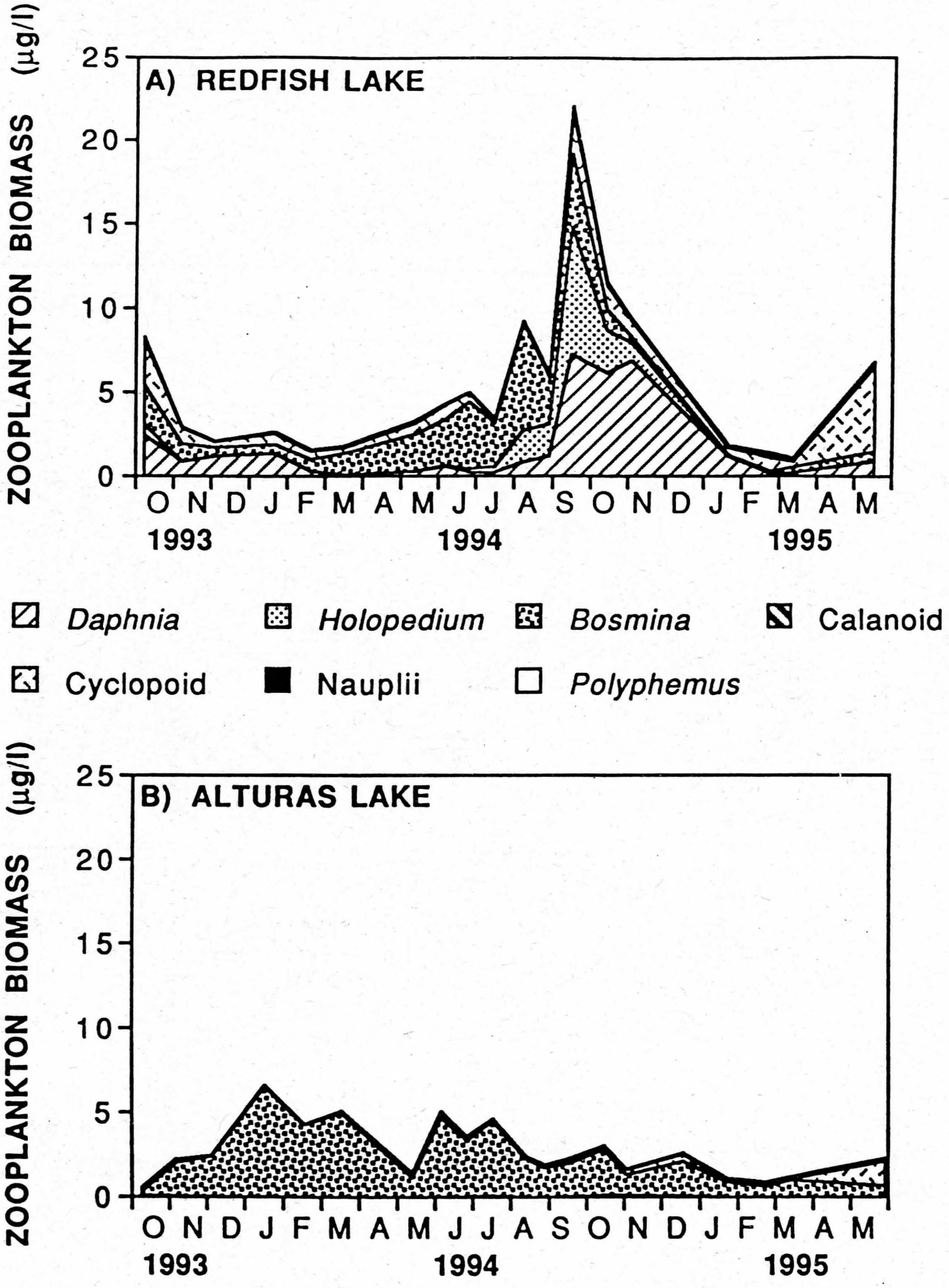


Figure 3. Depth of 1% surface light intensity and thickness of snow, slush and ice for: A) Redfish, B) Alturas, C) Pettit, and D) Stanley Lakes in 1993-95.



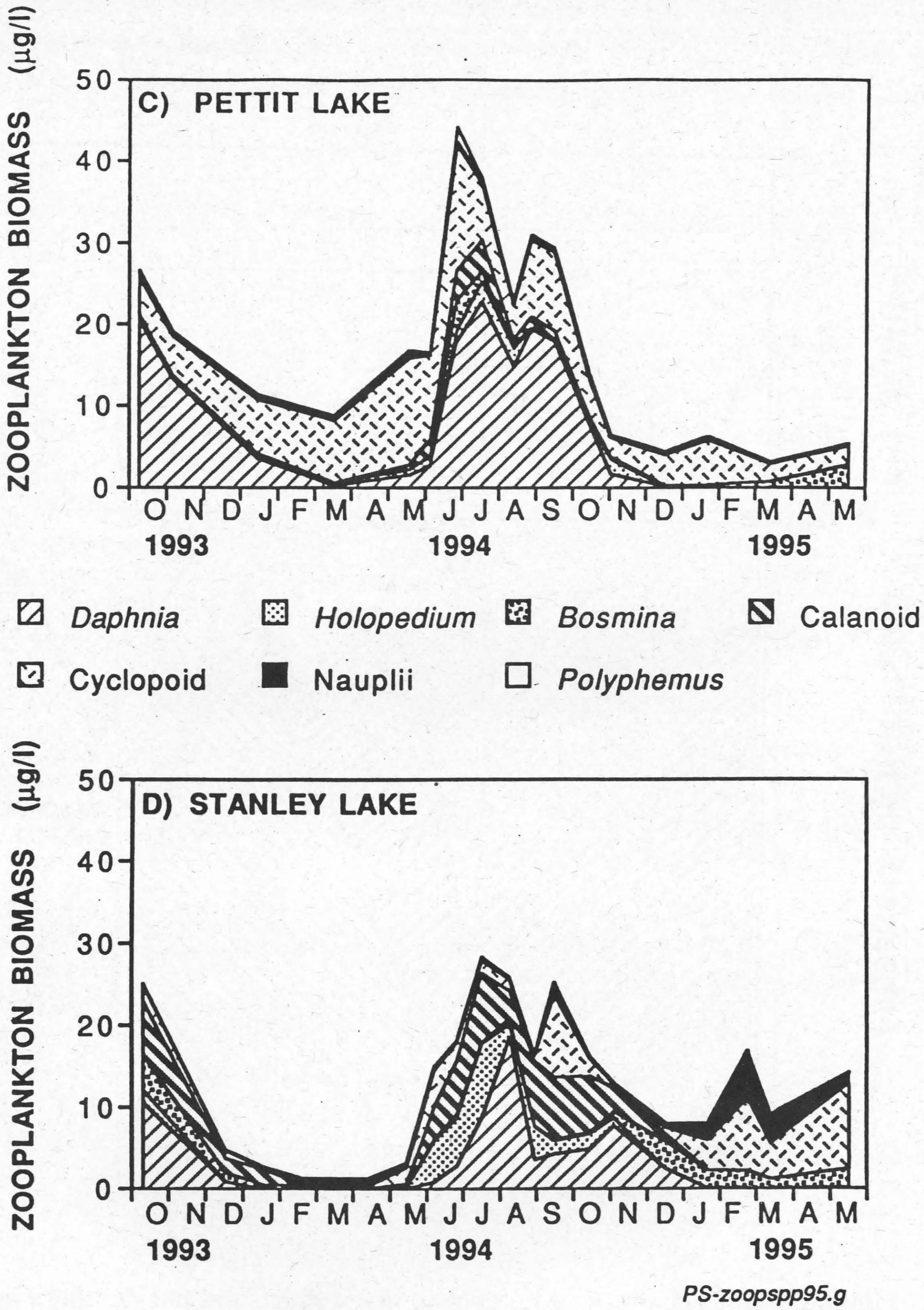
Epichla.g
0-6 all.1993-95

Figure 4. Epilimnetic (0-6 m) chlorophyll concentrations for the Sawtooth Valley Lakes during 1993-95. The bars denote periods of ice-cover.



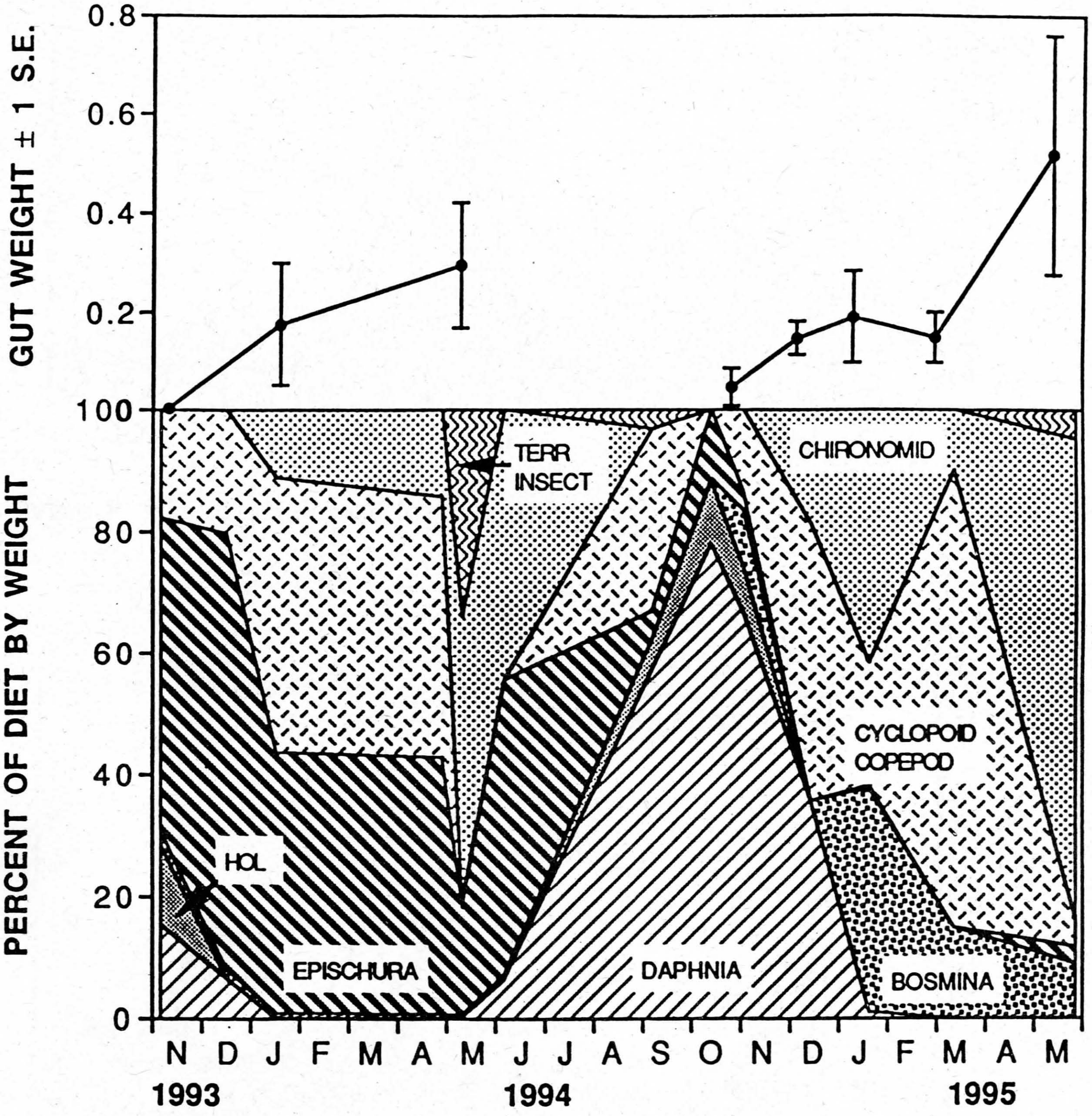
RA-zoopspp95.g
Zoopspp1993-95.d

Figure 5. Crustacean zooplankton biomass for the Sawtooth Valley Lakes during 1993-95. A) Redfish, B) Alturas, C) Pettit, and D) Stanley Lakes.



PS-zoopspp95.g
Zoopspp1993-95.d

Figure 5. Continued.



S-dietw.g
S-diet.dat

Figure 6. Gut weight, as a percent of dry weight, and gut contents of kokanee salmon captured in Stanley Lake during 1993-95. Note that the maximum consumption of a 10 g sockeye in one feeding is approximately 9% (Brett 1979). Summer diet data are from Teuscher and Taki (1995).

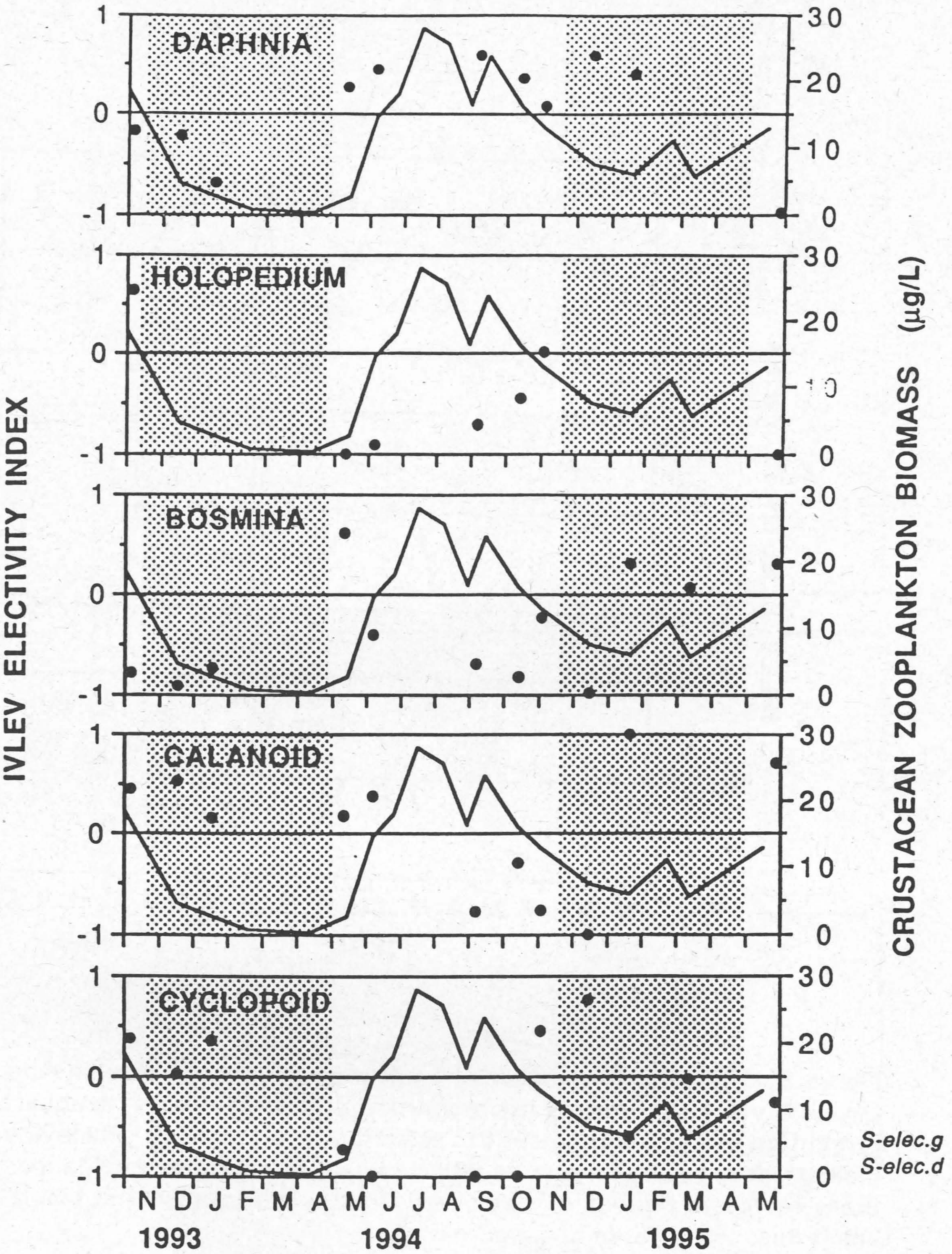
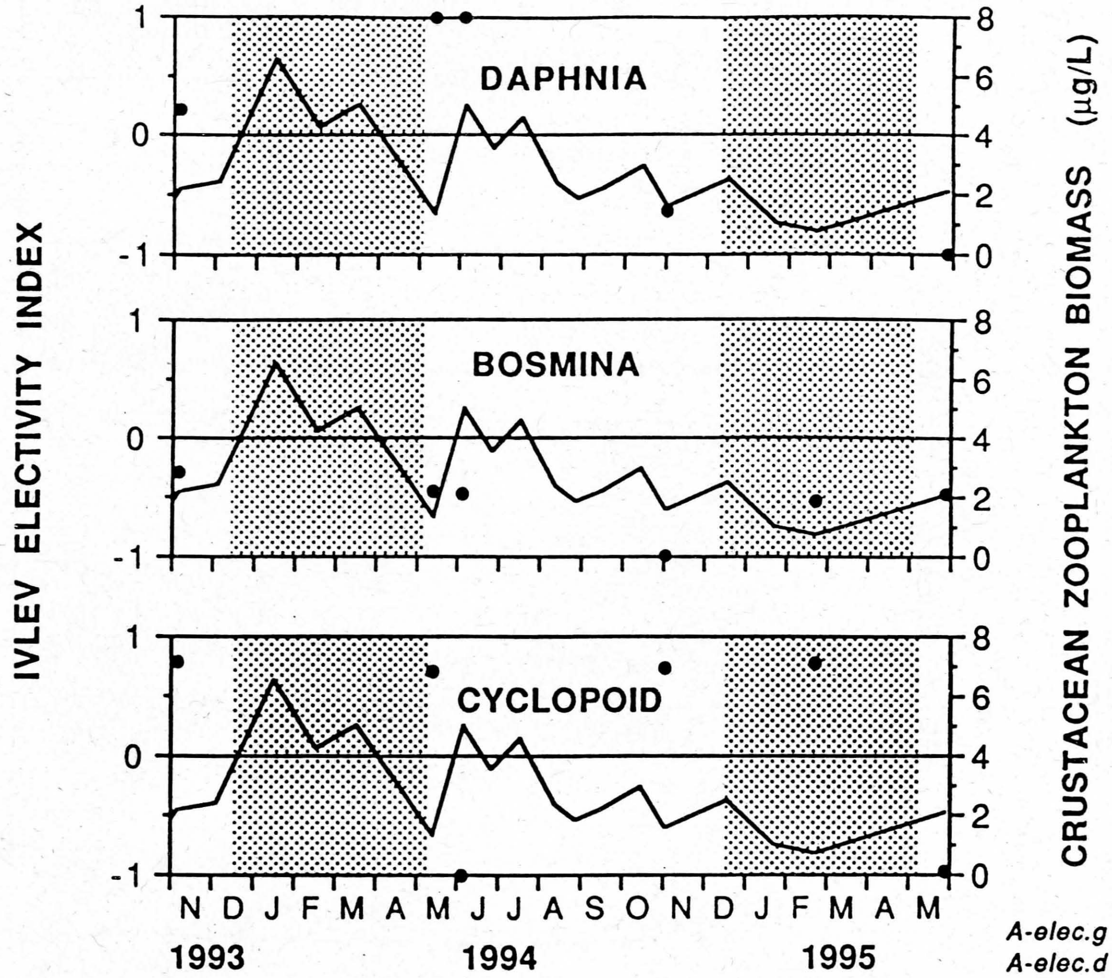


Figure 7. Ivlev electivity indices (•) for prey in kokanee diets in Stanley Lake. A value of 0 means the proportion of prey in the diet is equal to the proportion in the environment (no selection or avoidance). Positive values indicate prey selection, and negative values indicate prey avoidance. Shaded areas represent periods of ice-cover. Crustacean zooplankton biomass (solid lines) does not include copepod nauplii.



A-elec.g
A-elec.d

Figure 8. Ivlev electivity indices (•) for prey in kokanee diets in Alturas Lake. A value of 0 means the proportion of prey in the diet is equal to the proportion in the environment (no selection or avoidance). Positive values indicate prey selection, and negative values indicate prey avoidance. Shaded areas represent periods of ice-cover. Crustacean zooplankton biomass (solid lines) does not include copepod nauplii.

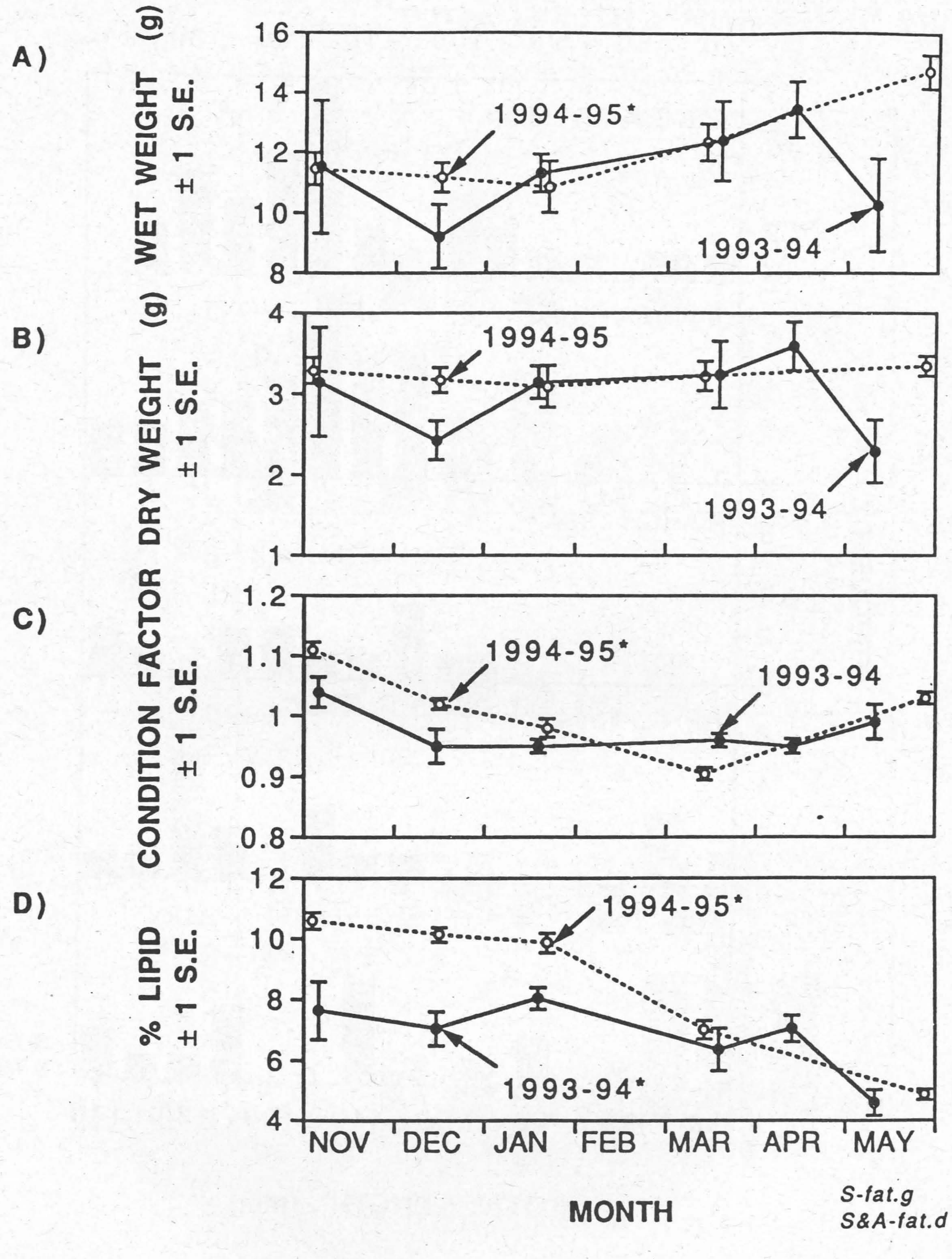


Figure 9. Over-winter changes in A) wet weight, B) dry weight, C) condition factor, and D) lipid (as a percent of dry weight) for age-0 kokanee from Stanley Lake. The asterisks indicate significant differences between November and May samples ($p < 0.05$).

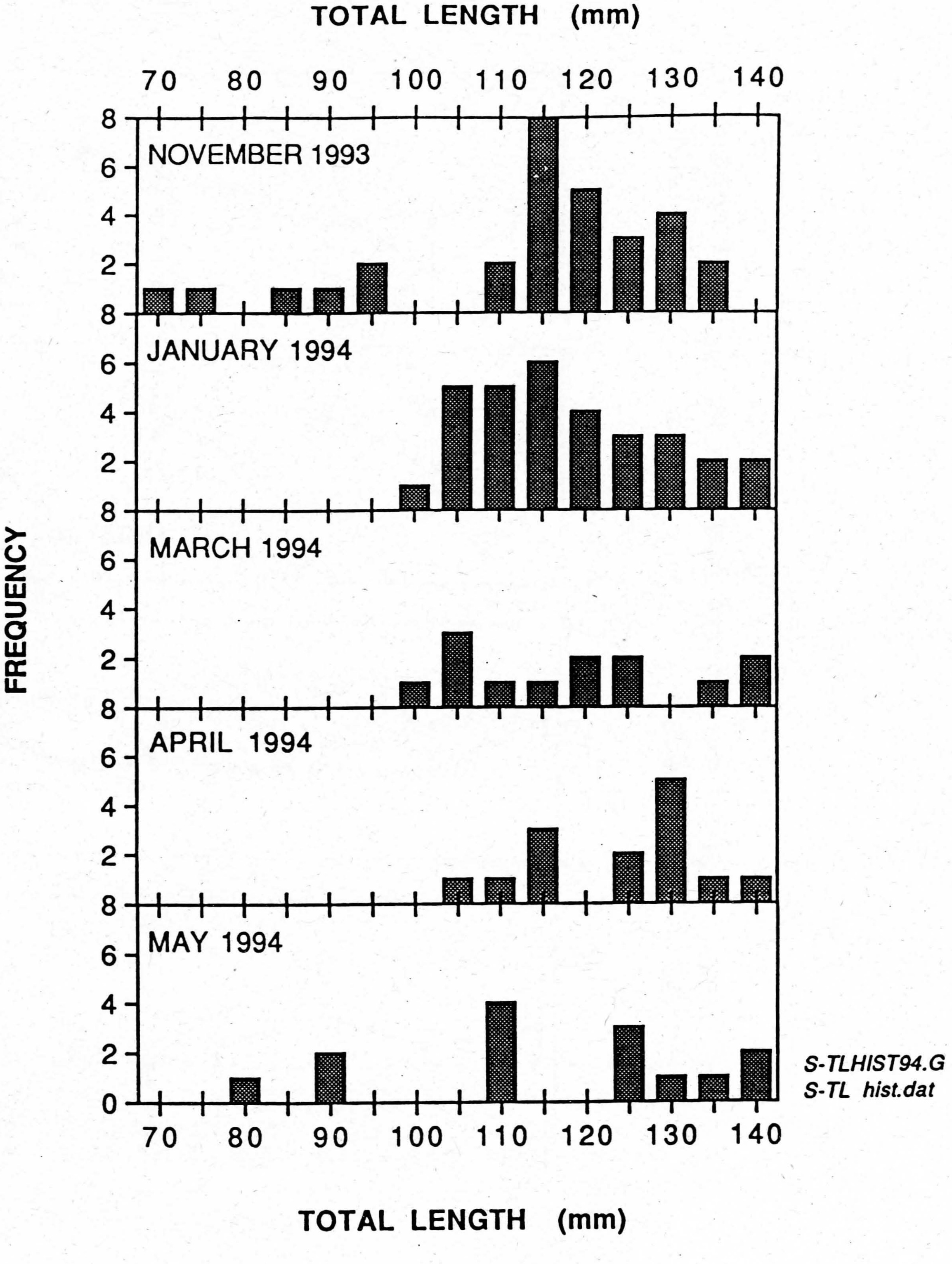


Figure 10. Total length histograms for age-0 kokanee caught in Stanley Lake during 1993-94. Fish were captured by mid-water trawl in November and May. During the other sampling periods, the lake was frozen and kokanee were captured with gillnets.

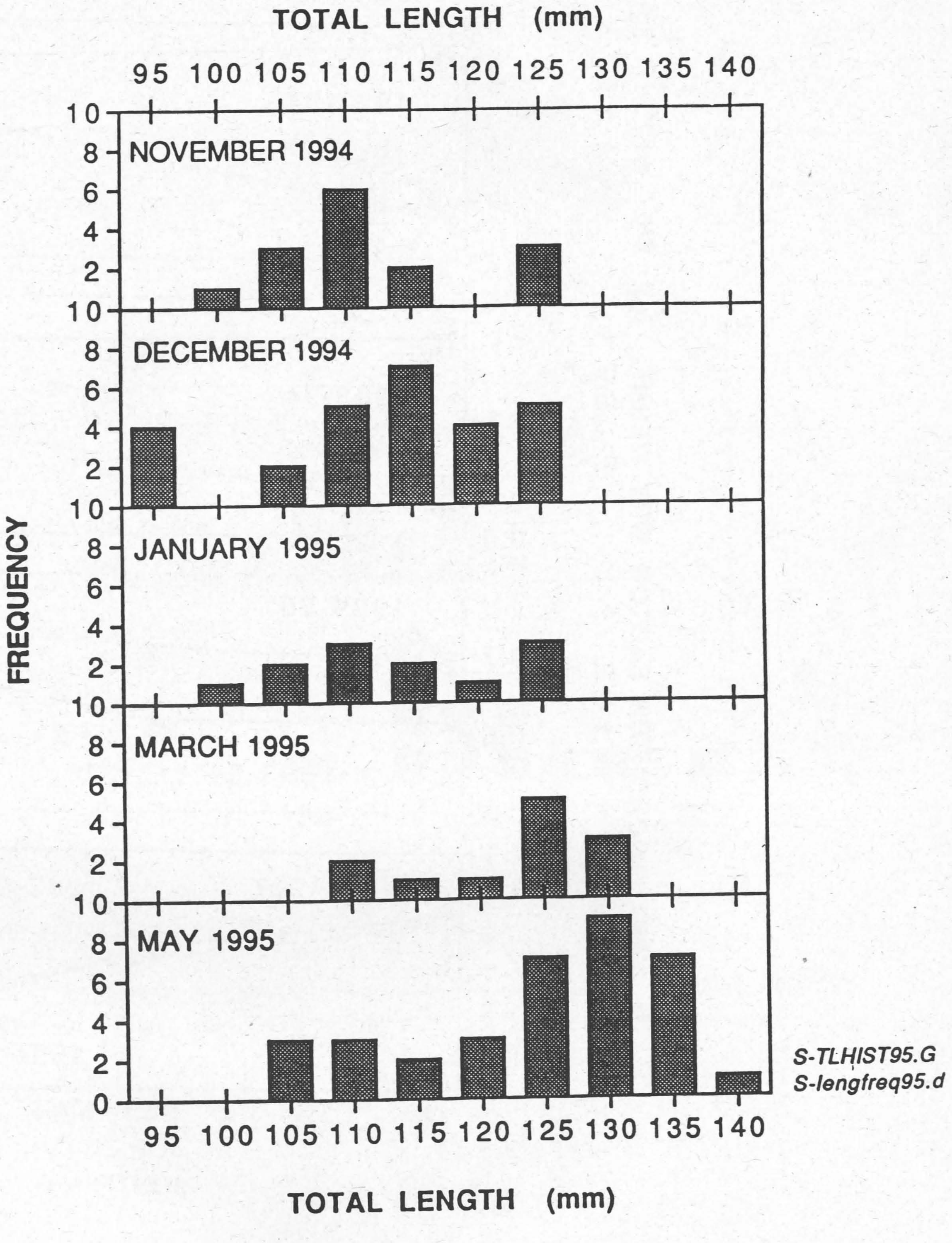


Figure 11. Total length histograms for age-0 kokanee caught in Stanley Lake during 1994-95. Fish were captured by mid-water trawl in November and May. During the other sampling periods, the lake was frozen and kokanee were captured with gillnets.

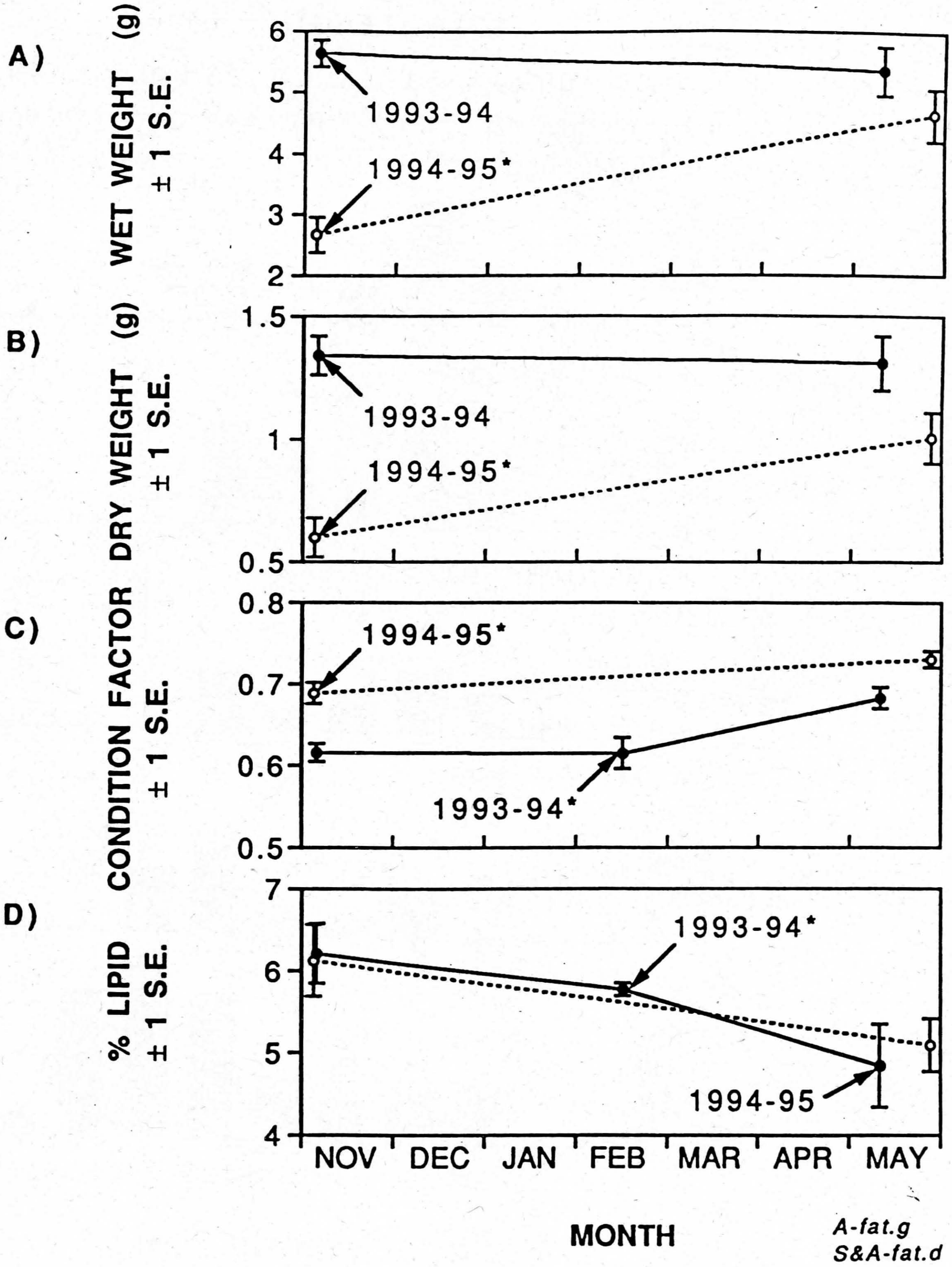


Figure 12. Over-winter changes in A) wet weight, B) dry weight, C) condition factor, and D) lipid (as a percent of dry weight) for age-0 kokanee from Alturas Lake. The asterisks indicate significant differences between November and May samples ($p < 0.05$).

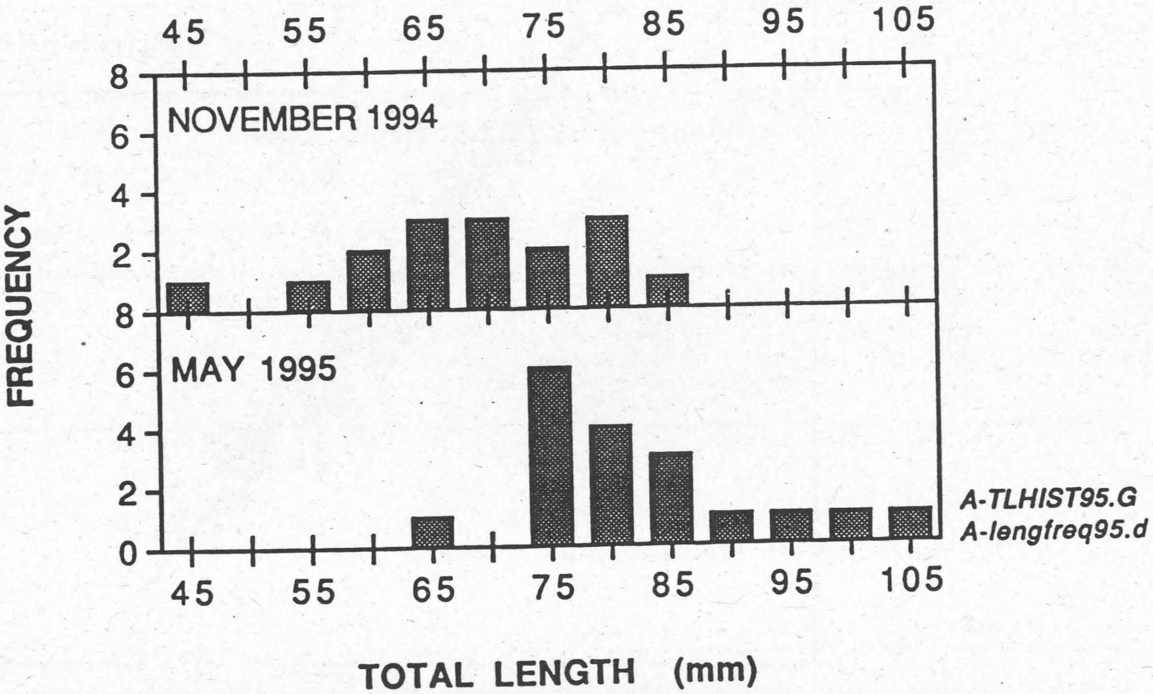


Figure 13. Total length histograms for age-0 kokanee caught in Alturas Lake during 1993-94. Fish were captured by mid-water trawl.

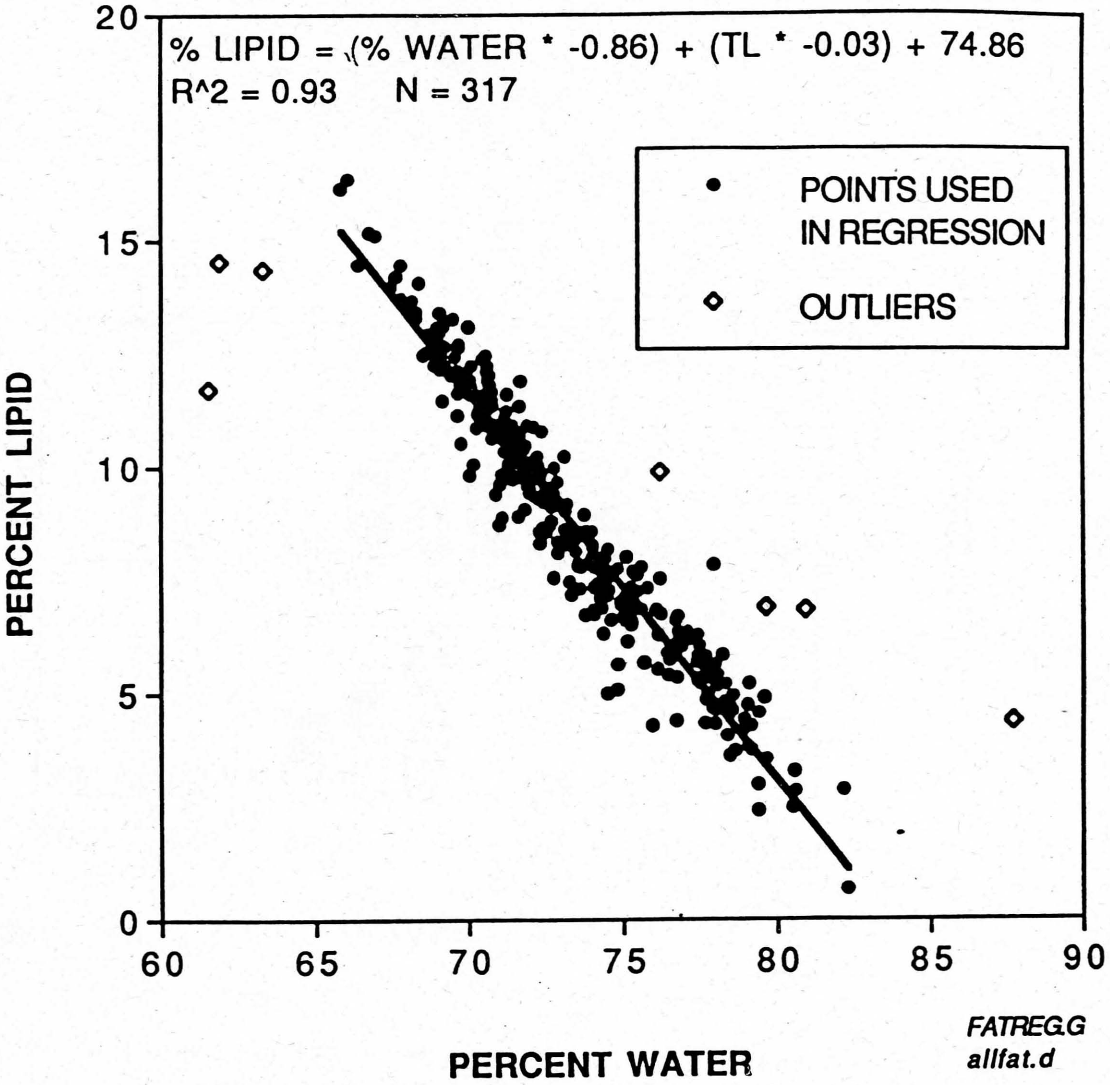


Figure 14. Relationship between lipid (as a percent of dry weight) as a function of percent water and total length (mm). Data are for kokanee salmon captured in Stanley and Alturas Lakes, as well as kokanee salmon raised in the laboratory. The plot shown is standardized for a fish of 100 mm total length.

APPENDIX 1

Table 1. T-test (two-tail) results or differences in wet and dry weights, condition factor, and % lipid for juvenile kokanee salmon in Stanley Lake from November 1993 to May 1994.

Variable	df	t	p
Wet weight	18	0.48	0.64
Dry Weight	18	1.15	0.27
Condition Factor	18	1.45	0.16
% Lipid	18	3.11	0.006

Table 2. T-test (two-tail) results or differences in wet and dry weights, condition factor, and % lipid for juvenile kokanee salmon in Stanley Lake from November 1994 to May 1995.

Variable	df	t	p
Wet weight	48	-3.49	0.001
Dry Weight	48	-0.28	0.78
Condition Factor	48	4.57	0.009
% Lipid	48	18.17	0.0001

Table 3. T-test (two-tail) results or differences in wet and dry weights, condition factor, and % lipid for juvenile kokanee salmon in Alturas Lake from November 1993 to May 1994.

Variable	df	t	p
Wet weight	6	0.27	0.80
Dry Weight	6	0.096	0.93
Condition Factor	27	-3.79	0.0008
% Lipid	27	2.05	0.051

Table 4. T-test (two-tail) results or differences in wet and dry weights, condition factor, and % lipid for juvenile kokanee salmon in Alturas Lake from November 1994 to May 1995.

Variable	df	t	p
Wet weight	32	-3.75	0.0007
Dry Weight	32	-3.10	0.004
Condition Factor	32	-2.55	0.016
% Lipid	32	1.94	0.062

CHAPTER 4

LIMNOLOGY OF SAWTOOTH VALLEY LAKES IN 1995

Chris Luecke
Michael Slater
Phaedra Budy

INTRODUCTION

In 1992 we began sampling limnological characteristics of four lakes in the Sawtooth Valley of central Idaho to assess the condition of rearing habitat for endangered Snake River sockeye salmon. Before the turn of the century, these lakes contained self-sustaining populations of sockeye salmon, but presently anadromous sockeye salmon return to only Redfish and perhaps Alturas Lakes. Significant barriers to fish migration have been created in the other two lakes (Pettit, and Stanley) such that anadromous sockeye salmon populations have been eliminated. From 1992-1994 we assessed the relative capacity of each of the four lakes to grow juvenile salmon. Analyses of these results led us to recommend that Redfish and Pettit Lakes were best suited for stocking juvenile sockeye salmon, and that addition of fertilizer to Redfish Lake would enhance growth and survival of stocked sockeye salmon.

In 1995 we continued to monitor the limnological conditions of the four lakes. Alturas and Stanley were monitored as control lakes to assess the impact that basin-wide changes in meteorology may have had on limnological characteristics. Redfish Lake was monitored to assess impacts of a whole-lake fertilization and fish stocking event. Pettit Lake was monitored to assess the impacts of stocking in the absence of lake fertilization.

In this chapter, we report on the limnological characteristics of the four lakes. Among lake comparisons were made for each measured parameter and referenced to their potential effect on growth and survival of juvenile sockeye salmon. Physical parameters included light penetration, Secchi transparency, and water temperatures. Chemical parameters included oxygen, and both dissolved and particulate forms of nitrogen and phosphorus. Phytoplankton parameters included chlorophyll concentration, biovolume of dominant taxa, and rates of primary production. Zooplankton parameters included density and biomass estimates, length frequencies, and the number of eggs carried by female cladocerans.

METHODS

During 1995 we sampled Pettit, Alturas, and Stanley Lakes approximately twice a month from May until October. Some parameters in Redfish Lake were sampled more frequently because of the whole-lake fertilization. Sampling occurred at the deep region of each lake. On each date we sampled water column profiles of light penetration, temperature, oxygen concentration and conductivity. Epilimnetic water samples were collected and sampled for chlorophyll *a*, total phosphorus (TP) and total nitrogen (TN). Secchi transparency, chlorophyll *a* and depth-stratified zooplankton samples were collected at three stations on each sampling date. On alternate sampling dates, we collected depth-stratified water samples for chlorophyll *a* and phytoplankton samples, and for total and dissolved nutrients in addition to the basic limnological sampling. Measurements of primary production at seven depths in each lake were made on four dates during the summer. Limnological methods are described in detail in Budy et al. 1994 and are briefly described below.

Temperature, Oxygen, Conductivity and Light

Temperature, dissolved oxygen concentration and conductivity profiles were measured using a Hydrolab H20 Multiparameter Water Quality Data Transmitter. On some sampling dates a Yellow Springs Instrument Model 58 Dissolved Oxygen Meter was used to measure temperature and oxygen.

Light profiles were measured using a Li-Cor Model LI-1000 Data Logger, a deck cell, and a underwater spherical quantum sensor that measured photosynthetic active radiation (400-700 nm). Water transparency was measured using a 25-cm black and white Secchi disk.

Water Chemistry

Water was collected and analyzed for total Kheldahl nitrogen (TKN), total phosphorus (TP), soluble reactive phosphorus (SRP), NO_3+NO_2, and NH_4. The epilimnion was sampled with a depth-integrating 6-m Tygon tube. Four to six additional depths were sampled using a 4-L

Van Dorn bottle. We also sampled water from the 1% light level because it represents the nominal depth of the bottom of the photic zone. Water samples collected for nutrient analysis were stored in polyethylene containers first rinsed with 0.1 N HCl and three times with sample water. All water chemistry samples were stored on ice in a cooler out on the lake and then kept frozen until analysis.

Dissolved nutrient samples [nitrate+nitrite (NO_3-N) , ammonia (NH_4-N), and soluble SRP were filtered before freezing with a 0.45 μm membrane filter (Steinhart et al. 1993). Unfiltered samples were analyzed for total Kjeldahl nitrogen (TKN) and total phosphorous (TP). Total nitrogen (TN) was calculated from the sum of TKN + NO_3-N. All nutrients were analyzed by the University of California, Davis. TP and SRP were analyzed colorimetrically using the molybdate - ascorbic acid method. Nitrogen analyses were conducted using the Kjeldahl digestion method for TKN, the hydrazine method for NO_3, and the indophenol method for NH_4.

Chlorophyll a, Phytoplankton, and Primary Productivity

Chlorophyll samples were collected from the epilimnion with 6-0 m depth-integrated tube samples and from greater depths with the Van Dorn bottle. Epilimnetic and discrete depth samples were taken at the central station, and epilimnetic samples were also taken at two additional stations 200-500 m from the index station. Two 50-ml aliquots per sample were filtered through 0.45 µm cellulose acetate filters. Filters were usually placed directly into 6 ml of buffered methanol and extracted for 24 hours in the dark. Occasionally filters were frozen immediately and then extracted as described. The extracts were analyzed using a Turner model 111 fluorometer according to (Budy et. al. 1994; Holm-Hansen and Riemann 1978).

Samples for phytoplankton enumeration and identification were also taken using the 6-0 m Tygon tube above and the Van Dorn water bottle described above. Water samples for phytoplankton analyses were preserved using Lugol's iodine solution. Phytoplankton analyses were performed by ECO-LOGIC Ltd., British Columbia, Canada.

Rates of *in situ* primary production

(PPR) were measured once a month during using the ^{14}C-technique (Wetzel and Likens 1990). Water from each of 7-11 depths sampled at the deep station in each lake was incubated in three 25-ml glass scintillation vials. Each vial was inoculated with 80 µl of 25 µCi/ml of $^{14}CHO_3$, and one vial at each depth was inoculated with 150 µl of a saturated solution of photosynthetic inhibitor, Diuron (dichloro-phenyl-dimethyurea; DCMU) to measure non-photosynthetic ^{14}C uptake. The vials were incubated at the appropriate depth for 4 h in clear acrylic plastic tubes. At the end of the incubation, the entire contents of each vial was filtered through a 0.45 µm cellulose nitrate filter and rinsed with 0.01 N HCl. They were then air dried, and subsequently counted by liquid scintillation spectrometry. Production rates were calculated by subtracting carbon uptake in the DCMU treatments from the light treatments. Dissolved inorganic carbon was estimated from pH and alkalinity measurements determined with the Gran procedure (Wetzel and Likens 1990).

In order to determine which, if any, nutrient(s) were continuing to control algal growth in Redfish Lake, we began an *in situ* bioassay on 27 September using the same methodologies employed during assays conducted in 1992 (Gross et al. 1993). Lake water was collected from the epilimnion (7 m) and metalimnion (14 m) and placed in 10-L translucent cubatainers. Two containers from each depth served as untreated controls, while pairs of other cubatainers received additions of nitrogen (460 *u*g/L), phosphorus (60 *u*g/L), carbon (6000 *u*g/L) added as sucrose, or a combination of all three. These were incubated at 7 or 14 m for 7 days. Fifty milliliter aliquots were then taken, filtered, and analyzed for chlorophyll *a* using methodologies described previously. Water from the treatments receiving C+N+P clogged the membrane filters and we could not filter the entire 50 ml. For the analysis we assumed that 25 ml of water passed before the filters clogged completely.

Zooplankton

Zooplankton were sampled at three stations in each lake. Zooplankton were collected using a closing zooplankton net (35.5-cm diameter by 158-cm long with 80-µm mesh). A General Oceanics flow meter, modified to prevent reverse flow, was used to determine net efficiency. In Redfish, Alturas and Pettit Lakes zooplankton tows were made in three depth strata: 10-m to surface, 30-m to 10-m, and bottom to 30-m. In shallower Stanley Lake, zooplankton tows were only taken from 10-0 m and from the bottom to 10 m. Zooplankton samples were preserved in a buffered 10% sucrose-formalin solution.

Most zooplankton samples were counted and measured in replicated sub-samples taken with a Hensen-Stempel pipette. Some zooplankton samples were counted in their entirety, if zooplankton were not overly abundant in that sample. In previous work, zooplankton were identified to species including *Daphnia rosea* and *pulex*, *Holopedium gibberum*, *Bosmina longirostris*, *Polyphemus pediculus*, *Epischuira nevadadensis*, and at least two species of cyclopoids. In this report we refer to each species by genus. Eggs were counted for cladocerans, and egg ratios (eggs/female) calculated for each species to identify periods of reproduction, and as a surrogate measure of surplus energy.

Lake Fertilization

Redfish Lake was fertilized weekly from 5 June through 18 October 1995. Fertilizer was added as ammonium nitrate ($(NH_4)_2HPO_4$) and ammonium phosphate(NH_3NO_3). Approximately 14 kg of phosphorous and 259 kg of nitrogen were added each week throughout the summer (see Figure 1 for loading rates) yielding N:P ratios by mass of 18:1. If all of the nutrients stayed in an 8 m thick epilimnion, resulting weekly nutrient additions were 0.3 *ug*/L of phosphorus and 6.0 *ug*/L of nitrogen. Close monitoring of the water transparency and chlorophyll a levels were conducted to insure that we stayed within the limits given us by the Department of Environmental Quality. During the end of June and early part of July these parameters nearly exceeded those limits, causing us to decrease fertilization rates for two weeks until desired water transparency and chlorophyll a levels returned.

Each week the concentrated nitrogen and phosphorous nutrients were loaded into portable tanks on a 5.5m aluminum boat. The nutrients were then pumped out of these tanks into the rotor-wash of the boat into the epilimnion of the lake. Two technicians followed 20 predetermined transects surveyed across the lake to help insure even distribution of the nutrients (Appendix 1.). Completion of weekly fertilization took 4 hours traveling at approximately 2.2 m/sec.

RESULTS

Physical and chemical parameters

Temperature and oxygen profiles were collected on each of the lakes on 10 dates from May 19 through Oct. 18 of 1995. Ice was gone from each of the lakes approximately 1 week prior to this first sampling, and each of the lakes showed some initial thermal stratification. Surface temperatures ranged from 4.5°C in Pettit to 6.9°C in Redfish (Figure 2 a-d). Thermal stratification continued to develop such that a discrete epilimnion, metalimnion and hypolimnion was established in all lakes by early August. Epilimnetic temperatures peaked in late August and ranged from 16.6°C in Pettit to 15.2°C in Alturas. Surface waters began cooling and eroding the thermocline in September, but thermal stratification was still apparent in all the lakes in mid-October. Epilimnetic temperatures in October ranged from 10.3°C in Redfish to 9.0°C in Stanley Lake. Hypolimnetic temperatures remained near 4°C for Redfish, Alturas and Pettit Lakes throughout the summer. Hypolimnetic temperatures in Stanley Lake rose to almost 6oC in August (Figure 2 a-d).

As in previous years, oxygen concentrations varied with depth in the different lakes. Alturas and Stanley Lakes mixed completely at ice-out and began the ice-free season with between 6 and 9 mg/L oxygen throughout the water column. Pettit and perhaps Redfish Lakes did not completely mix at ice-out as evident by the low early spring oxygen concentrations near the bottom of these lakes. Pettit Lake contained little oxygen below 43 m and Redfish contained little oxygen below 88 m. The volume of this anoxic water in Redfish Lake was small compared to lake volume (2%) but the monomolimnion of Pettit Lake was more substantial (9% of volume) and likely reduced rearing area for juvenile salmonids.

Oxygen concentrations declined in the hypolimnion of all four lakes throughout the summer. In Redfish and Alturas this decline was slight and likely had no impact on sockeye salmon in that even very deep water remained above 5 mg O_2/L. In Stanley Lake oxygen concentrations in the hypolimnion decline steadily throughout the summer such that water below 18 m deep was likely marginal habitat for salmonids in August and September.

Conductivity varied among lakes, ranging from relatively high values in Alturas and Stanley Lakes to lower values in Redfish and Pettit Lakes (Table 1). Conductivity increased slightly with depth in Redfish, Alturas, and Stanley Lakes. In Pettit Lake, conductivities increased substantial in the bottom 10 m indicating the meromictic (non-mixing) status of this lake. Little change in conductivities within a lake were observed during the course of the summer. This pattern is similar to that observed from previous years (Budy et al. 1995).

Total and dissolved nutrient concentrations varied among the lakes (Table 2). Low concentrations of TP and TN were low in all lakes. ANOVA indicated that TP and TN were not significantly different among lakes (TP: $F_{3,19}$=1.06, P=0.389; TN: $F_{3,18}$=1.64, P=0.215), although TP was generally higher in Alturas and Stanley Lakes and TN was generally higher in Redfish and Stanley. Dissolved nutrients were low and near the detection limits for all lakes. The TN:TP ratio varied between 11 and 14 by mass.

Water transparency

Rates of light extinction varied among the lakes (Table 1). Light extinction was greatest for Stanley lake where 1% of the surface light (photic zone depth) generally was present at a depth of approximately 12 m. Photic zones extended to approximately 18 and 24 m in Alturas and Pettit Lakes respectively. Redfish Lake had the lowest mean extinction coefficient with the photic zone extending to an average of 31 m. This clear water occurred in spite of the fertilization of Redfish Lake. Mean summer water transparency ranged from 6.3 to 11.9 m (Figure 3). Secchi depth generally decreased from May to early July, and then began to increase throughout the rest of the season in Redfish, Pettit, and Alturas lakes. Secchi depth transparency remained low in Stanley Lake for the duration of the summer. An ANCOVA, using date as the covariate, indicated that Secchi transparencies were significantly different among lakes in 1995

($F_{3,31}$=30.8, P<0.0001). A Bonfferoni multiple comparison of means indicated that Stanley Lake had significantly lower transparency compared to the other lakes ($F_{1,31}$=77.4, P<0.0001). The highest water clarity was observed in early August in Redfish and Alturas lake, in September in Stanley Lake, and in October in Pettit Lake.

Chlorophyll a

Mean summer epilimnetic chlorophyll *a* (*u*g/L) ranged from 0.3 - 1.7 ug/L among the four lakes (Figure 4). In Redfish, Alturas, and Pettit lakes, epilimnetic chlorophyll a increased steadily from May until mid-June, at which point it began to decline gradually over the rest of the summer. In Stanley Lake, epilimnetic chlorophyll a was initially high (1.7 ug/L) and remained substantially higher than the other three lakes until September. During the first month after ice-out, Redfish Lake exhibited the lowest epilimnetic chlorophyll a concentrations. After the nutrient additions to Redfish Lake reached their maximum rates, chlorophyll concentrations in Redfish Lake increased to levels exhibited in Pettit and Alturas lakes. An ANCOVA, using sampling date as the covariate, indicated that differences in chlorophyll a among lakes were significant ($F_{3,35}$=11.2, p<0.001). Bonfferoni multiple comparisons of means indicated that Stanley epilimnetic chlorophyll a concentrations were higher than the other lakes ($F_{1,35}$=33.6, p<0.001), but that the other lakes were not significantly different from one another ($F_{1,35}$=0.69, p=0.79).

Deep chlorophyll maxima were typical of the Sawtooth Valley Lakes in 1995 (Fig. 5). In Alturas, Pettit and Stanley Lakes these layers of chlorophyll were most concentrated at depths of approximately twice the Secchi transparency. In Redfish Lake the deep chlorophyll maximum was most pronounce between 30 and 50 m, approximately three times the Secchi depth.

Phytoplankton

Total phytoplankton biovolume (*u*m^3/m^3) in Redfish Lake was much greater than in Alturas, Pettit, and Stanley Lakes (Figure 6 top). This high level of phytoplankton biomass was due to increased abundances of *Synechococcus* in Redfish compared to the other lakes. This small (< 2 *u*m in diameter) cyanobacteria comprised greater than 70%

of the biovolume of Redfish Lake, compared to less than 20% of the phytoplankton biovolume of the other lakes. Phytoplankton species composition in Redfish Lake at the beginning of the summer included almost equal representation of Bacillariophytes, Chryso-Cryptophytes, and Dinophytes, and almost no Cyanophytes (Figure 6, bottom). By the end of the summer, however, the species composition was dominated by the Cyanobateria, *Synochococcus*, with lesser percentage of both Dinophytes and Chryso-Cryptophytes. Chlorophytes were present only for the first half of the summer and in very small numbers relative to other groups. The addition of fertilizer to Redfish Lake in 1995 appeared to stimulate the growth of *Synochococcus*. The small size of this phytoplankter would allow it to rapidly uptake available nutrients. Although this species is a cyanobacteria, its small size and lack of toxins allows it to be eaten by zooplankton, and keeps it from forming the surface blooms associated with other cyanobacteria.

Depth profiles of species composition for three dates in Redfish Lake showed the greatest total biovolume between 20 and 35 m in most cases (Figure 7). On 20 June and 18 August, the smallest amount of total phytoplankton biovolume was found from 55-60 m, but on 17 July, the phytoplankton biovolume was second greatest at 55 m. The depth profiles showed Cyanophytes only in the metalimnion at the start of the summer, but by 17 July and on 18 August, the Cyanophytes dominated the other depth strata as well. Bacillariophytes were present in near equal quantities in most depth strata for the three dates analyzed.

Primary production

The seasonality of primary production varied among the lakes in 1995. Primary production was highest in Redfish Lake in late July averaging 1.29 mgC/m^3/h (Fig. 8). Primary production in Redfish Lake averaged 0.91 mgC/m^3/h during the summer months (Table 3). Primary production in Stanley Lake also peaked during the mid-summer months averaging 0.71 and 0.84 mgC/m^3/h in July and August respectively. Primary production in Alturas and Pettit Lakes was generally low, averaging 0.37 and 0.56 mgC/m3/h respectively throughout the summer. Redfish Lake exhibited the most variable rates of primary production, possibly a consequence of the lake fertilization Cloudy conditions may have resulted in lower primary production measured in June in all the lakes.

Euphotic zone rates of primary production indicated that Redfish Lake supported the greatest rate of carbon fixation in 1995 (Table 3). Integration of primary production through the euphotic zone and expressed as an areal measure indicated that Redfish Lake exhibited almost twice the primary production of the other lakes in 1995. These high rates of primary production were likely due to the increased abundance of small picoplankton *Synechococcus* in Redfish Lake. These small cells are capable of fixing carbon at much faster rates than larger algal cells. In previous years areal rates of primary production were similar among lakes (Budy et al. 1994).

Nutrient addition bioassay

In the nutrient addition bioassay conducted in late September only the addition of combined N+P+C significantly stimulated chlorophyll production of the phytoplankton (Figure 9). In both the epilimnion and metalimnion, nitrogen additions provided more stimulation than the other nutrients added alone, but the increases were small. The exact amount of stimulation due to the N+P+C addition could not be determined because the samples would not filter completely. We expect that this clogging was due to bacterial populations that were stimulated by the sucrose addition. If this assumption is correct, it indicates that the bacteria in Redfish Lake are co-limited by a source of organic carbon and by phosphorus and/or nitrogen, because the sucrose additions alone did not cause the filters to clog. The phytoplankton populations in the epilimnion and metalimnion also appeared to be co-limited by N and P (and possibly C), as these nutrients added alone did not stimulate chlorophyll production.

Zooplankton

Shortly after ice-out, total crustacean zooplankton biomass (*u*g/L) was very low in Redfish, Alturas and Pettit Lakes and increased gradually throughout the summer (Figure 10, top). In Stanley Lake, total zooplankton biomass was initially higher and generally increased throughout the summer.

Total crustacean zooplankton biomass was much greater in Stanley Lake in August and September compared to the other lakes. An ANCOVA, using date as the covariate, indicated that significant differences among lakes occurred for total zooplankton biomass ($F_{3,36}$=6.17, P=0.002). Total biomass of crustacean zooplankton in Redfish Lake increased throughout the autumn and slightly exceeded biomass in Stanley Lake in October. A Bonferroni multiple comparison of means indicated that zooplankton biomass was significantly higher in Stanley and Redfish Lakes compared to Pettit and Alturas Lakes ($F_{1,36}$=11.7, P=0.002). Total zooplankton density (#/L) showed the same general pattern observed for zooplankton biomass (Figure 10, bottom).

The seasonal pattern of the species composition of crustacean zooplankton biomass differed among the lakes (Figure 11). In Redfish and Stanley Lakes cyclopoid copepods dominated the biomass in May and June. By August, large cladocerans (*Daphnia* and *Holopedium*) accounted for most of the zooplankton biomass in these two lakes. Crustacean biomass remained high into the autumn with Redfish remaining dominated by *Daphnia*, whereas biomass of crustacean zooplankton in Stanley Lake was composed of *Daphnia, Holopedium*, cyclopoid and calanoid copepods.

Crustacean zooplankton biomass remained low throughout the year in Pettit and Alturas Lakes. *Bosmina* and cyclopoid copepods dominated the biomass in spring and early summer in both lakes. In late summer and fall *Bosmina* dominated the biomass of both lakes. The small size (*Bosmina*) and evasive ability (cyclopoids) of these zooplankton taxa make the zooplankton of these lakes less valued as food for juvenile salmonids.

DISCUSSION

The summer of 1995 was cool and wet, and preceded by a winter of above average snowfall. Mean discharge of the Salmon River USGS gauging station for the October 1994 - September 1995 water year was 2108 cfs. This runoff was slightly higher than the long-term mean of 1913-1993 (1944 cfs), and similar to the October 1992 - September 1993 water year value of

1912 cfs. These conditions provided for greater amounts of spring and early summer runoff into the lakes in 1993 and 1995 as compared to 1992 and 1994 spring runoff periods (1103 and 1024 cfs respectively). If previous water runoff to nutrient inflow patterns held in 1995 (Grossman 1995), these flows would have increased nutrient loadings to the lakes. Surface temperatures and thermal stratification patterns in the lakes in 1995 were similar to those of 1993, another cool, wet summer. Epilimnetic temperatures in all the lakes were lower in 1993 and 1995 compared to the drought years of 1992 and 1994.

The limnological characteristics of Alturas and Stanley Lakes were similar in 1995 to those of previous years (1992-1994). Stanley Lake continued to be the most productive having the highest levels of chlorophyll and zooplankton biomass, and lowest water clarity and light penetration. Zooplankton biomass in Stanley Lake continued to be dominated by large-bodied cladocerans and copepods. Alturas Lake continued to contain extremely low biomass of crustacean zooplankton. Water transparency and chlorophyll concentrations were similar to previous years.

In contrast to Alturas and Stanley Lakes, the limnological characteristics of Redfish and Pettit Lakes were different in 1995 compared to previous years. In Redfish Lake epilimnetic chlorophyll concentration, and zooplankton biomass was generally ranked second (after Stanley) among the four lakes. Phytoplankton biovolume and whole-lake primary production was greater in Redfish lake compared to the other lakes. Zooplankton biomass was especially high in the late summer and autumn, exceeding the biomass of Stanley Lake in October. Zooplankton biomass in Redfish Lake in 1995 was composed of a higher proportion of large Daphnia than had occurred in previous years. *Daphnia pulex* first appeared in Redfish Lake in 1995, replacing *D. rosea* as the dominant daphnid species. This larger-bodied species should provide greater food value for planktivorous fish in Redfish Lake.

In Pettit Lake crustacean zooplankton biomass was greatly reduced in 1995 compared to previous years and was dominated throughout the summer by small-bodied forms. This reduction in crustacean zooplankton biomass occurred while amounts of nutrients, chlorophyll, and rates of primary production were similar to previous years. These changes to the limnological characteristics of Redfish and Pettit Lakes may have occurred because of the lake fertilization and salmon stocking programs that were initiated in 1995. In the following chapter we will consider these interactions in more detail.

The limnological characteristics of the Sawtooth Valley Lakes in 1995 reflect on the potential using these lakes in recovery plans for endangered Snake River sockeye salmon. Alturas Lake continued to look like a poor candidate to receive progeny from the broodstock program because of extremely lower concentrations of zooplankton food resources. The slow growth rate of native kokanee in Alturas Lake support this conclusion.

The dramatic decline in zooplankton food resources in Pettit Lake compared to previous years makes us suspect of our previous conclusions that Pettit Lake could support a higher density of juvenile sockeye salmon compared to Redfish Lake (Luecke et al. 1996). Low abundances of chlorophyll and rates of primary production may make this lake especially vulnerable to the top down effects associated with stocking of planktivorous fish.

The high primary and secondary productivity of Stanley Lake makes it attractive as a secondary lake for stocking broodstock progeny. The summer hypolimnetic oxygen decline and the potentially large number of piscivores in this lake continue to reduce its value as habitat for juvenile sockeye salmon. Subsequent evaluation of these habitat variables should be given high priority in future sampling efforts.

The increased abundance of large-bodied zooplankton in Redfish Lake under fertilization and salmon stocking suggests that the increased production of zooplankton due to enhanced nutrient loadings was at least able to keep up with the increased mortality imposed by stocked juvenile salmonids. Results from the 1995 limnological sampling suggest that lake fertilization should accompany future attempts to increase the density of juvenile salmonids in Redfish Lake. The long developmental period for crustacean zooplankton populations in Redfish Lake suggests that the feeding conditions for juvenile sockeye salmon would be best after August 1. The retention of high Secchi transparency in Redfish Lake in 1995, during the lake fertilization, indicates that the fertilization rates could be increased slightly with little chance that water quality would be jeopardized. Monitoring of water transparency and epilimnetic chlorophyll concentrations indicated that rates of nutrient additions to Redfish Lake could have been increased in 1995, particularly after zooplankton populations began increasing in early August. This additional nutrient loading might have improved growing conditions for the stocked sockeye salmon during fall and early winter, and resulted in better overwinter survival of these endangered fish.

REFERENCES

Budy, P.B., C. Luecke, W. Wurtsbaugh, and H. Gross. 1993. p. 3-28. *In* Spaulding, S. [ed.] Snake River Sockeye Salmon (Oncorhynchus nerka) habitat/ limnologic research, U.S. Department of Energy, Portland,OR. Project number 91-71. Contract number DE-B179-91BP22548.

------, ------, ------, and ------. 1995. Limnology of Sawtooth Valley Lakes with respect to potential growth of juvenile Snake River Sockeye Salmon. Northwest Science 69:133-150.

------, ------, ------, and ------. 1994. Effects of nutrient enhancement on juvenile sockeye salmon growth. Lake and Reservoir Management 9:140-145.

Gross, H.P., W.A. Wurtsbaugh, C. Luecke, and P. Budy. 1993. Nutrient limitation of phytoplankton in oligotrophic lakes of the Sawtooth Valley, Idaho. pp. 29-52 *in* S. Spalding (ed.) Snake River sockeye salmon (*Oncorhynchus nerka*) habitat/limnologic research. Bonneville Power Administration, DE-BI79-91BP22548. Portland, OR.

Gross, H.P. 1995. Evaluation of lake fertilization as a tool in the recovery of Snake River sockeye salmon. M.S. Thesis, Utah State University, 151 p.

Teuscher et al. 1994. Snake River sockeye salmon habitat and limnological research. Annual report 1994. US Dept. Energy, Bonneville Power Administration, Portland, Oregon.

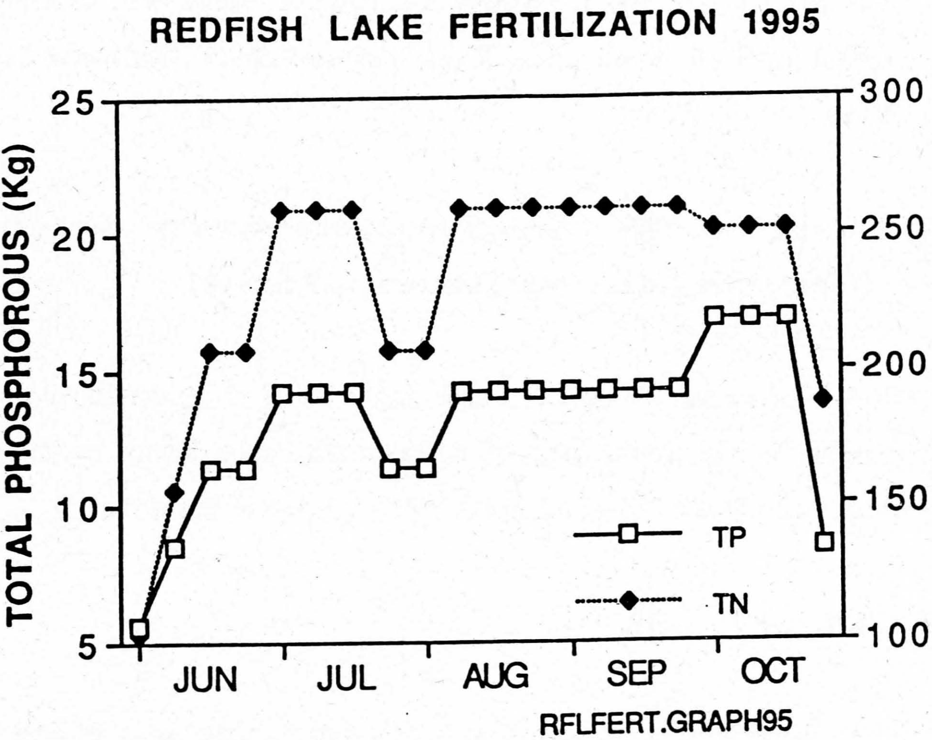


Figure 1. Rates of nutrient additions to the epilimnion of Redfish Lake during the ice-free period of 1995.

Figure 2. Temperature (°C) and oxygen (mg/l) profiles for each of the lakes during the May-October period of 1995. A) Redfish, B) Alturas, C) Pettit, D) Stanley Lakes.

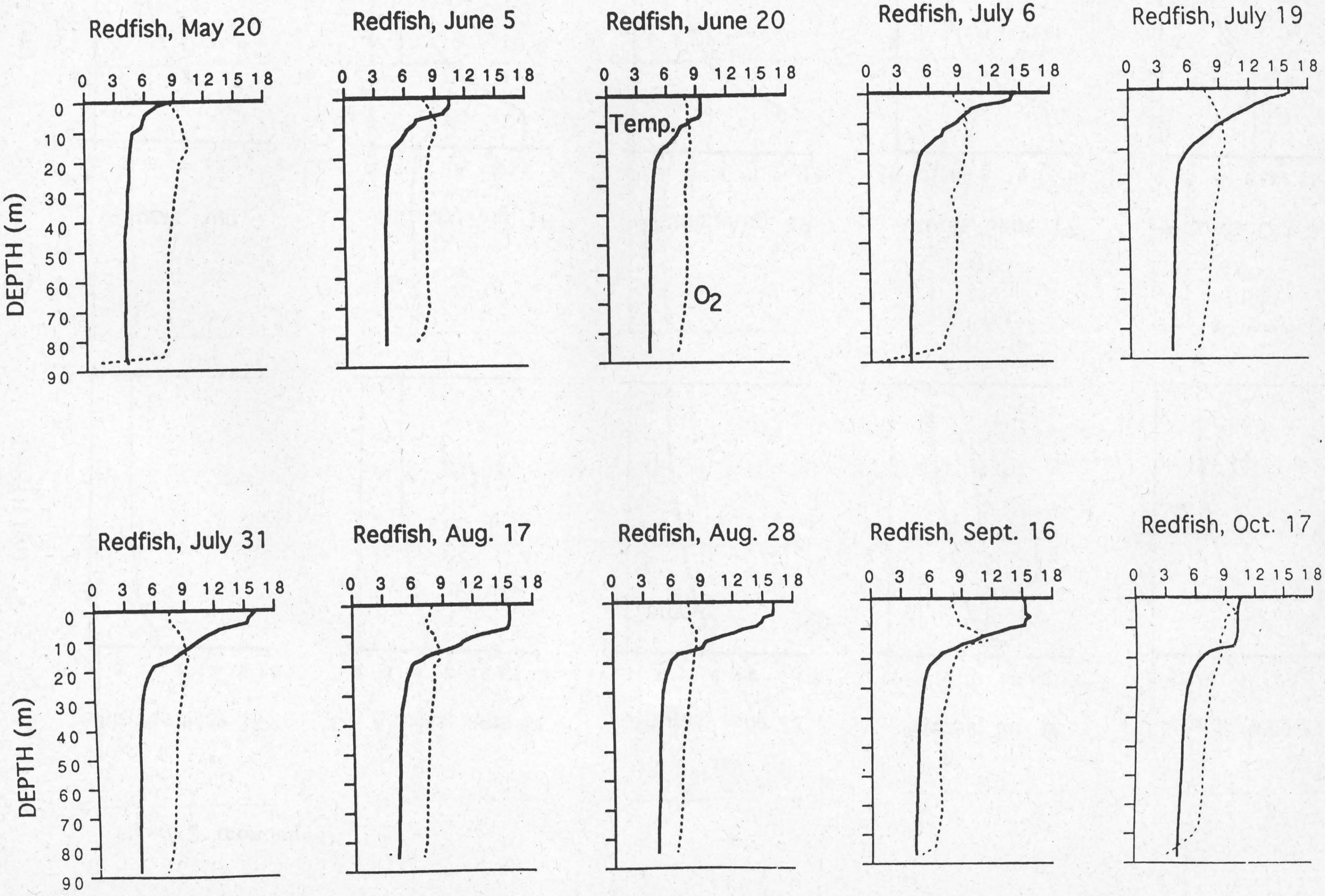


Figure 2. (continued).

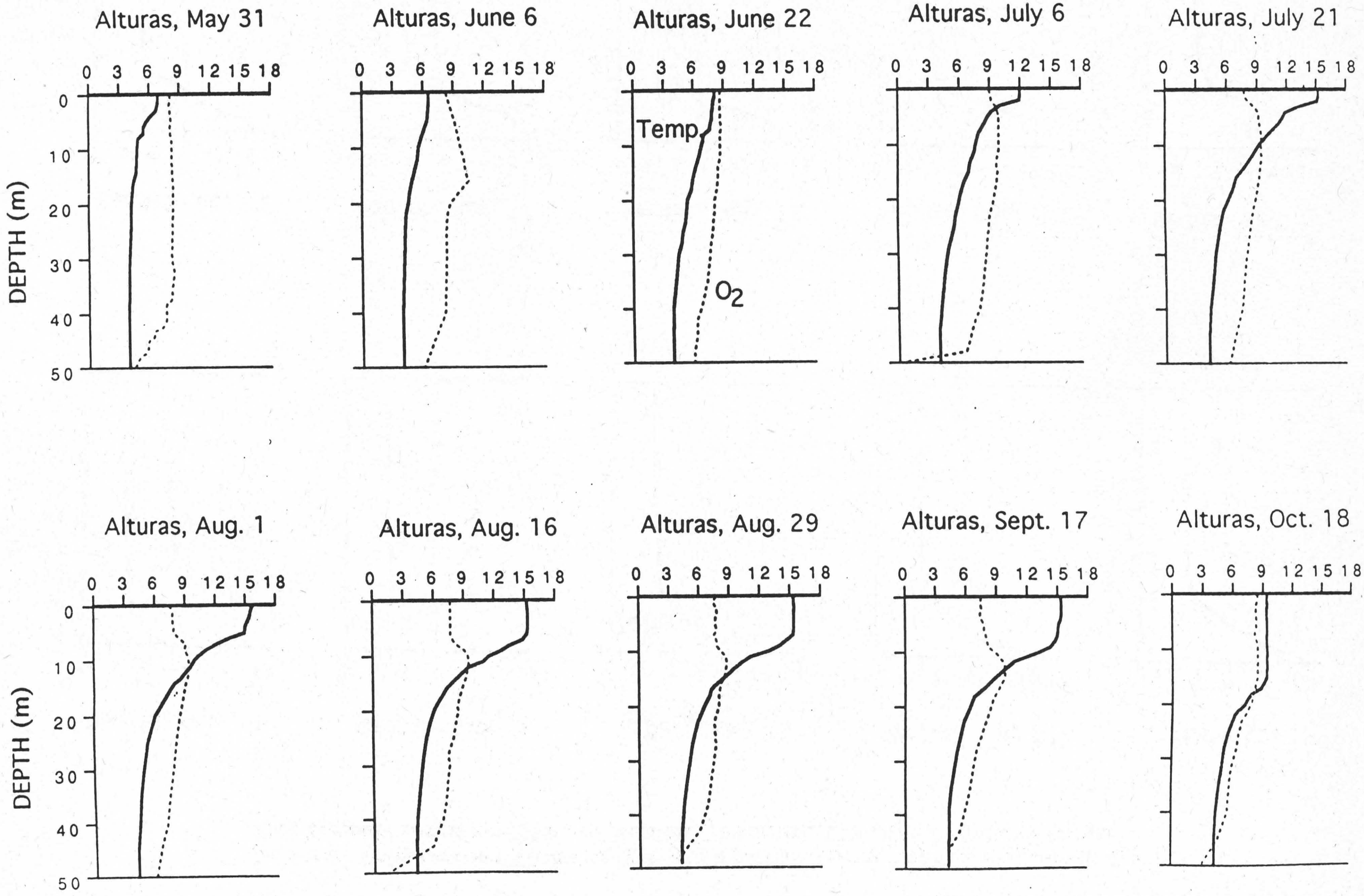
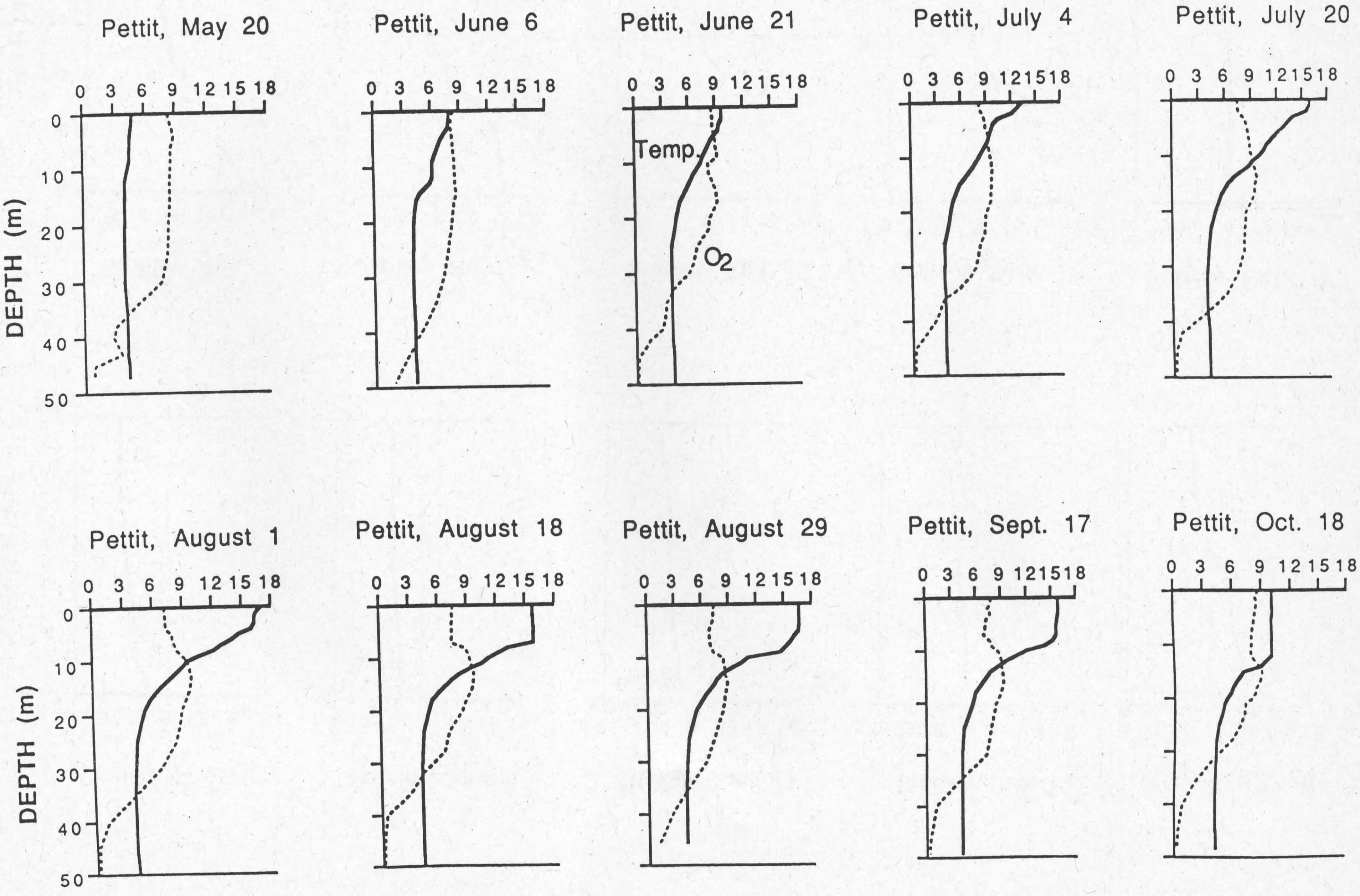


Figure 2. (continued).



Pettit, May 20

Pettit, June 6

Pettit, June 21

Pettit, July 4

Pettit, July 20

Pettit, August 1

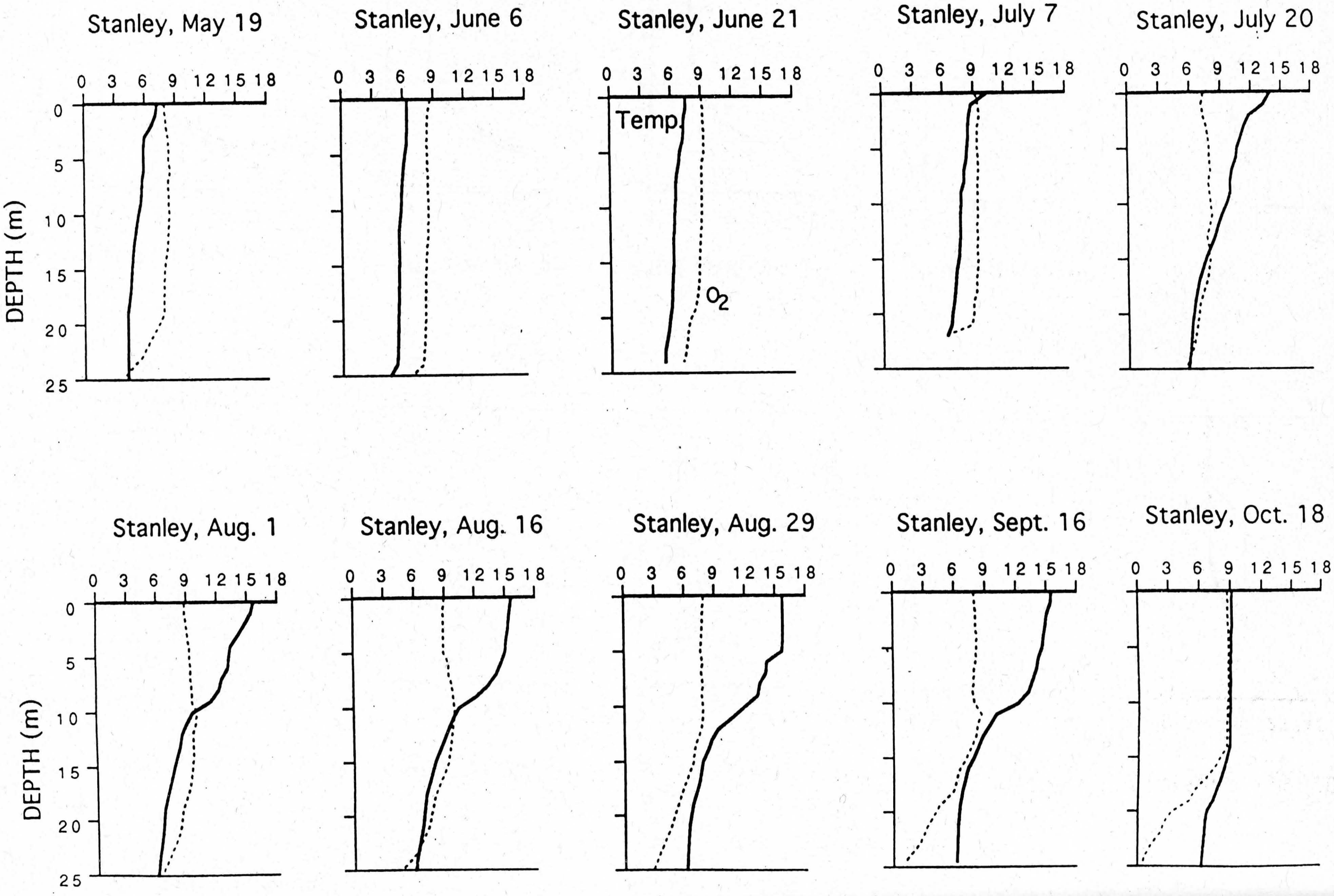
Pettit, August 18

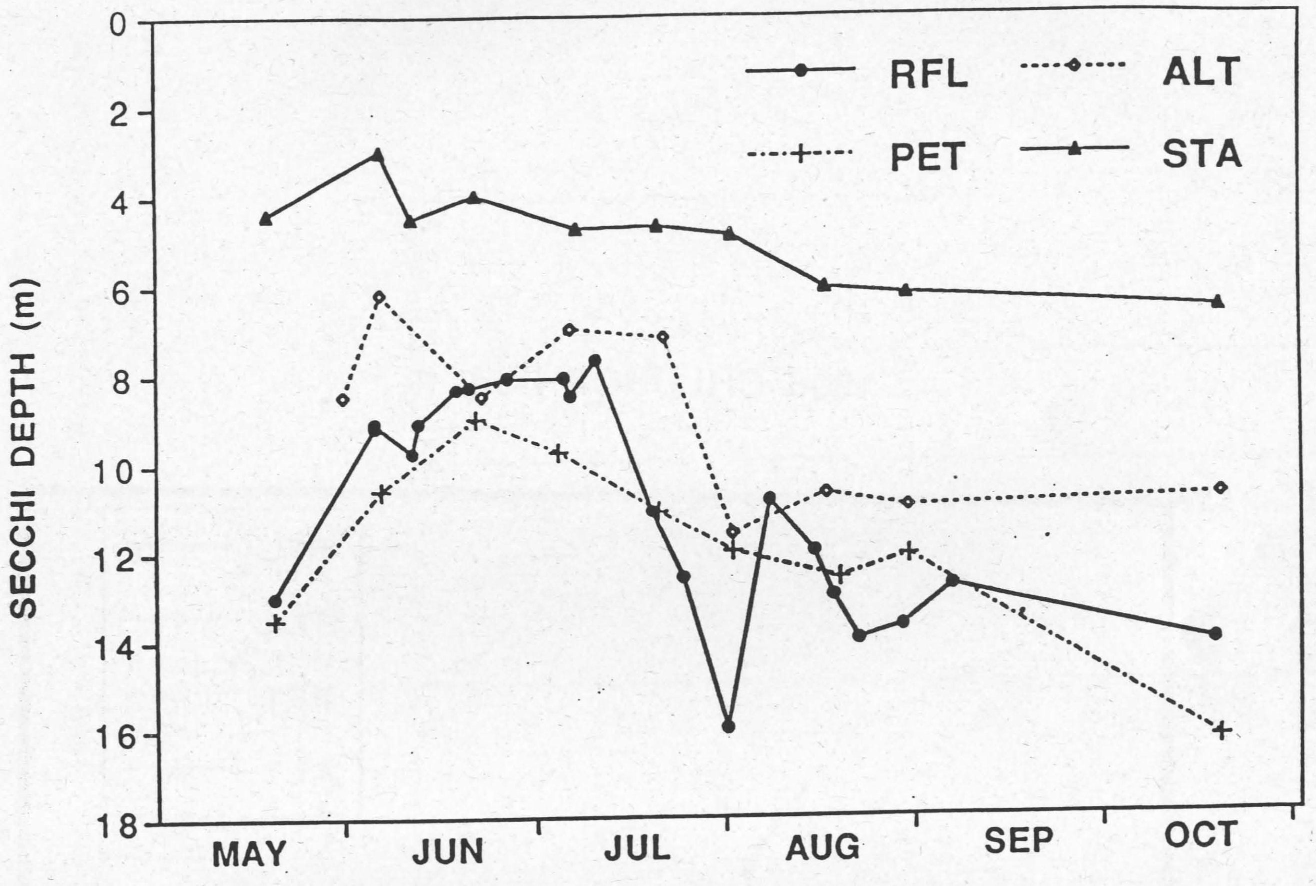
Pettit, August 29

Pettit, Sept. 17

Pettit, Oct. 18

Figure 2. (continued).





95SECCHI.g

Figure 3. Secchi depth transparency for four Sawtooth Valley Lakes in 1995.

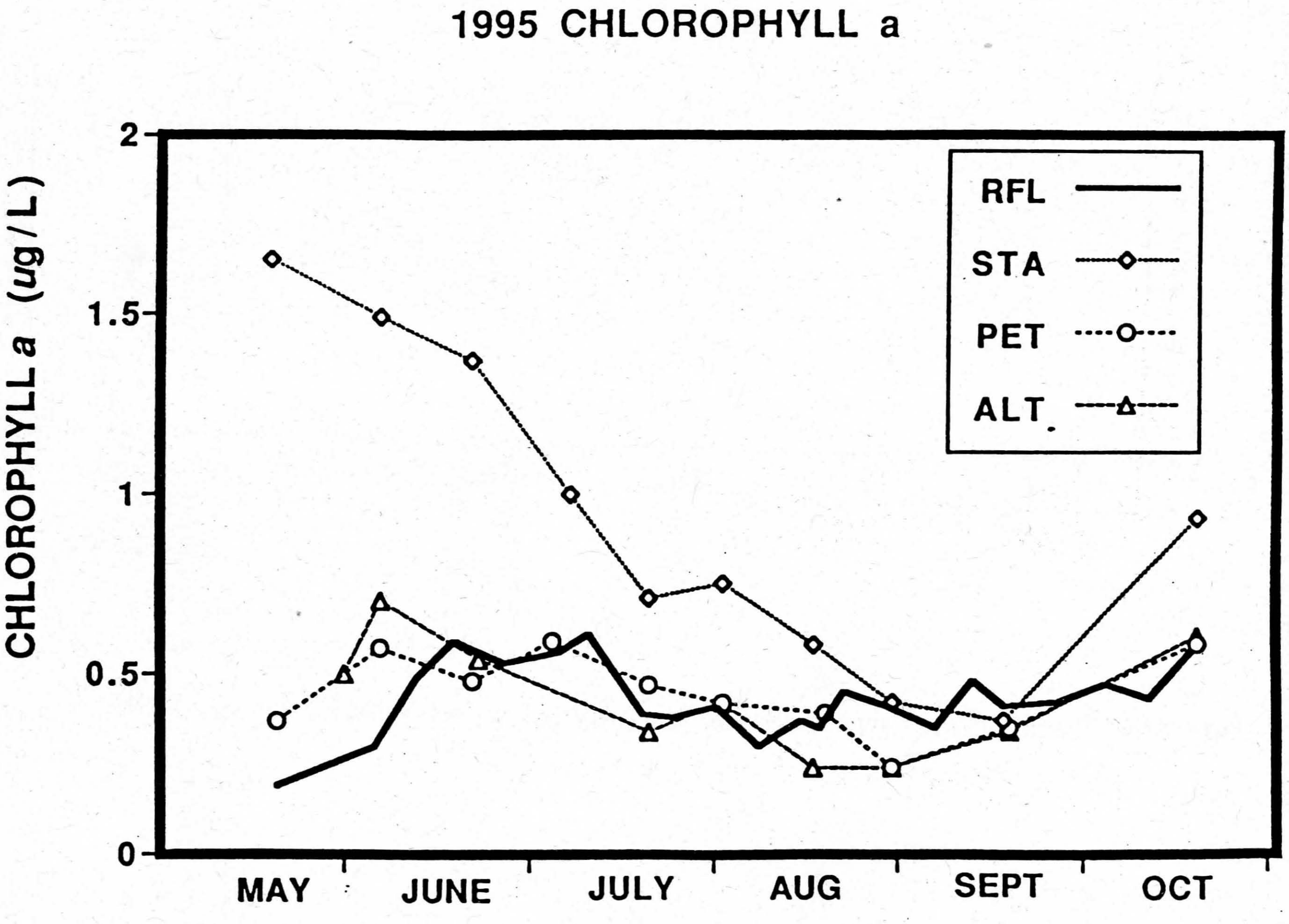
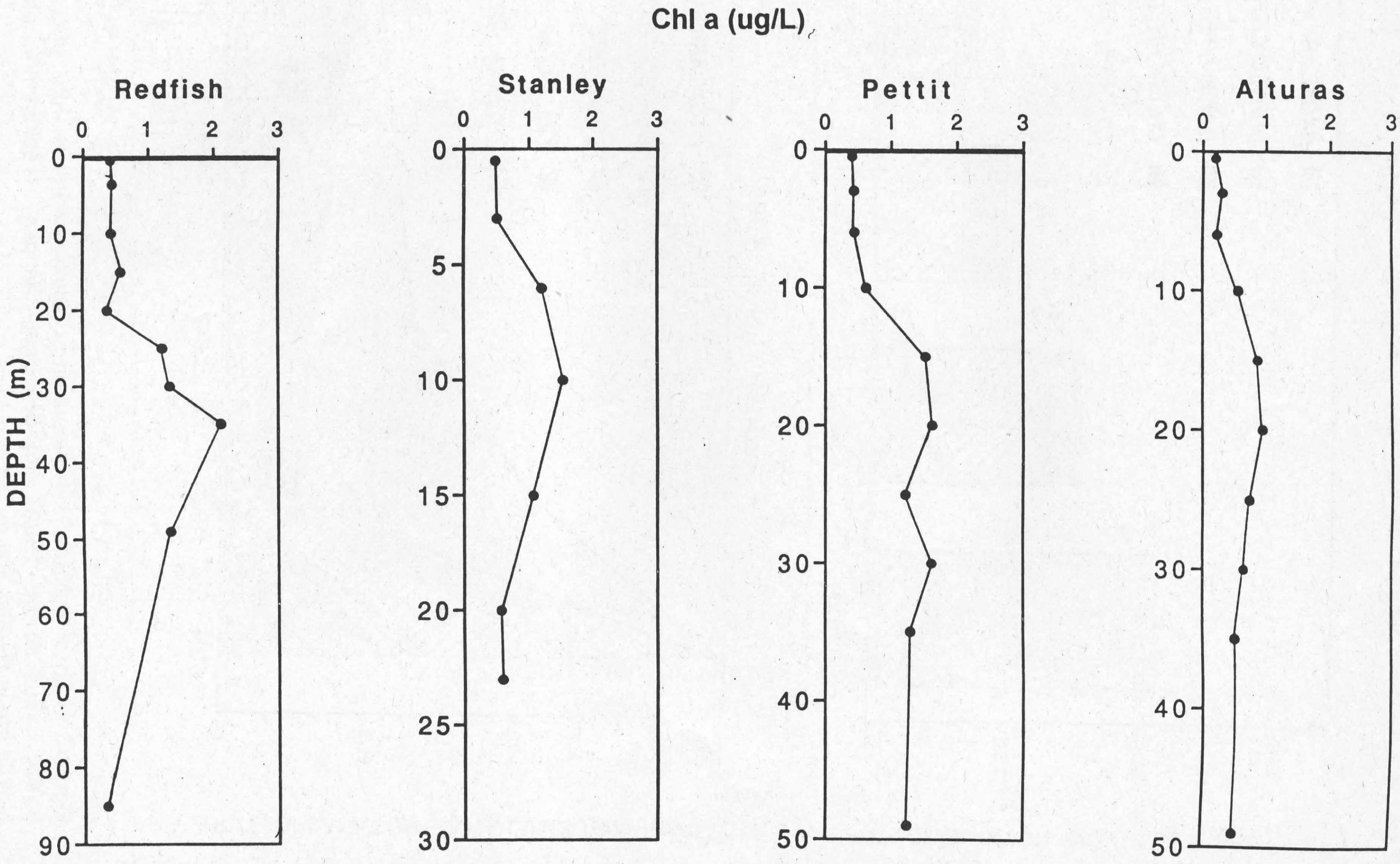
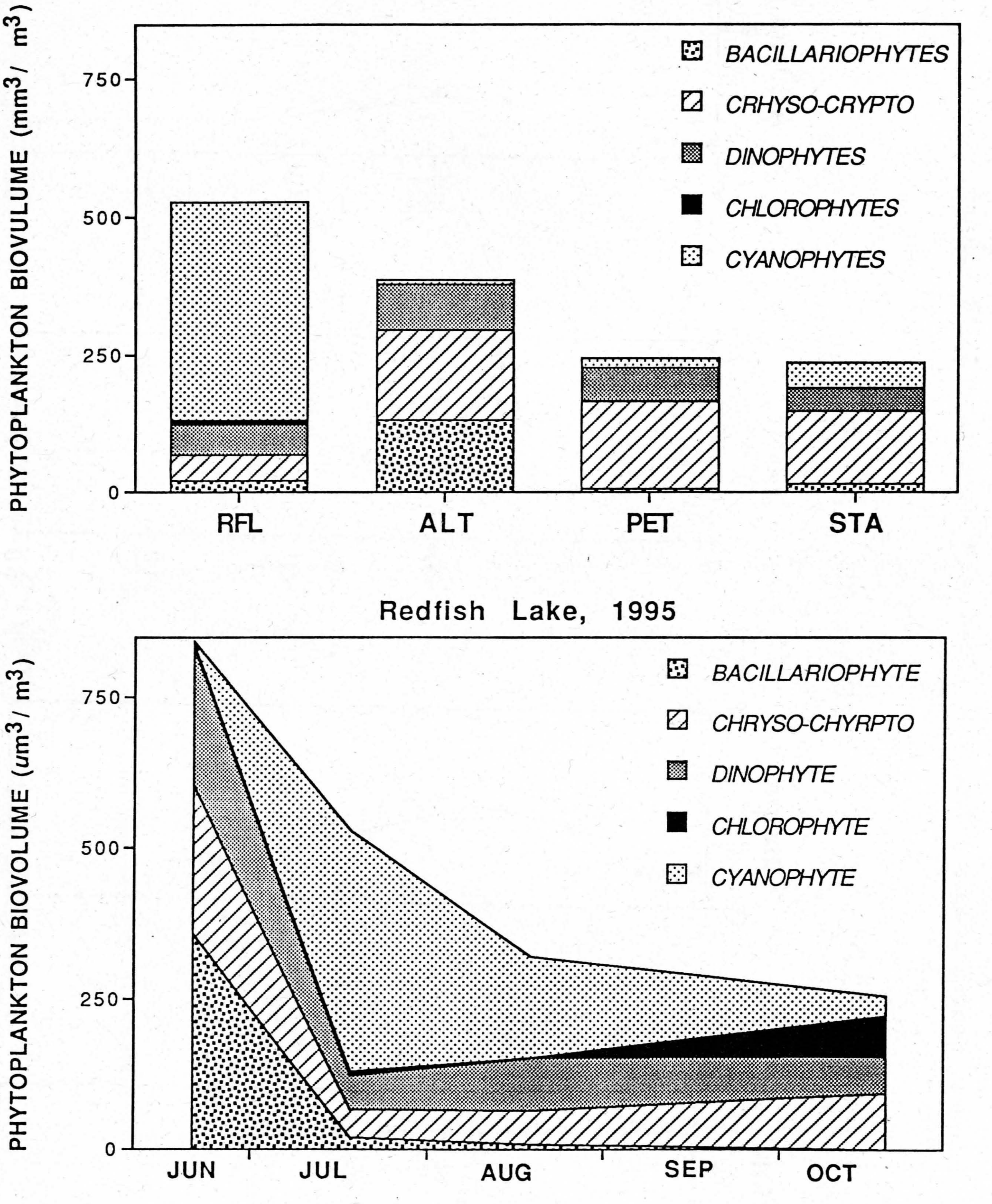


Figure 4. Chlorophyll *a* concentrations in the epilimnion of four Sawtooth Valley Lakes in 1995. Samples were collected from a 0-6 m integrating tube.

Figure 5. Depth profiles of chlorophyll *a* in Sawtooth Valley lakes on 16-18 August 1995.





PHYTO#2.g new

Figure 6 a) Total epilimnetic phytoplankton biovolume in late July of four Sawtooth Valley Lakes during 1995. b) Seasonal changes in epilimnetic phytoplankton biovolume in Redfish Lake.

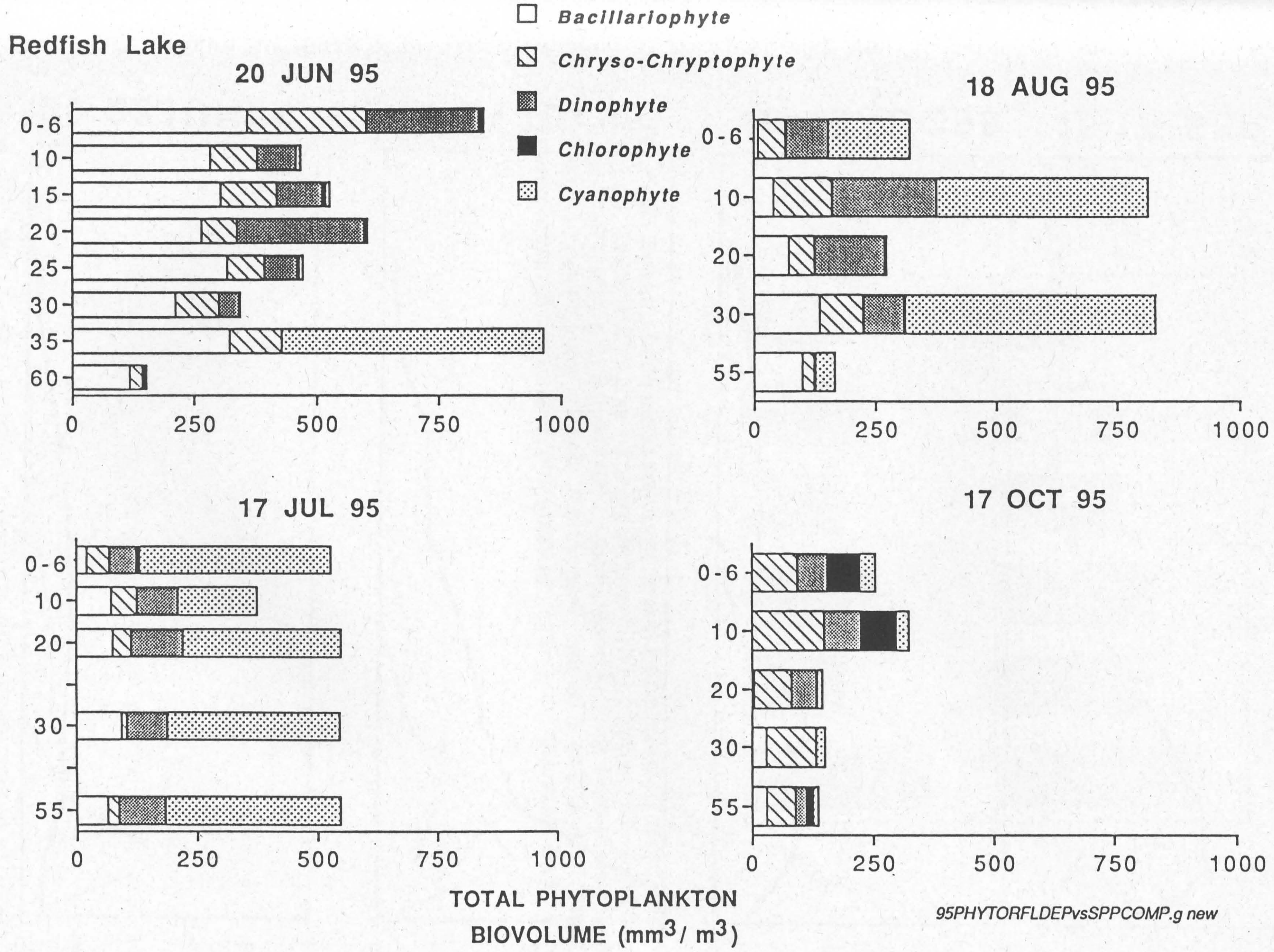


Figure 7. Depth-stratified species composition of phytoplankton in Redfish Lake during the summer months of 1995.

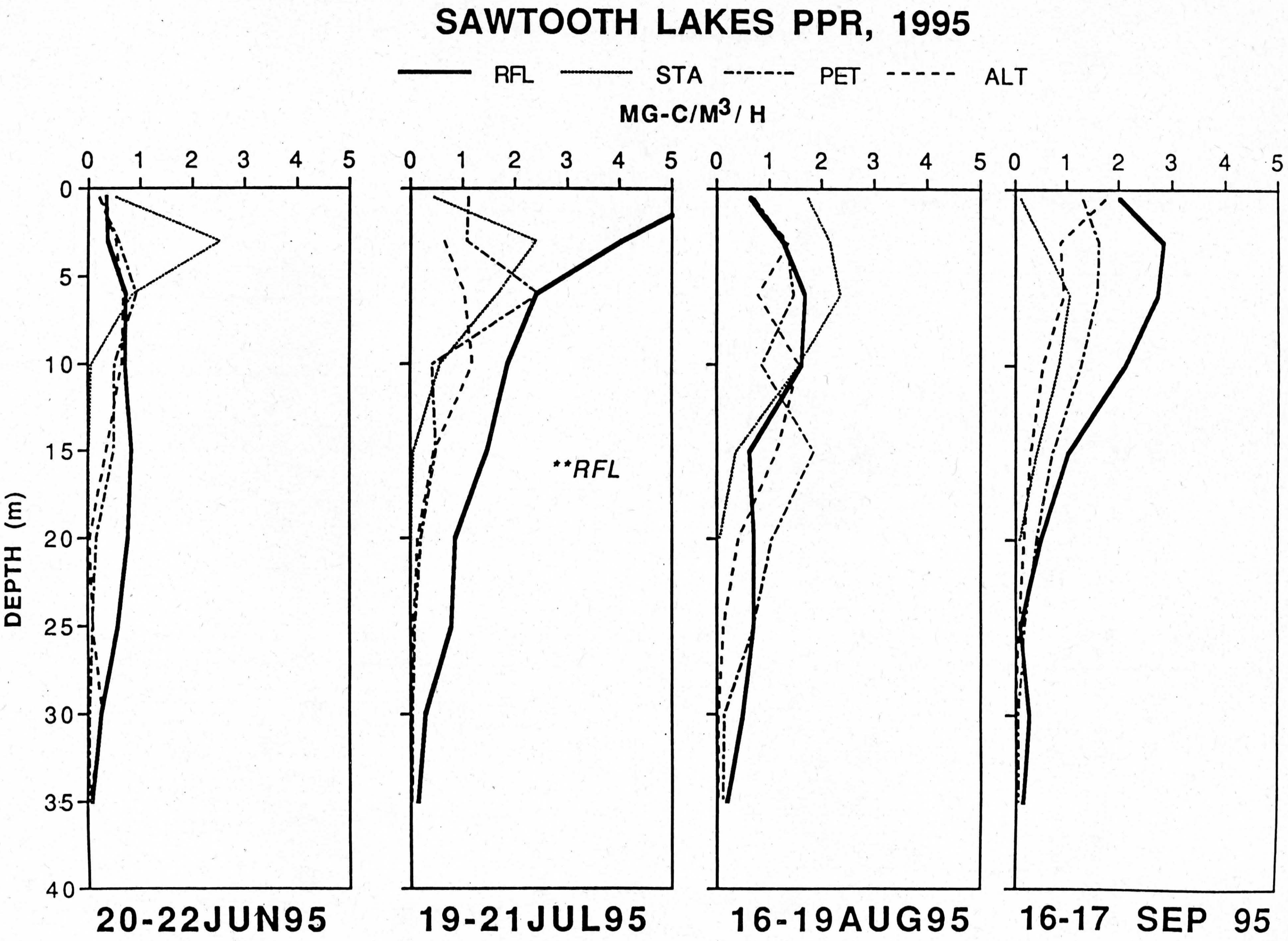


Figure 8. Rates of primary production as measured with the ^{14}C method in four Sawtooth Valley Lakes on four dates in 1995.

95LKPPR.g

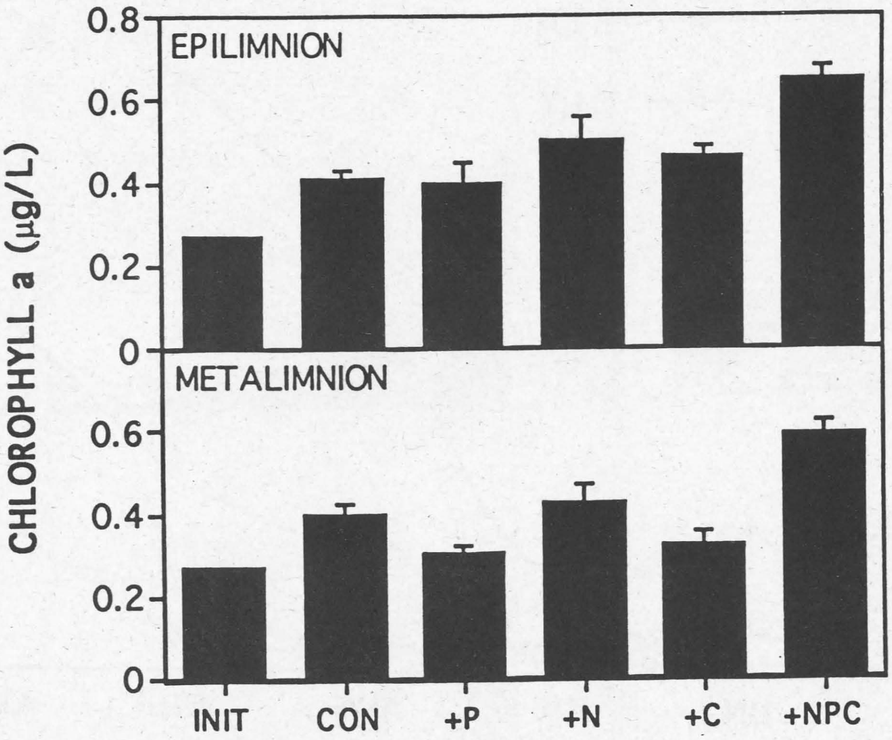


Figure 9. Results from bioassay experiment conducted 27 September - 3 October 1995. Initial lake concentrations (INIT), control (CON), phosphorus added (+P), nitrogen added (+N) carbon added (+C) and the three nutrients combined (+NPC) are shown. Chlorophyll values from the +NPC treatments are approximate. Error bars (when shown) represent 1 standard error.

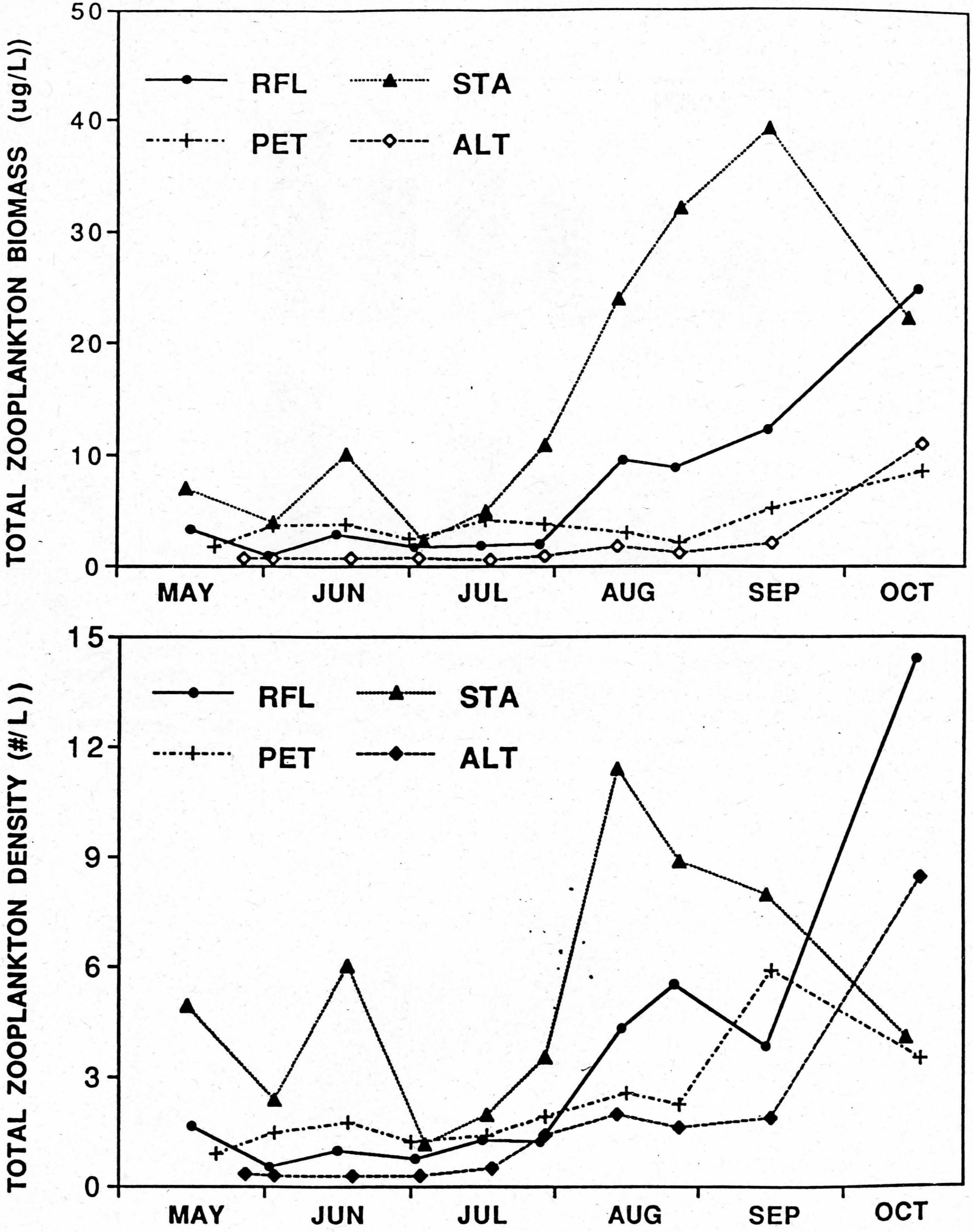


Figure 10. Total crustacean zooplankton biomass (top) and density (bottom) in four Sawtooth Valley Lakes in 1995. Depth-weighted means of 0-10 m, 10-30 m, and bottom-30 m strata are shown.

95LKZOOPag

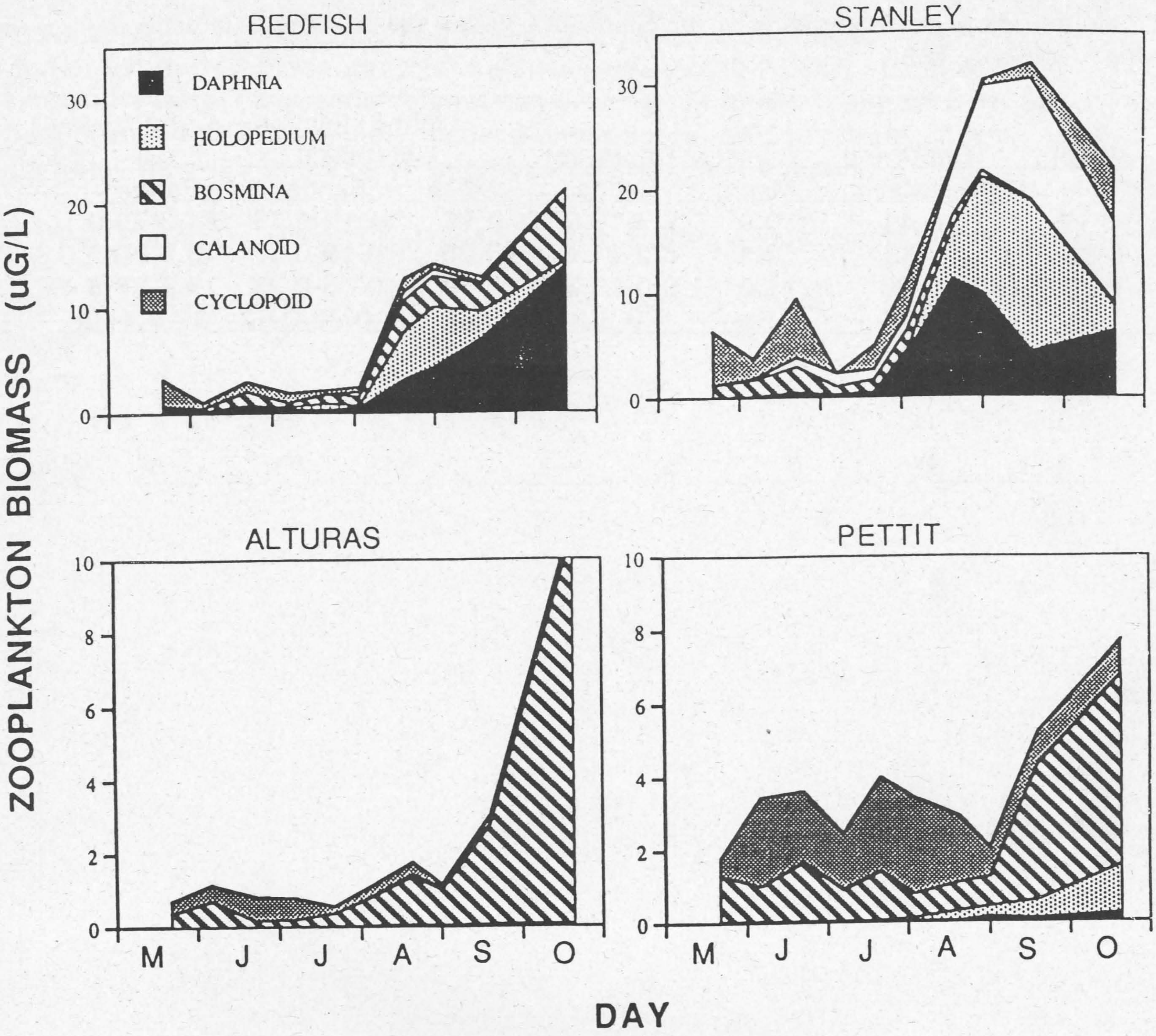


Figure 11. Species composition of crustacean zooplankton in four Sawtooth Valley Lakes in 1995. Data are from depth-weighted tows as in Figure 10. Note the difference in scales on the y-axis.

Table 1. Limnological characteristics of the Sawtooth Valley Lakes. Means are for the ice-free period, May-October, 1995.

Lake	Area (km2)	Depth (m)		Secchi Depth (m)		Light Extinction Coefficient		Conductivity
		Max	Mean	Mean	Range	Mean	Range	Range
Redfish	6.17	92	41	10.9	7.7-16.0	0.15	0.13-0.18	24.3-29.0
Alturas	3.38	53	33	9.1	6.2-11.6	0.25	0.19-0.32	43.5-50.2
Pettit	1.63	52	28	11.9	9.0-16.3	0.19	0.16-0.23	19.7-38.8
Stanley	0.73	26	26	4.9	3.0-6.6	0.40	0.22-0.62	37.9-47.6

Table 2. Mean nutrient concentrations (*ug*/L) in Sawtooth Valley Lakes in 1995. Spring overturn represents water column averages 20-28 May. Epilimnetic values are from 0-6 m wate column samples. TP is total phosphorus, TN is total nitrogen, SRP is soluble reactive phosphorus, NO3 is nitrate, NH4 is amonia. The ratio of TN to TP is presented in the last column.

	Spring Overturn		Mean Seasonal Epilimnetic Nutrient Concentration					
Lake	TP	TN	TP	SRP	NO3	NH4	TN	TN:TP
Redfish	5	70	7	2	4	8	81	11.6
Alturas	8	56	9	2	2	8	109	12.1
Pettit	4	55	6	1	1	4	83	13.8
Stanley	8	103	8	2	2	8	93	11.6

Table 3. Rates of primary production (mg C / m^3 / h) in Sawtooth Valley Lakes in 1995. Values represent depth-weighted means of profiles data shown in Figure 8. Means for each lake and each month are included in the right column and in the bottom row. The values in parentheses are the means of rates of primary production measured on four sample dates, integrated through the water column, and expressed on an areal basis (mg C / m^2 / h).

Lake	Jun	Jul	Aug	Sep	Lake Mean
Redfish	0.54	1.29	0.81	0.99	0.91 (31.9)
Alturas	0.23	0.37	0.59	0.30	0.37 (13.0)
Pettit	0.29	0.47	0.87	0.61	0.56 (19.6)
Stanley	0.48	0.69	1.08	0.52	0.69 (17.3)
Monthly Mean	0.39	0.71	0.84	0.61	

Appendix 1. Redfish lake fertilization transect coordinates beginning at the boat ramp on the north, traveling in a crisscross pattern to the southern end of the lake, and returning to the boat ramp.

	West Shore		East Shore
1	44 08 136	2	44 07 657
	114 55 689		114 55 287
3	44 07 418	4	44 06 912
	114 56 020		114 55 393
5	44 06 818	6	44 06 486
	114 56 208		114 55 580
7	44 06 523	8	44 06 112
	114 56 410		114 56 220
9	44 06 317	10	44 05 850
	114 56 733		114 56 762
11	44 06 095	12	44 05 885
	114 57 042		114 56 564
13	44 06 385	14	44 06 202
	114 56 626		114 55 852
15	44 06 673	16	44 06 646
	114 56 360		114 55 528
17	44 07 055	18	44 07 179
	114 56 076		114 55 353
19	44 07 796	20	44 07 955
	114 55 811		114 55 159

Appendix 2

SAWTOOTHS LAKES NUTRIENT DATA
RFL 1995

DATE	INDEX #	DEPTH (m)	TP ug/l	TN ug/l	SRP ug/l	NO3 ug/l	NH4 ug/l
20MAY95	52,51,67,65,66	6-0	5	70	1	2	5
	54,53,70,68,69	10	5	82	1	2	2
	56,55,73,71,72	20	5	45	1	0	1
	58,57,76,74,75	30	5	80	1	4	1
	60,59,79,77,78	40	5	115	1	2	1
	62,61,82,80,81	60	5	53	1	13	2
	64,63,85,83,84	85	6	112	1	25	15
5JUN95	111,112,125,126,127	6-0	6	71	1	0	3
	113,114,128,129,130	10	6	31	1	0	1
	115,116,131,132,133	18	6	47	1	0	1
	117,118,134,135,136	25	7	49	1	0	0
	119,120,137,138,139	35	6	43	1	3	1
	121,122,140,141,142	55	5	101	1	9	3
	123,124,143,144,145	75	6	51	1	1	5.5
20JUN95	150,151,168,169,170	6-0	4	57	1	0	5
	152,153,171,172,173	10	3	59	1	0	2
	154,155,176,175,174	15	3	35	1	0	2
	156,157,177,179,178	20	4	55	1	0	1
	158,159,180,181,182	25	3	55	1	1	2
	160,161,183,184,185	30	3	52	1	1	3
	162,163,186,187,188	35	3	48	1	4	3
	165,164,189,190,191	60	3	62	1	9	7
	166,167,192,193,194	85	3	57	1	21	14
06JUL95	283,282,270	6-0	8	73	5	3	6
	271,272,266	10	8	61	1	0	1
	273,274,267	15	7	73	1	0	4
	275,276,268	20	10	70	1	0	3
	277,278,269	30	8	146	1	0	1
	279,280,284	55	8	89	1	7	4
	281,288,285	85	8	199	1	25	10
19JUL95							
31JUL95	364,365,357	6-0	13	58	1	5	1
	366,367,358	10	8	94	1	1	0
	368,369,359	15	9	43	1	0	0
	370,371,360	20	11	31	1	0	0
	372,373,361	25	11	47	1	1	0
	374,375,362	55	12	34	1	9	3
	376,377,363	85	8	38	1	22	1
17AUG95	385,386,378	6-0	5.5	???	1	0	14
	387,388,381	10	7	66	1	4	1
	389,390,382	20	9	73	1	1	0
	391,392,383	30	9	35	1	0	0
	393,394,384	35	10	54	1	4	2
	395,396,379	55	8	44	1	9	1
	397,398,380	85	8	38	1	31	4
28AUG95	458,459,451	6-0	14	125	1	13	14
	460,461,452	10	13	77	2	2	2
	462,463,453	20	12	66	1	0	0
	464,465,454	30	13	44	1	0	1
	466,467,455	35	11	37	1	4	1
	468,469,456	55	9	42	1	15	3
	470,471,457	85	14	34	1	24	2
16SEPT95	505,506,549	6-0	5	100	3	6	13
	507,508,550	10	3	138	2	1	4
	509,510,551	20	4	67.5	2	1	5
	511,512,552	30	4	47	3	2	5
	513,514,553	35	4	65	2	5	4
	515,516,554	55	4	43.5	2	14	4
	517,518,555	85	4	25	3	56	5
17OCT95	583,584,576	6-0	5	91	2	8	9
	585,586,577	10	4	56	2	5	4
	587,588,578	20	3	60	2	0	1
	589,590,579	30	4	42	2	10	4
	591,592,580	35	3	52	2	15	5
	593,594,581	55	4	46	3	25	2
	595,596,582	85	7	96	4	14	18

Appendix 2

SAWTOOTHS LAKES NUTRIENT DATA

ALT 1995

DATE	INDEX #	DEPTH (m)	TP ug/l	TN ug/l	SRP ug/l	NO3 ug/l	NH4 ug/l	*COMMENTS*
31MAY95	86,87,96,97,98	6-0	8	56	2	0	9	
	88,89,99,100,101	10	8	46	1	0	2	
	90,91,102,103,104	15	8	51	1	0	2.5	
	92,93,105,106,107	21	7	34	1	0	1	
	94,95,108,109,110	45	10	145	2	29	3	
22JUN95	241,242,243,244,245	6-0	7	135	1	2	3	
	247,248,249,246,250	10	7	75	1	0	2	
	251,252,253,254,255	20	7	95	1	1	3	
	256,257,258,259,260	30	7	82	1	3		
	261,262,263,264,265	48	8	54	2	5	3	
21JUL95	315,316,310	6-0	14	130	1	2	5	
	317,318,311	10	13	94	1	1	1	
	319,320,312	20	13	63	1	1	1.5	
	321,322,313	30	13	76	1	3	2	
	323,324,314	50	17	118	4	27	3	
16AUG95	401,402,412	6-0	12	145	1	6	14	
	404,403,413	10	11	116	1	5	7	
	405,406,114	20	11	79	1	0	0	
	408,407,415	30	13	64	1	1.5	0	
	410,409,416	49	14	81	2	16	8	
17SEPT95	529,530,560	6-0	4	96	3	1	4	
	535,536,563	20	6	91	2	0	2	
	539,540,566	48.5	8	53	3	24	2	
18OCT95	632,633,627	6-0	7	93	2	3	10	
	634,635,628	10	6	58	2	0	2	
	636,637,629	20	6	40	2	0	2	
	638,639,630	30	7	24	2	5	2	
	640,641,631	48	5.5	51	2	22	2	

Appendix 2

SAWTOOTHS LAKES NUTRIENT DATA
PET 1995

DATE	INDEX #	DEPTH (m)	TP ug/l	TN ug/l	SRP ug/l	NO3 ug/l	NH4 ug/l	*COMMENTS*
20MAY95	27,26,38,36,37	6-0	4	55	1	1	5	
	29,28,41,39,40	12	5	72	1	0	3	
	31,30,44,42,43	25	5	81	1	0	3	
	33,32,47,45,46	35	5	147	1	47	3	
	35,34,50,48,49	45	17	331	2	5	219	*looks like we hit bott*
21JUN95	200,201,202,203,204	6-0	4	117	1	1	3	
	205,206,207,208,209	10	3	86	1	0	3	
	210,211,212,213,214	15	3	96	1	0	3	
	215,216,217,218,219	20	4	92	1	0	3	
	220,221,222,223,224	25	4	53	1	7	3	
	230,231,232,233,234	35	4		1	52	4	
	235,236,237,238,239	48	18		1	7	210	
20JUL95	330,331,325	6-0	8	104	1	1	1	
	332,333,326	10	6	77	1	0	0	
	334,335,327	20	7	99	1	0	1	
	336,337,328	30	7	70	1	0	0	
	338,339,329	49	18	185	1	1	126	
18AUG95	439,440,433	6-0	10	94	1	2	4	
	441,442,434	10	9	64	1	1	0	
	443,444,435	20	9	57.5	1	1	1	
	445,446,436	30	11	52	1	0	0	
	447,448,437	49	23	312	3	4	213	
17SEPT95	571,572,573	6-0	4	54	2	1	5	
	543,544,568	20	5	68	2	0	1	
	547,548,570	48	6	218.5	2	3.5	179	
18OCT95	602,603,597	6-0	5	75	2	1	3	
	604,605,598	10	4	87	2	0	1	
	606,607,599	20	5	67	2	0	1	
	608,609,600	30	5	81	2	1	3	
	610,611,601	47	14.5	288	2	5	163	

Appendix 2

SAWTOOTHS LAKES NUTRIENT DATA
STA 1995

DATE	INDEX #	DEPTH (m)	TP ug/l	TN ug/l	SRP ug/l	NO3 ug/l	NH4 ug/l	COMMENTS
19MAY95	2,1,12,11,13	6-0	8	103	2	1	17	
	4,3,14,15,16	9	7	93	1	3	7	
	6,5,17,18,19	13	7	80	1	5	9	
	8,7,20,21,22	17	6	103	1	8	12	
	10,9,23,24,25	22	7	136	1	29	45	
21JUN95	260,261,262,263,264	6-0	6	63	3	5	9	
	265,266,267,268,269	10	7	81		4	5	
	270,271,272,273,274	15	6	83	1	6	7	
	275,276,277,278,279	20	6	84	1	7	12	
	280,281,282,283,284	23	6	101	1	9	17	
20JUL95	345,346,340	6-0	10	116	1	1	2	
	347,348,341	10	10	103	1	0	0	
	349,350,342	15	10	78	1	2	3	
	351,352,343	20	9	91	1	10	20	
	353,354,344	25	18	146	1	12	61	
16AUG95	422,423,417	6-0	12	114	1	5	7	
	424,425,418	10	12	79	1	0	0	
	426,427,419	15	13	67	2	1	2	
	428,429,420	20	13	78	1	14	29	
	430,431,421	23	14	138	1	17	44	
16SEPT95	527,528,564	6-0	6	67	2	1	4	
	521,522,557	15	4	83	2	1	2	
	525,526,559	23	4	68.5	2	35	3	
18OCT95	617,618,612	6-0	5	92	2	1	6	
	619,620,613	10	5	100	2	0	2	
	621,622,614	15	4	54	2	2	2	
	623,624,615	20	6	54	2	45	4	
	625,626,616	25	14	187	2	53	56	

CHAPTER 5

A COMPARISON OF FOUR YEARS OF LIMNOLOGY IN THE SAWTOOTH LAKES WITH EMPHASIS ON FERTILIZATION IN REDFISH LAKE

Phaedra Budy
Chris Luecke
Wayne A. Wurtsbaugh

INTRODUCTION

The stimulation of fish growth and production through the addition of nutrients at the bottom of aquatic food webs has been applied to a variety of different experimental questions and systems. Experiments investigating the effects of nutrient additions on aquatic food webs have ranged from very small scale laboratory assays to lake enclosure studies and to whole-lake or ecosystem manipulations (Kyle et al. 1995; Peterson 1993; Stockner 1981, 1987; Stockner and Hyatt 1984; Langeland 1982; Liebold 1989). Further, fertilizer addition has been used to stimulate production in fish aquaculture for decades (Nelson and Edmundson 1955).

Experimental studies where nutrients were added to aquatic food webs have demonstrated increases in biomass of phytoplankton, zooplankton, and fish. Langeland (1982) reported a rapid increase in biomass and production of zooplankton and fish accompanied by a change in foraging behavior of arctic char in response to enhanced zooplankton populations. Liebold (1989) gives an summary of measured bottom-up effects from controlled nutrient addition experiments. Similarly, McQueen et al. (1986) concluded, using a series of regressions plotting the dependent variable (consumer) as a function of the independent variable (producer), that the maximum biomass at each trophic level is controlled by the nutrient input at the bottom of the food web through . Alaskan and Canadian sockeye biologists have also demonstrated repeatedly that whole-lake fertilization has increased juvenile salmon growth and production through increased lake productivity (Stockner 1981, 1987; Stockner and Hyatt 1984; Kyle 1994). Consequently, whole-lake fertilization is now accepted as a viable management approach for increasing the carrying capacity of salmon rearing lakes in these areas.

However, fertilization experiments sometimes yield equivocal results, and regression-based comparisons have been used to demonstrate both top-down and bottum-up control of aquatic food webs (McQueen et al. 1986; Del Melo et al. 1992; Carpenter and Kitchell 1992). Further, the results of whole fertilizations in sockeye systems have often been confounded by continuously changing fish densities and lack of understanding of mechanisms of energy transfer (Stockner and Hyatt 1984; Stockner 1981; Shortreed and Stockner 1990; Kyle 1994, 1995). In many of these cases, nutrient enhancement had measurable effects on primary and secondary productivity while benefits to fish growth and production were less apparent. Much ecological debate centers around the effects of top-down selective planktivory on primary consumers in determining lake productivity versus the effects of nutrients from the bottom of the food web (Carpenter and Kitchell 1988; DeMelo et al. 1992; Carpenter and Kitchell 1992; Matson and Hunter 1992). This top-down control over ecosystem function, in addition to omnivory or "middle-out" effects, can mask or overcome the effects of nutrients on ecosystem productivity (Carpenter et al. 1994; Polis et al. 1989; Devries and Stein 1992).

In this paper, we use four years of limnological sampling in four historical sockeye nursery lakes to compare inter- and intra-annual variability in lake productivity in order to evaluate potential rearing conditions for juvenile, endangered sockeye salmon (*Oncorhynchus nerka*). Snake River Sockeye salmon were listed as endangered in 1991 in response to the limited number of adults returning to spawn in Redfish, Alturas, Pettit, and Stanley lakes located in the Sawtooth Mountains, Id (Bevan et al. 1994). At this time, a brood-stock program was developed with several thousand brood stock progeny slated for stocking into Redfish Lake in 1995, 1996, and 1997.

We initiated limnological sampling in the summer of 1992 and continued sampling each lake approximately once every two weeks from May until October in 1993, 1994, and in 1995. In 1994, we fertilized limnocorrals in Redfish Lake to evaluate juvenile *O. Nerka* growth under fertilized and control treatments. Based on the results from this experiment, in addition to data from limnological sampling and simulation modeling, we recommended the whole-lake fertilization of Redfish Lake in 1995 (Budy et al. *submitted Ecological Applications*). We use limnological data to evaluate the effects of nutrient enhancement, annual weather patterns, and planktivore consumption on lake productivity in Redfish, Alturas, Pettit, and Stanley lakes. We compare Redfish Lake productivity in 1995 to Redfish Lake in 1994, 1993, and 1992 and to the productivity in the other three lakes, Alturas, Pettit, and Stanley lakes in 1995 to evaluate whole-lake fertilization.

METHODS

Limnological sampling began in April of 1992 and continued through October of 1995. The lakes were sampled at least twice a month from June until September (all years) and then monthly for the rest of the year weather permitting (1993, 1994, and 1995). Water temperature, light, water transparency (Secchi depth), epilimnetic chlorophyll *a* , and zooplankton biomass, were sampled and analyzed according to methods described in Budy et al. 1995, Budy et al. 1993, and in Chapters 1 and 2 of this report. Planktivore densities were obtained through combined trawling and hydroacoustic estimates of fish abundance (done by Shoshone Bannock Tribes and Fish and Game in years 1994-5).

Lake fertilization in Redfish Lake was initiated on 5 June 1995 and continued on a weekly basis throughout the summer until 18 October 1996. Nitrogen and phosphorous were added to the epilimnion of the lake in the form of $(NH_4)_2)HPO_4$ and NH_3NO_3 at a 18:1 N:P ratio (by mass). Approximately 14 kg of phosphorous and 259 kg of nitrogen were added each week. If all of the nutrients stayed in a 8 m thick epilimnion, resulting weekly nutrient additions were 0.3 *ug*/L and 6.0 *ug*/L. Concentrated nutrients were loaded into portable plastic tanks aboard a 22' aluminum boat and then pumped into the rotor wash of the boat. The nutrients were distributed along 20 pre-determined transects using GPS coordinates for navigation.

Long term changes in lake productivity were evaluated using the limnological parameters from the four years and four lakes. For evaluation of bottom up effects, we compared total zooplankton biomass as a function of chlorophyll *a* using summer data (May through October) in the four lakes over the four years. For some analyses, indices of zooplankton production (biomass and egg ratios) were lagged by sampling period to assess links between primary and secondary production. We also evaluated zooplankton production using egg ratios for cladoceran zooplankton (eggs/adult) as a function of chlorophyll *a* (also time lagged).

To evaluate top-down effects of zooplanktivorous fish (primarily sockeye and kokanee salmon) on zooplankton biomass and size composition, we first compared estimates of *O. nerka* density in the four lakes over the four years using reported hydroacoustic and trawl estimates of fish abundance. We then compared mean cladoceran zooplankton length as a function of fish density among the four lakes for the four years.

For evaluation of inter- and intra-annual differences in lake productivity, we compared epilimnetic temperature (°C), 1% light level (m), water transparency (Secchi depth-m), epilimnetic chlorophyll *a* (*u*g/L), total zooplankton biomass (*u*g/L), and an estimate of *Daphnia* biomass (*u*g/L) in the four lakes, Redfish, Alturas, Pettit, and Stanley Lakes over the four years, 1992-1995. We applied a multi-variate cluster analysis in order to describe the four lakes among the four years in terms of general productivity. Our objectives were several fold. We wanted to determine if lakes would cluster together by overall productivity or if annual differences in precipitation and weather events lead to differences in annual productivity causing the lakes to cluster by year instead of by lake. And finally, we were interested in determining whether Redfish Lake in 1995, the year of whole-lake fertilization, was most similar to Redfish Lake in other years, or to a different group of lake-years. The data were sorted by lake and year, and complete linkage cluster methods were used. The data were standardized to mean=0 and SD=1 before running the cluster analysis because the variables all used different metritics (ie. meters versus *u*g/L). All statistics were done in S.A.S. version 6.11.

We also used multiple dimensional scaling (MDS) as a second type of cluster analysis. MDS standardizes the mean of all parameters to 1.0, and then partitions the variation among parameters to two orthoganal multivariate axes. Each year for each lake can then be plotted in this two dimensional space. We conducted two MDS analyses. In the first, we used the data from the most frequently sampled limnological parameters (Table 1). This data included three physical parameters (epilimnetic temperature, light extinction coefficient, and Secchi transparency), one phytoplankton parameter (epilimnetic chlorophyll), and two zooplankton parameters (crustacean zooplantkon biomass, daphnid biomass). In the second analysis we eliminated epilimnetic temperature and light extinction, and focussed on the biological parameters that were most relevant to lake fertilization and sockeye salmon growth.

In order to determine if there was a statistically significant difference between the general productivity of Redfish Lake in 1995 (when it was fertilized) as compared to the other 3 years --when it was not fertilized, MANOVA's for all four lakes were used. The limnological variables, Secchi depth (m), epilimnetic chlorophyll *a* (*u*g/L*)*, and total zooplankton biomass (*u*g/L*)*, were tested as dependent on year --using 1992, 1993, 1994, and 1995. Further, if the MANOVA's detected a significant difference in productivity among the four years, we then contrasted 1995 against 1992, 1993, 1994. The same analyses were performed on limnological data from Pettit, Alturas, and Stanley lakes. This approach allowed us to determine if Redfish Lake was significantly more productive in 1995, than in the other years, and to determine if there were factors other than fertilization that may have effected the productivity of these lakes in 1995. Univariate tests of the independent limnological variables versus year were used to evaluate MANOVA results.

We compared rates of primary productivity in Redfish Lake in 1993, 1994, and 1995 in addition to phytoplankton biovolume and species composition from 1992 through 1995. Because of unequal numbers of sample dates, we were unable to use these parameters in our multivariate comparisons. Daphnia density (#/L) and eggs ratios (#/adult) were also compared among the four years as part of our analysis of whole-lake fertilization.

RESULTS

Part I: Limnology in Redfish, Alturas, Pettit, and Stanley Lakes --1992-1995

Water transparency

Secchi depth transparency varied both seasonally and annually among the four lakes over the four years (Figure 1). Secchi depths usually started increasing in July and increased throughout the summer and early fall. Water transparency then decreased dramatically during fall overturn in October or November and remained shallow until mid-summer. Stanley Lake consistently showed the shallowest, and Redfish and Pettit the deepest, Secchi depth transparency for all four years sampled. Secchi depths were shallowest in all four lakes in 1995 as compared to the other three years.

Chlorophyll a

Epilimnetic chlorophyll *a* showed a pattern of winter and spring peaks and late summer lows among the four lakes from 1992 to 1995 (Figure 2). Epilimnetic chlorophyll *a* typically increased from late September until May or June, at which point it began to decline again. The highest epilimnetic chlorophyll *a* concentration was consistently observed in Stanley Lake. Epilimnetic chlorophyll *a* was similar among Redfish, Pettit, and Alturas Lakes for all four years sampled.

Zooplankton

Total zooplankton biomass showed the same seasonal pattern of high biomass from July through October and lower biomass from December to June in all four lakes for the four years sampled (Figure 3). Zooplankton biomass was generally higher in Stanley and Pettit Lakes, lower in Redfish Lake, and lowest in Alturas Lake. This pattern dominated for all years sampled except in 1995 at which time zooplankton biomass dropped dramatically in Pettit Lake. The highest overall zooplankton biomass was observed in the summer of 1992 in Stanley and Pettit Lakes while Redfish and Alturas showed biomass similar to other years in 1992. It is important to note, however, that in 1993 we changed our sampling regime for zooplankton, and the difference in 1992 zooplankton biomass may be in part a result of that change. In 1992 we sampled with two types of tows, an epilimnetic tow (10-0 m) and a whole water column tow (bottom-0 m --shown in Figure 3). In order to minimize net clogging, we changed our sampling regime 1993 through 1995 and sampled with the same epilimnetic tow, but then we used a closing zooplankton net to sample the metalimnion (30-10 m) and hypolimnion (bottom-30 m) in Redfish, Alturas, and Pettit Lakes.

Chlorophyll a and zooplankton

Crustacean zooplankton biomass showed little relationship when regressed as a function of epilimnetic chlorophyll *a* (Figure

4). Weak negative relationships (R^2<0.2) between zooplankton biomass and epilimnetic chlorophyll were observed in Redfish, Pettit, and Stanley lakes while Alturas showed an even weaker positive relationship. Egg ratios were always loe (< 3/ femaile)(Figure 5). Eggs per female showed weak positive relationships with epilimnetic chlorophyll for *Daphnia* (R^2 = 0.063), *Bosmina* (R^2 = 0.061), and *Holopedium* (R^2 = 0.002) whether the egg ratios were lagged or unlagged with chlorophyll *a* concentrations.

Fish and Zooplankton

The number of fish per hectare and the number of fish per 1000 m^3 for the four lakes and the four years are shown in Figure 6 *top* and *bottom*. The number of fish per hectare was highest in Alturas and Stanley Lakes in 1992 and 1993. In Stanley Lake in 1994, the fish density remained high but dropped dramatically in 1995. Similarly, Alturas showed much lower fish densities in 1994 and 1995 as compared to 1992 and 1993. Fish per hectare increased ~ 400% in 1995. In Pettit Lake while Redfish Lake changed little over the four years. When fish densities were calculated per volume (1000 m^3), a similar overall pattern was observed (Figure 6, *bottom*). Fish densities were greatest in Stanley and Alturas lakes in all years except 1995 when they were greatest in Pettit Lake after a strong recruitment of kokanee and the stocking of ~3000 juvenile sockeye from the brood stock program. Calculating fish density as a function of lake volume made the differences among Stanley Lake, the shallowest lake, and the three deeper lakes more pronounced.

Cladoceran zooplankton length and fish density showed a weak negative relationship when all the lakes were pooled (Figure 7, *top*), but when each lake was considered independently with fish densities calculated as a function of lake volume (Figure 7, *bottom*) distinct relationships emerged. Pettit Lake showed a strong negative relationship between the mean cladoceran length and fish density (R^2 = 0.988) and Alturas a weaker one (R^2= 0.237) (Figure 6, *bottom*). In Redfish and Stanley Lakes, however, cladoceran length and fish density were not correlated.

Part II: Redfish Lake Fertilization

Epilimnetic (6-0 m) chlorophyll *a* in

Redfish Lake was similar and not significantly different (see Appendix 1) in 1995, the year of fertilization, as compared to the other three years (Figure 8). Mean summer epilimnetic chlorophyll *a* (*u*g/L) ranged from a low of 0.39 in 1994 to a high of 0.69 in 1993 (Table 1). Chlorophyll *a* in 1995 began to increase steadily in late May until reaching a high of 0.8 (*u*g/L) in late August.

Although epilimnetic chlorophyll *a* showed little increase under fertilization, primary productivity (mg C/m^3/h) (mean depth integrated) in Redfish Lake, 1995, was ~ 100 % higher than in the other lakes (RFL= 1.21, ALT=0.54, PET=0.58, and STA= 0.76 *--see Chapter 2*). Further, if we compare mean primary productivity in Redfish Lake in 1995 with values from control limnocorrals in 1994 and lake values from 1993, again values were ~ 100% greater in 1995. Mean water column primary productivity ranged from 1.21 (mg C/m^3/ h) in 1995, to 0.61 in 1994, and 0.49 in 1993. This comparison likely overestimates rates of primary production in 1994 in that only the 0-17 m water column tow in control limnocorrals was included. Mean primary productivition for 1993 and 1995 include depths to 35 m.

Phytoplankton biovolume varied annually among lakes (Figure 9). In Redfish Lake, phytoplankton biovolume was greatest in 1993 and second highest in 1995. Species composition varied from equal representation of Chlorophytes, Chryso-Chryptophytes, and Bacillariophytes in 1992 to domination by Dinophytes in 1994. In 1995, under fertilization, Redfish phytoplankton biovolume was dominated by Cyanophytes with lesser representation from Chryso-Chryptophytes, Bacillariophytes, and Dinophytes. In Alturas, Pettit, and Stanley Lakes, phytoplankton biovolume was also greatest in 1993 and lowest in 1992. In these lakes, Chlorophytes and Chryso-Chryptophytes dominated with smaller proportions of Cyanophytes, Dinophytes, and Bacillariophytes for all three years.

Phytoplankton biovolume data was not available for Alturas, Pettit, and Stanley Lakes in 1994 due to emphasis on limnocorral fertilization experiments in Redfish Lake. It is also important to note that in 1995 we switched phytoplankton analyses techniques and counters. Based on an increase in resolution in 1995, we can expect the Chryso-Cryptophyte category to be underestimated in 1992, 1993, and 1994 as compared to 1995. Similarly, many of the extremely small picoplankton dominating the Cyanophytes in 1995 were not accounted for in 1992, 1993, and 1994.

Consistent with the small changes in chlorophyll *a* (Figure 8), water transparencies were only slightly shallower in 1995 in Redfish Lake as compared to the other three years (Figure 10). Secchi depth transparency showed the same pattern of increase throughout the summer for all four years sampled. Secchi depths were never less than 8 m -- levels still associated with high water transparency.

Total zooplankton biomass was generally similar in 1995 to biomasses in other years (Figure 11) until a substantial increase occurred in August and October. In 1995, zooplankton biomass in Redfish Lake was low in May and then increased gradually throughout the summer and exceeded 20 *ug*/L by October. In other years, zooplankton biomass in Redfish Lake was also highest near the end of summer, but values were never greater than 12 *ug*/L. Thus, the fertilization appeared to stimulate zooplankton production in late summer and fall months to a greater extent than observed in previous years.

In addition to fall increases in total zooplankton biomass, *Daphnia* biomass and density was greater st in 1995 as compared to the other three years (Figure 12). In 1992 and 1993, *Daphnia* density increased slightly through the summer but never exceeded 0.2 (#/L). In 1994, *Daphnia* density was considerably greater reaching 1.2/L by the end of the summer. In 1995, however, *Daphnia* biomass not only increased earlier in the season but was also dramatically higher than in 1992, 1993, or 1994 and reached a summer high in mid-October of 2 /L. *Daphnia* egg ratios (#eggs/adult) were similar in 1995 to other years (Figure 13). *Daphnia* egg ratios were greatest in late June and mid-July in 1992, 1993, and 1994. In 1995, however, *Daphnia* egg ratios were greatest in early August.

STATISTICS

Cluster Analysis

The complete linkage cluster analysis of the four years and four lakes of limnological data resulted in clusters composed primarily of observations from one lake in combination with a few isolated observations from other lakes. The complete cluster analysis is summarized in Table 2, and the seasonal means for the limnological variables used in these analyses are shown in Table 1. There are four distinct clusters with 2-5 observations in each cluster. Cluster #1 appears to be a cluster of extremely oligotrophic conditions with three years of Alturas and Pettit Lake in 1992, a low precipitation/ productivity year. Cluster #2 contains three years of Stanley Lake data, the most productive lake, in addition to Pettit in 1995 and Redfish in 1992. The Redfish 1992 point, however, was added in a final cluster step and thus is the least similar observation in the cluster. Cluster #3 contains three years of the Redfish Lake including 1995, the year of fertilization and Pettit Lake. Again, it is important to note that Redfish in 1995 clustered first with Pettit 1994, a moderately productive lake with high zooplankton and especially high *Daphnia* biomass. Redfish and Pettit Lakes are also similar in their deep water transparency and depth of 1% light level (Table 1). Finally, cluster #4 contains Alturas in 1993 and Stanley in 1995, a cluster that appears to be driven by cool epilimnetic temperatures and moderately high epilimnetic chlorophyll *a*.

The MDS analysis clustered lakes along two dominant axes (Figure 14, top). In general each lake was put in a unique cluster, and variation among years had small impacts on the placement in multvariate space. Axis 1 represented a gradient of zooplankton biomass with Alturas and Redfish Lakes scoring positive numbers (low zooplankton biomass), and Pettit and Stanley exhibiting negative values. Axis 2 represented a gradient of parameters associated with chlorophyll concentration and temperature (positve numbers represented high chloroophyll and temperatures). Interestingly, Pettit Lake in 1995 was clustered with Redfish and Alturas Lakes stressing the decrease in zooplantkon biomass in that year. When epilimnetic temperature and light extinction coefficient were taken out of the MDS analysis (Figure 14, bottom) Redfish Lake in 1995 shifts towards the Pettit and Stanley clusters, stressing the increased zooplankton biomass of Redfish Lake during the fertilization year.

The results from the MANOVAs of Secchi, chlorophyll *a*, and total zooplankton biomass as a function of year and the results from independent univariate F tests of these limnological variables as dependent on year are summarized in ***Appendix I***. The MANOVA for Redfish Lake showed both significant difference in limnological variables among the four years and a significant contrast of 1995 to 1992, 1993, and 1994 (P>0.10). The univariate F test for Redfish Lake was significant for Secchi depth differences among the four years, but no significant contrast of 1995 to the other years was observed (P>0.10). The overall difference among years appeared to be driven primarily by the extremely high water transparency in 1994. The univariate F tests for Redfish Lake showed no significant differences in mean epilimnetic chlorophyll *a* or zooplankton biomass among the four years (P>0.10). MANOVAs for Alturas, Pettit, and Stanley Lakes were all significantly different among the four years, and the contrast of 1995 to 1992, 1993, and 1994 was also significant for these lakes (P< 0.05). In Alturas, both Secchi and chlorophyll a were significantly different among the years and showed a significant contrast of 1995 to 1992, 1993, and 1994 (P<0.10). Exceptionally low chlorophyll associated with relatively shallow Secchi depths appeared to be responsible for this relationship. Zooplankton biomass in Alturas, however, was not significantly different among the four years (P>0.10).

In Pettit Lake, both Secchi depth and zooplankton biomass were significantly different among the four years and showed a significant contrast of 1995 to 1992, 1993, and 1994 (P<0.10). Zooplankton biomass in Pettit Lake was exceptionally low and Secchi depths were moderately shallow in 1995 which appeared to be responsible for these results. In Stanley Lake, Secchi was significantly different among the four years and in 1995 in contrast to the other three years (P<0.10). Chlorophyll a and zooplankton biomass were significantly different among the four years but <u>not</u> significantly different in 1995 as compared to the other years. Thus, the extremely low Secchi depth water transparency in 1995 in Stanley alone is driving the MANOVA's significant difference of 1995 in contrast with the other 3 years.

DISCUSSION

Overall lake productivity in these four lakes showed inter and intra-annual differences among the four lakes and the four years. The seasonal patterns of gradual increase in algal biomass and zooplankton biomass throughout the summer and decrease in late fall were generally consistent among the four lakes. Despite the oligotrophic nature of all the four lakes, there was a distinct gradient in productivity with Stanley and Pettit being the most productive lakes and Redfish and Alturas being the least productive lakes during 1992, 1993, and 1994. In 1995, however, Redfish Lake under fertilization had higher overall productivity than Pettit lake (second to Stanley Lake).

Annual variation in precipitation and other weather patterns had detectable effects on lake productivity. In 1992, a warm, drought year, Pettit and Redfish lakes were more similar to other lakes than they were to the observations of Redfish and Pettit in other years. Similarly, in 1993, a cool, high precipitation year, all the lakes had high concentrations of chlorophyll *a* but only moderately high concentrations of zooplankton. Further, Alturas Lake in 1993 and Stanley Lake in 1995 were clustered together presumably because they both remained very cool with epilimnetic temperatures never exceeding 12° C.

Temperature and precipitation appeared to affect phytoplankton and zooplankton populations in opposite ways. During cool, wet years, primary production and algal biomass were high (Table 1). Despite the greater amount of available food for herbivorous zooplankton during these years, zooplankton biomass was usually relatively low (1993, 1995). Zooplankton production appeared to be limited more by seasonal differences in temperature than by food availability. In 1992 and 1994, warm, dry years, zooplankton biomass was generally greater than during the other years.

Bottom-up effects of fertilization on the lakes were less apparent than top-down effects from planktivorous fish. We found little relationship between epilimnetic chlorophyll *a* and zooplankton biomass or production when averaged across lakes. This lack of relationship between time-lagged chlorophyll *a* over the entire season may be due to an inappropriate time scale and/or the dominating effects of temperature on egg production. We time-lagged this relationship based on the interval between sampling events, ~ 2 weeks, a time period that may be inappropriate given various physical factors. In these high mountain lakes, water temperatures do not increase substantially until late July. Thus the cold temperatures in June and early July may limit zooplankton production despite plentiful food resources.

Top-down effects of planktivorous fish on zooplankton explained some of the variation in annual zooplankton numbers among lakes and among years. In Alturas Lake, where fish densities were highest in 1992 and 1993, zooplankton numbers were extremely low. In Pettit Lake, fish densities increased several fold in 1995 after several thousand juvenile sockeye were stocked into the lake and a large natural recruitment of kokanee. This increase appears to be strongly associated with a dramatic drop in total zooplankton biomass in Pettit lake in 1995 as compare to the other years. Variation in fish density among lakes and years was also related to mean cladoceran length. Again, Pettit lake showed a strong negative relationship between fish density and mean cladoceran length, a relationship driven largely by the dramatic increase in fish density in 1995.

Changes in sampling regimes or techniques may also have affected overall differences in some variables among years. The change from whole water column tows in 1992 to depth stratified tows in 1993-1995 may have resulted in different relative differences in zooplankton biomass among the lakes. Similarly, the increase in resolution in phytoplankton analyses in 1995 may explain large differences in species composition within lakes and among the four years. Many of the small picoplankton dominating the Cyanophytes, in addition to species other than *Dinobryon* in the Cryso-cryptophytes, were included in 1995 but were not accounted for in 1992-1994.

The statistical analysis presented here helped describe and explain annual differences in limnological characteristics. Cluster analyses clustered lakes most often with observations from other years for that same lake. The few observations where one lake was clustered within a set of observations from another lake could usually be explained by anomalies in weather patterns (bottom-up effects) or large changes in fish density (top-down effects). Similarly, MANOVA's detected significant differences in lake productivity among all four years. It is important to note, however, that the statistical analyses presented here do not account for temporal effects. Statistical analyses that account for temporal trends, like repeated measures ANOVAS or time series analyses may yield different results. For example, when considering the total zooplankton biomass in Redfish Lake among the four years, the seasonal means were similar in all four years. However, zooplankton biomass was very low in spring but continued to increase into early fall of 1995 in Redfish Lake. This seasonal increase could be attributed to the fertilization, but mean zooplankton biomass for Redfish lake in 1995 was similar to previous years.

Overall, the whole-lake fertilization appeared to slightly decrease water transparency , had little effect on epilimnetic chlorophyll a concentration, dramatically increased primary production, and increased late summer zooplankton biomass. *Daphnia* density and biomass also increased significantly, and *Daphnia pulex*, a much larger zooplankter than the common *Daphnia rosea*, was observed for the first time in 1995 in Redfish Lake. Because large, cladoceran zooplankters like *Daphnia* are both preffered by juvenile sockeye and have a high energy density, the food base for juvenile sockeye in Redfish Lake was improved in addition to being increased (Lazzarro 1987; Luecke and Brandt 1993). Due to the time lag between initiation of fertilization and zooplankton response, juvenile fish benefit most from a mid- to late-summer stocking after fertilization began in spring.

Management strategies for the stocking of broodstock progeny should consider that these lakes appear to be especially sensitive to top-down control by planktivorous *O. nerka* salmon (Budy et al. 1994). Although zooplankton biomass was very high in Pettit Lake from 1992-1994, one large recruitment event resulted in zooplankton food resources declining to numbers paralleling those observed in plankton-poor Alturas Lake. The stocking of several thousand juvenile sockeye salmon into Redfish Lake over the next three years may therefore cause negative density-dependent effects on the zooplankton food resource and on the growth of the salmon. Results reported in this chapter, in addition to experiments from other years, suggest that whole-lake fertilization may increase the carrying capacity of these oligotrophic lakes and thus help overcome these potentially negative density effects on fish growth. The moderate responses to fertilization suggest, however, that higher levels of nutrients should be considered in the future. Our results also indicate that zooplankton biomass and species composition vary more in response to variation in fish density than to the modest changes in nutrient loadings acheived with the fertilization of Redfish Lake in 1995.

REFERENCES

Bevan, D., J. Harville, P.Bergman, T. Bjornn, J. Crutchfield, P. Klingerman, and J. Litchfield. 1994. Snake River salmon recovery team: final recommendations to National Marine Fisheries Service, Portland, OR. 375 p.

Budy, P.B., C. Luecke, W. Wurtsbaugh, and H. Gross. 1993. p. 3-28. *In* Spaulding, S. [ed.] Snake River Sockeye Salmon (Oncorhynchus nerka) habitat/ limnologic research, U.S. Department of Energy, Portland,OR. Project number 91-71. Contract number DE-B179-91BP22548.

------, ------, ------, and ------. 1995. Limnology of Sawtooth Valley Lakes with respect to potential growth of juvenile Snake River Sockeye Salmon. Northwest Science 69:133-150.

------, ------, ------, and ------. 1994. Effects of nutrient enhancement on juvenile sockeye salmon growth. Lake and Reservoir Management 9:140-145.

Carpenter, S.R. and J.F. Kitchell. 1988. Consumer control of lake productivity. BioScience 38: 764-769.

Carpenter, S.R. and J.F. Kitchell. 1992. Trophic cascade and biomanipulation: Interface of research and management -- A reply to the comment by DelMelo et al. Limnology and Oceanography 37:208-213.

DelMelo, R. R. France, D.J. McQueen. 1992. Biomanipulation: Hit or myth?. Limnology and Oceanography 37:192-207.

Devries, D.R. and R.A. Stein. 1992. Complex interactions between fish and zooplankton: quantifying the role of an open-water planktivore. Canadian Journal of Fisheries and Aquatic Sciences 49: 1216-1227.

Koenings, J.P. and R.D. Burkett. 1987. An aquatic Rubic's cube: Restoration of the Karluck Lake Sockeye Salmon (Oncorhynchus nerka), p.-419-434 *In* H.D. Smith,

L. Margolis, and C.C. Wood [ed.] Sockeye Salmon (Oncorhynchus nerka), population biology and future management. Canadian Special Publication of Fisheries and Aquatic Sciences 96.

Kyle, G.B. 1994. Assessment of trophic-level responses and coho salmon (*Oncorhynchus kitsutch*) production following nutrient treatment (1981-1986) of Bear Lake, Alaska. Fisheries Research 20:294-261.

-----, J.P. Koenings, and J.A. Edmundson. 1995. An overview of Alaska lake-rearing salmon enhancement strategy: Nutrient enrichment and juvenile stocking. *To be published as a chapter in a book:* Alaska Freshwaters.

Langeland, A. 1982. Interactions between zooplankton and fish in a fertilized lake. Holoartic Ecology 5:273-310.

Lazzarro, X. 1987. A review of planktivorous fishes: Their evolution, feeding behaviors, selectivities. and impacts. Hydrobiologia 146: 97-167.

Liebold, M.A. 1989. Resource edibility and the effects of predators and productivity on the outcome of trophic interactions. American Naturalist 134: 922-949.

Luecke, C. and D. Brandt. 1993. Estimating the energy density of daphnid prey for use with rainbow trout bioenergetics models. Transactions of the American Fisheries Society. 122:386-389.

Matson, P.A. and M.D. Hunter. 1992. Special Feature. The relative contributions of top-down and bottom-up forces in population and community ecology. Ecology 73(3):723-765.

McQueen, D.J., J.R. Post, and E.L. Mills. 1986. Trophic relationships in freshwater pelagic ecosystems. Canadian Journal of Fisheries and Aquatic Sciences 43: 1571-1581.

Nelson, P.R. and W.T. Edmundson. 1955. Limnological effects of fertilizing Bare Lake, Alaska. Fishery Bulletin 102:413-436.

Peterson, B.J., and 16 others. 1993. Biological responses of a tundra river to fertilization. Ecology 74: 653-672.

Polis, G.A., C.A. Myers, and R. Holt. 1989. The evolution and ecology of intraguild predation: competitors that eat each other. Annual Review of Ecology and Systematics 20:297:330.

Rieman, B.E. and D.L. Meyers. 1992. Influence of fish density and relative productivity on growth of kokanee in ten oligotrophic lakes and reservoirs in Idaho. Transactions of the American Fisheries Society 121:178-192.

Shortreed, K.S. and J.G. Stockner. 1990. Effect of nutrient additions on lower trophic levels of an oligotrophic lake with a seasonal deep chlorophyll maximum. Canadian Journal of Fisheries and Aquatic Sciences 47:262-273.

Stockner, J.G. 1981. Whole-lake fertilization for the enhancement of sockeye salmon (Oncorhynchus nerka) in British Columbia, Canada. Verhandlungen Internationale Vereinigung Limnologie 21:293-299.

----- and K.D. Hyatt. 1984. Lake Fertilization: State of the art after 7 years of application. Canadian Technical Report of Fisheries and Aquatic Sciences. No. 1324.

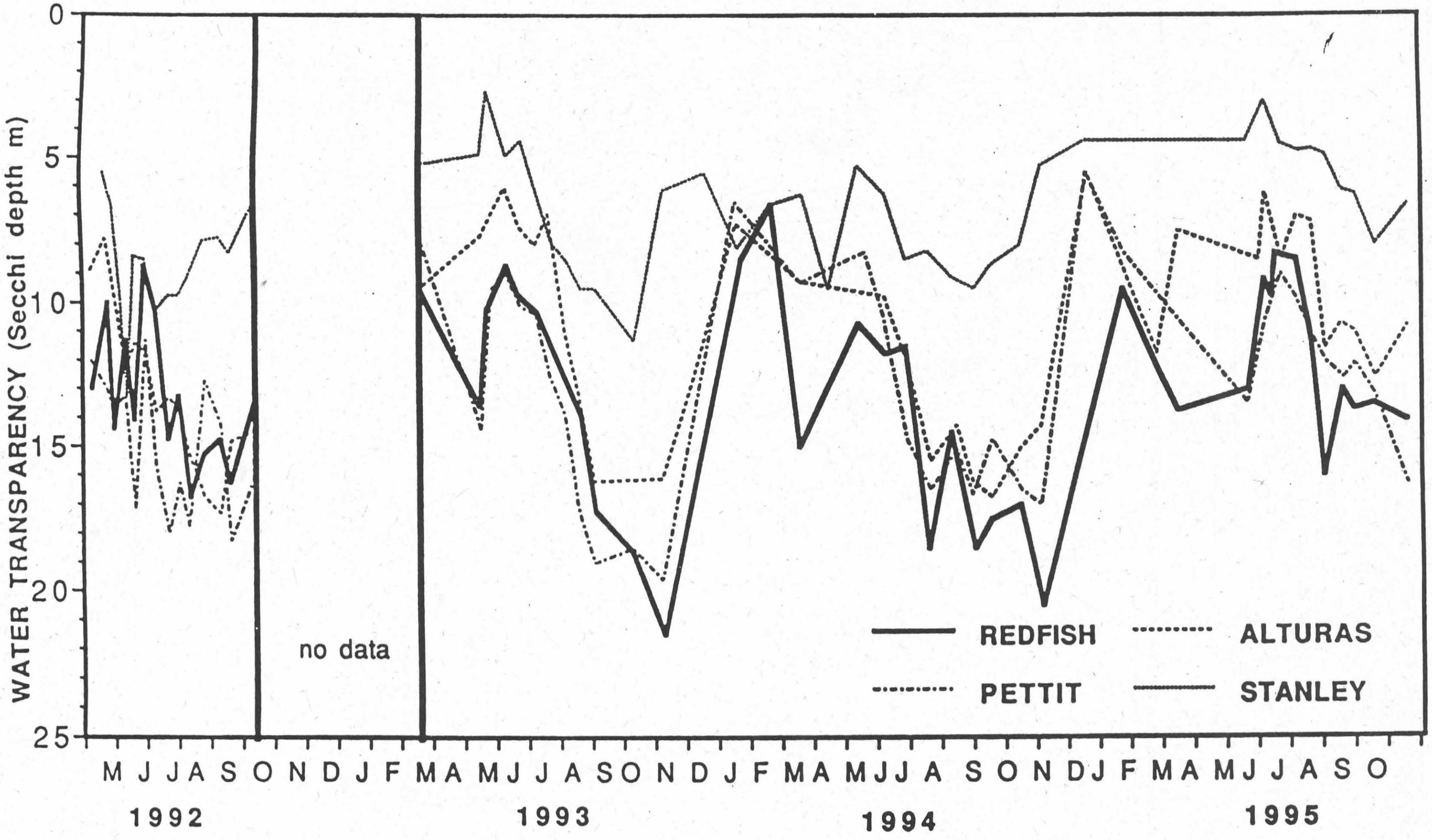
----- 1987. Lake fertilization: The enrichment cycle and lake sockeye salmon (Oncorhynchus nerka) production, p. 198-215. *In* H.D. Smith, L. Margolis, and C.C. Wood [ed.] Sockeye Salmon (Oncorhynchus nerka), population biology and future management. Canadian Special Publication of Fisheries and Aquatic Science 96.

Table 1. Limnological varables used in statistical analyses for the four years 1992-1995 and the four lakes, RFL= Redfish lake, STA= Stanley, PET= Pettit, ALT= Alturas. Values shown are the seasonal mean from sampling beginning in May and ending in October. The number of observations used to calculate the mean ranged from 8-14. Depth of 1% light and Secchi depth are measured in (m), epilimnetic chlorophyll *a* and zooplankton biomass are measured in *u*g/L, and epilimnetic temperature is measured in °C.

LAKE	YEAR	1 % LIGHT	SECCHI	EPI CHL *a*	ZOOP BIOM	EPI TEMP
RFL	1992	32.4	13.31	0.49	4.7	14.43
ALT	1992	24.39	12.95	0.53	3.2	14.31
PET	1992	29.26	14.98	0.44	30.74	14.94
STA	1992	18.05	8.6	0.83	29.66	14.17
RFL	1993	25.97	12.64	0.69	5.10	12.22
ALT	1993	20.79	9.54	0.93	0.93	11.8
PET	1993	23.06	13.46	0.61	18.75	12.68
STA	1993	15.08	6.96	1.25	18.91	11.12
RFL	1994	30.31	14.99	0.39	8.64	14.04
ALT	1994	23.13	14.21	0.48	3.76	13.4
PET	1994	30.18	14.11	0.34	30.17	14.52
STA	1994	15.47	7.94	0.49	23.75	14.13
RFL	1995	27.7	11.83	0.45	8.04	12.94
ALT	1995	17.45	9.41	0.4	2.14	11.98
PET	1995	23.08	12.02	0.44	3.83	12.73
STA	1995	11.11	5.2	0.89	17.05	11.44

Table 2. Summary of the complete cluster analysis showing the observations included in each cluster. RFL = Redfish Lake, STA= Stanley Lake, PET= Pettit Lake, and ALT= Alturas Lake.

Cluster 1	Cluster 2	Cluster 3	Cluster 4
ALT 1992	STA 1992	RFL 1993	ALT 1993
ALT 1994	STA 1993	RFL 1994	STA 1995
ALT 1995	STA 1994	RFL 1995	
PET 1992	RFL 1992	PET 1993	
	PET 1995	PET 1994	



92-95ALLSECHHI.g

Figure 1. Secchi depth water transparencies (m) in Redfish, Alturas, Pettit, and Stanley Lakes from May of 1992 through October of 1995. Note reversed Y-axis with 0 at the top of the graph.

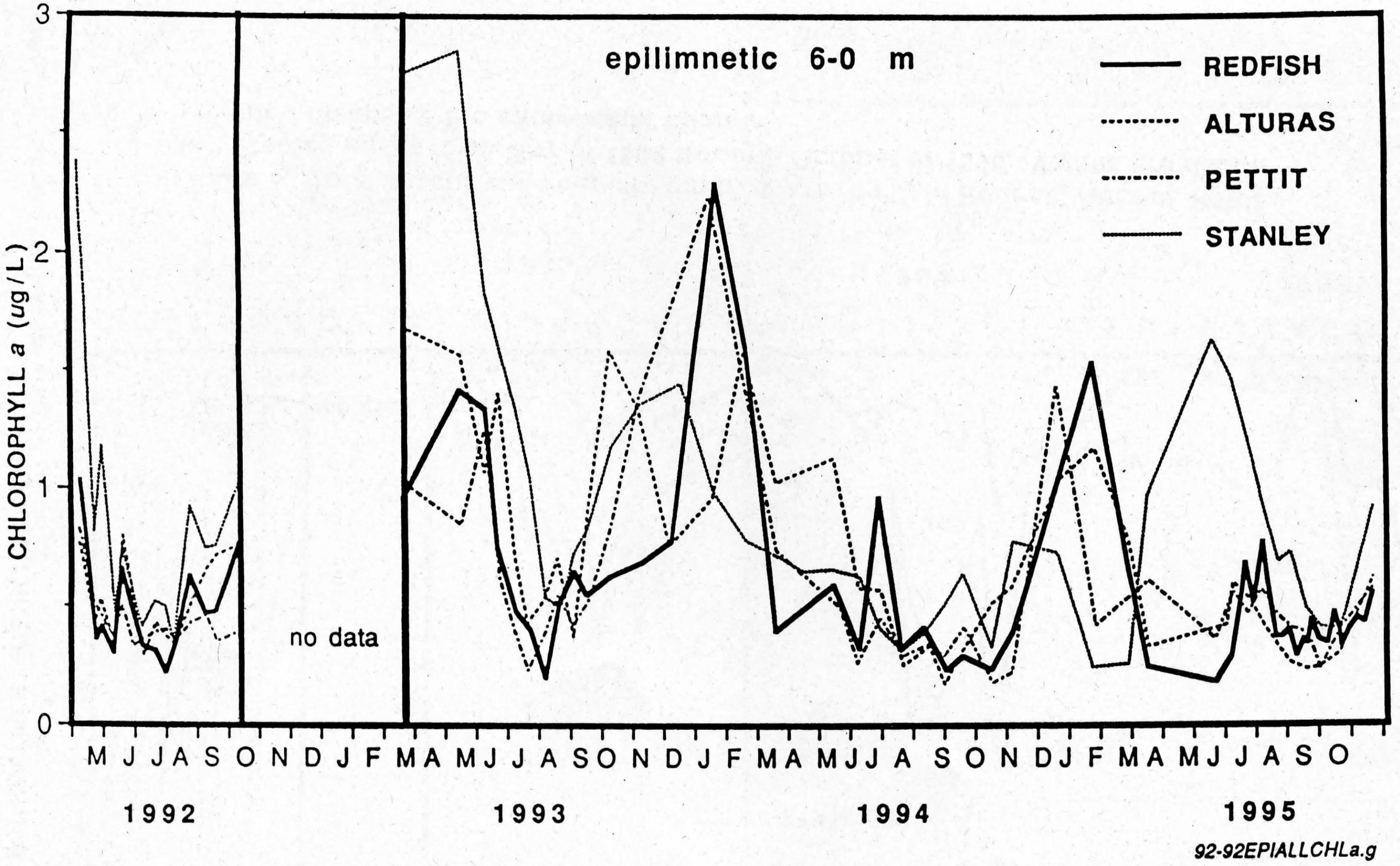


Figure 2. Epilimnetic (6-0 m) chlorophyll *a* concentrations (*ug*/L) in Redfish, Alturas, Pettit, and Stanley Lakes from May of 1992 through October of 1995.

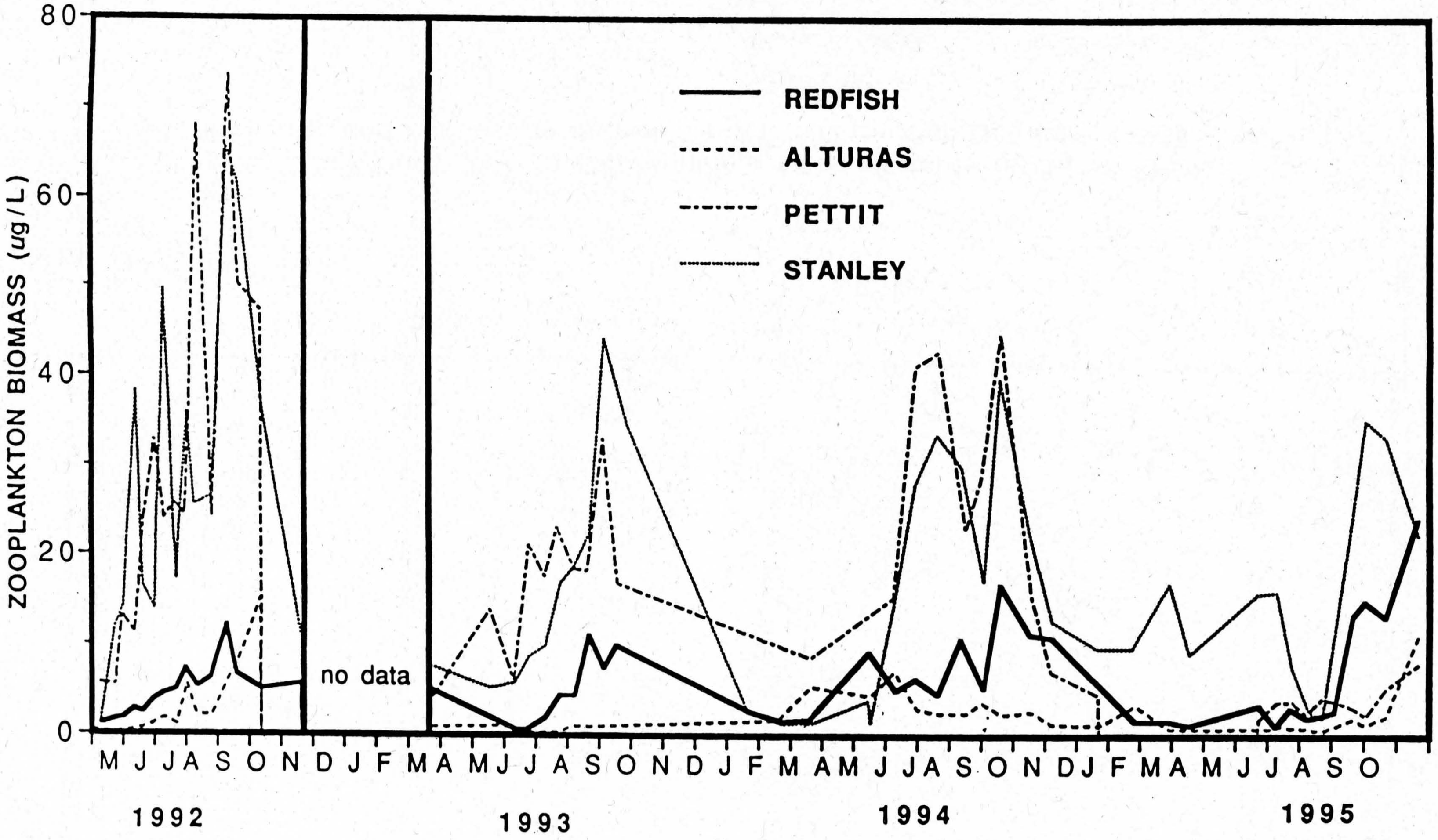
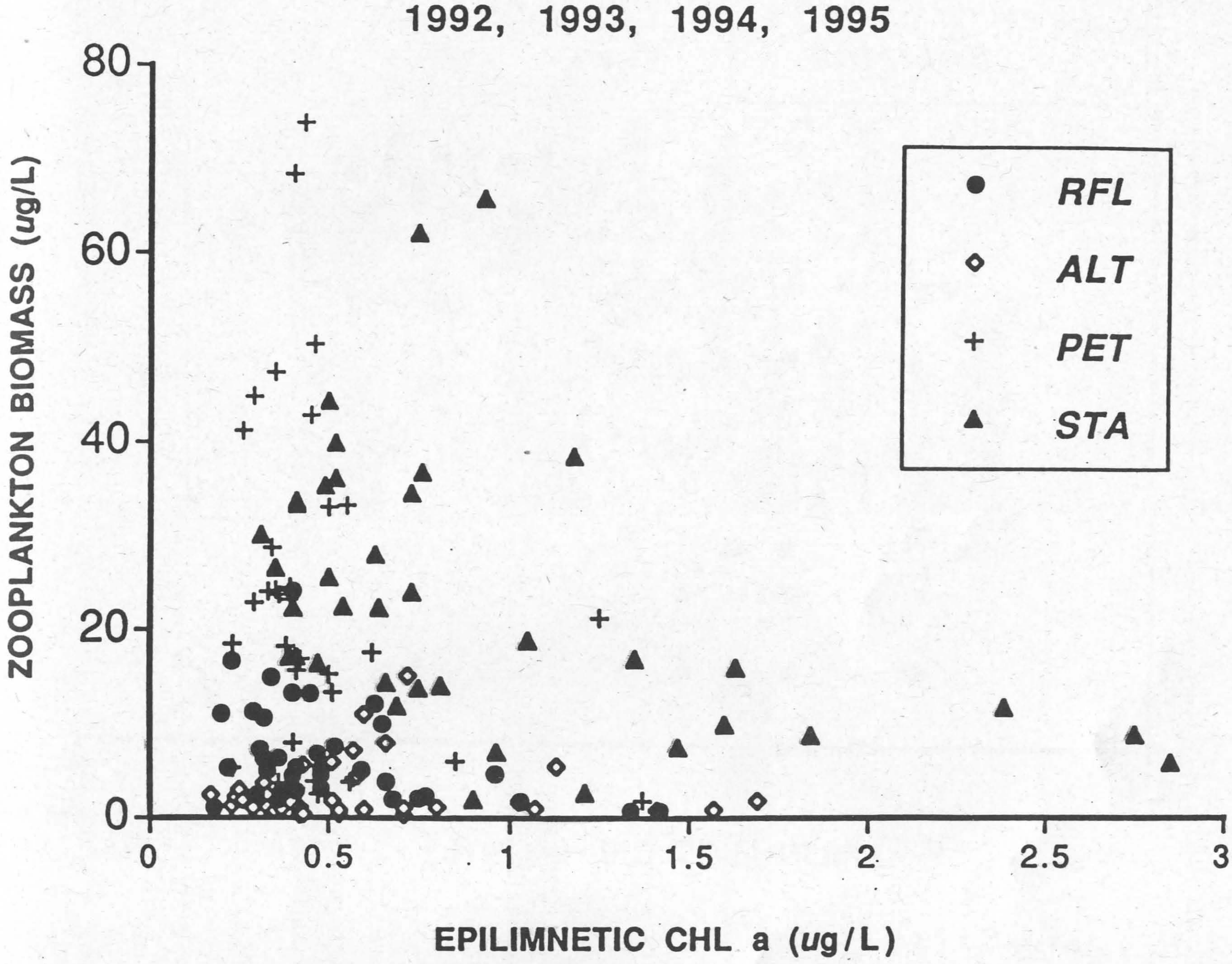


Figure 3. Total crustacean zooplankton biomass (*ug*/L) in Redfish, Alturas, Pettit, and Stanley Lakes from May of 1992 through October of 1995. Values are depth-weighted means of the entire water column.



$y = -7.104x + 10.246 \quad r^2 = 0.155$ RFL

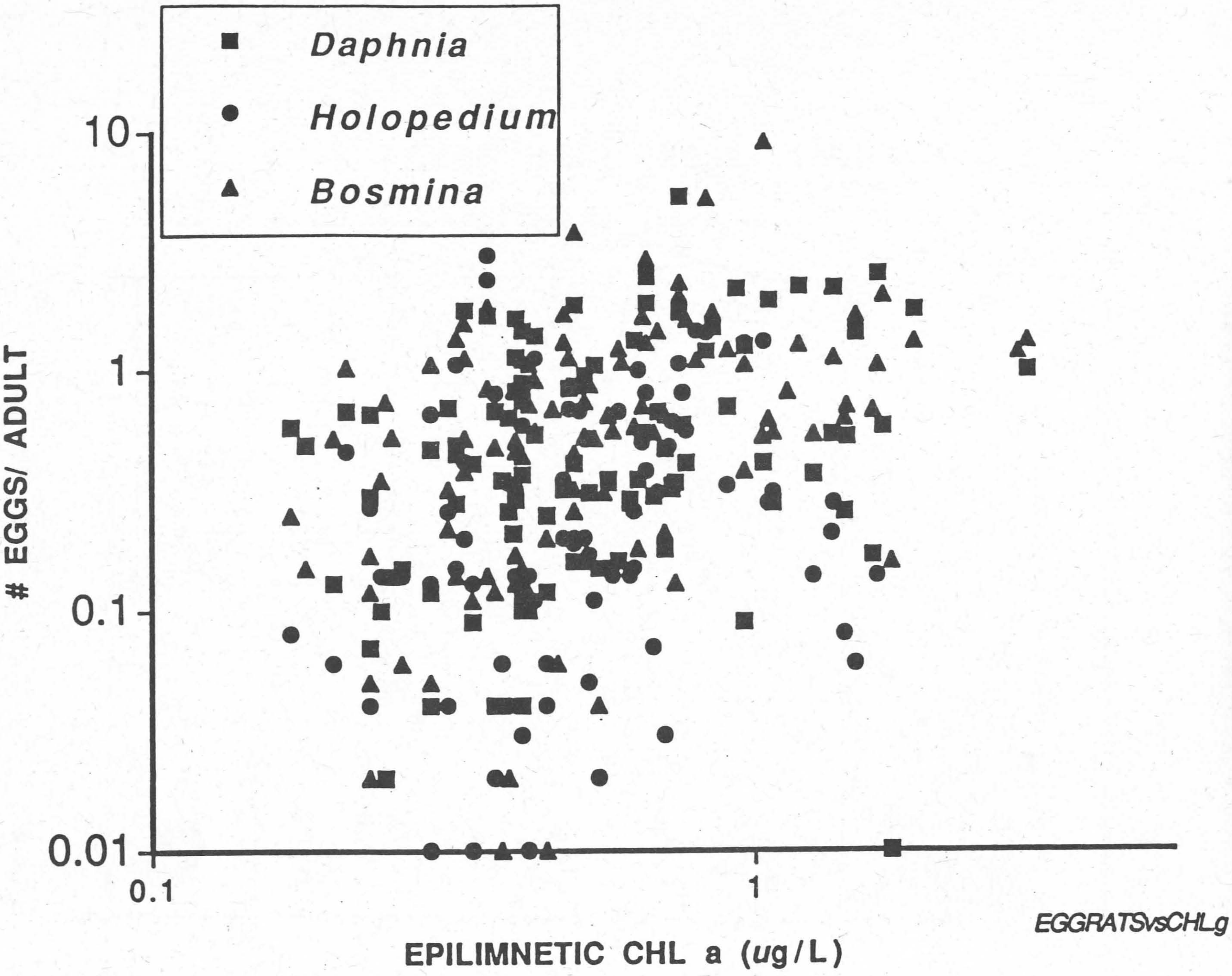
$y = 0.268x + 2.504 \quad r^2 = 0.001$ ALT

$y = -18.006x + 30.974 \quad r^2 = 0.055$ PET

$y = -10.124x + 32.643 \quad r^2 = 0.192$ STA

92-95AL:LCHLvsZOOP.g

Figure 4. Total crustacean zooplankton biomass (*ug*/L) as a function of epilimnetic (6-0 m) chlorophyll *a* (*ug*/L) in Redfish, Alturas, Pettit, and Stanley Lakes, 1992-1995, using limnological data from May until October, 1992-1995. Linear regressions of zooplaknton biomass as a function of chlorophyll are shown for each lake independently. Zooplankton biomass was time lagged behind chlorophyll by sample period, ~ 14 days for graphs and regressions.



y = 0.450x + 0.449 $r^2 = 0.063$

y = 0.076x + 0.351 $r^2 = 0.002$

y = 0.587x + 0.462 $r^2 = 0.061$

Figure 5. Cladoceran zooplankton egg ratios (eggs/adult) from May to October as a function of epilimnetic (6-0 m) chlorophyll *a* (*u*g/L) in Redfish, Alturas, Pettit, and Stanley Lakes (1992-1995). Data are shown on logged axis for clarity. Linear regressions of egg ratios as a function of chlorophyll are shown for each zooplankton species independently. Cladoceran egg ratios were time lagged behind chlorophyll by sample period, ~ 14 days for graphs and regressions.

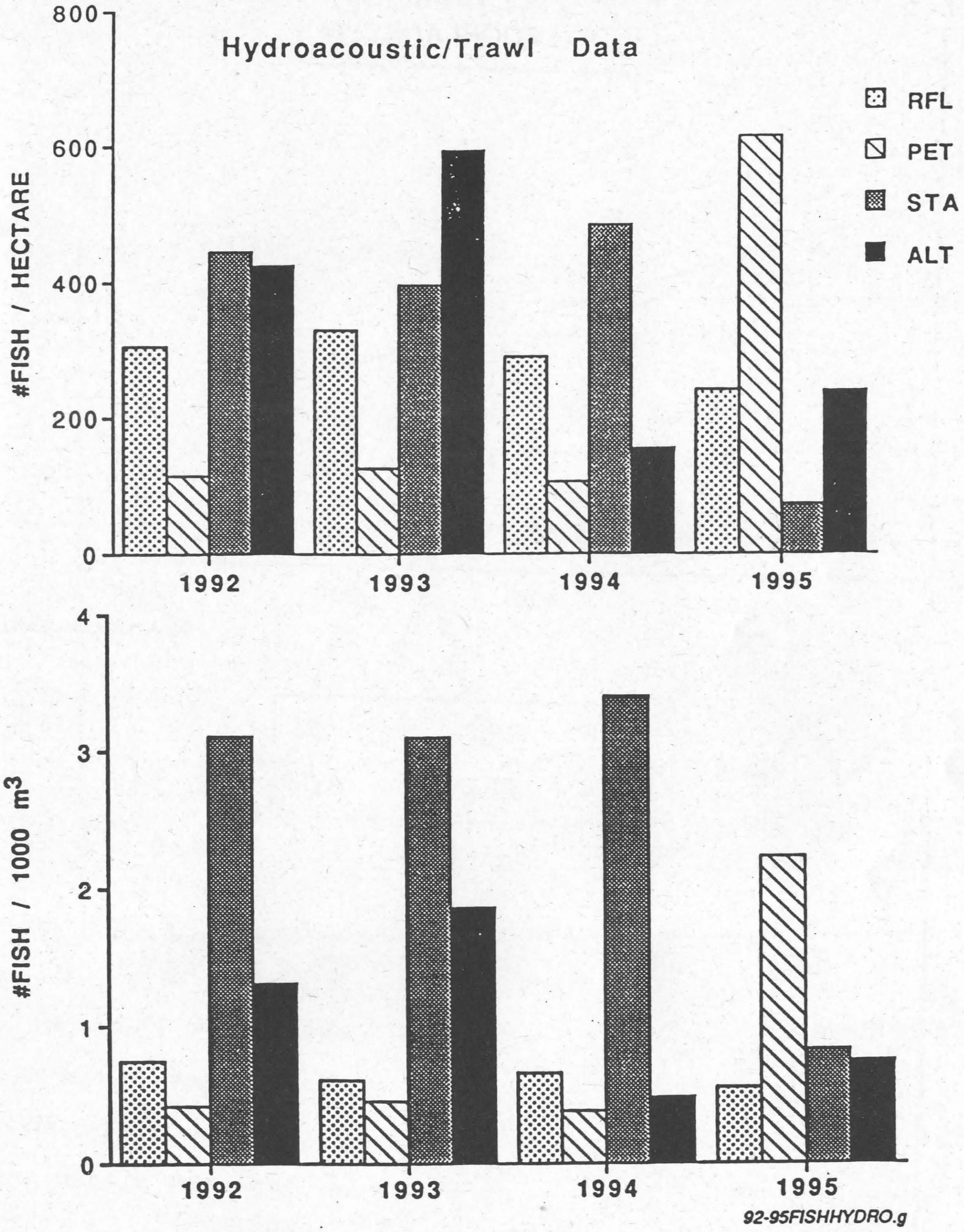
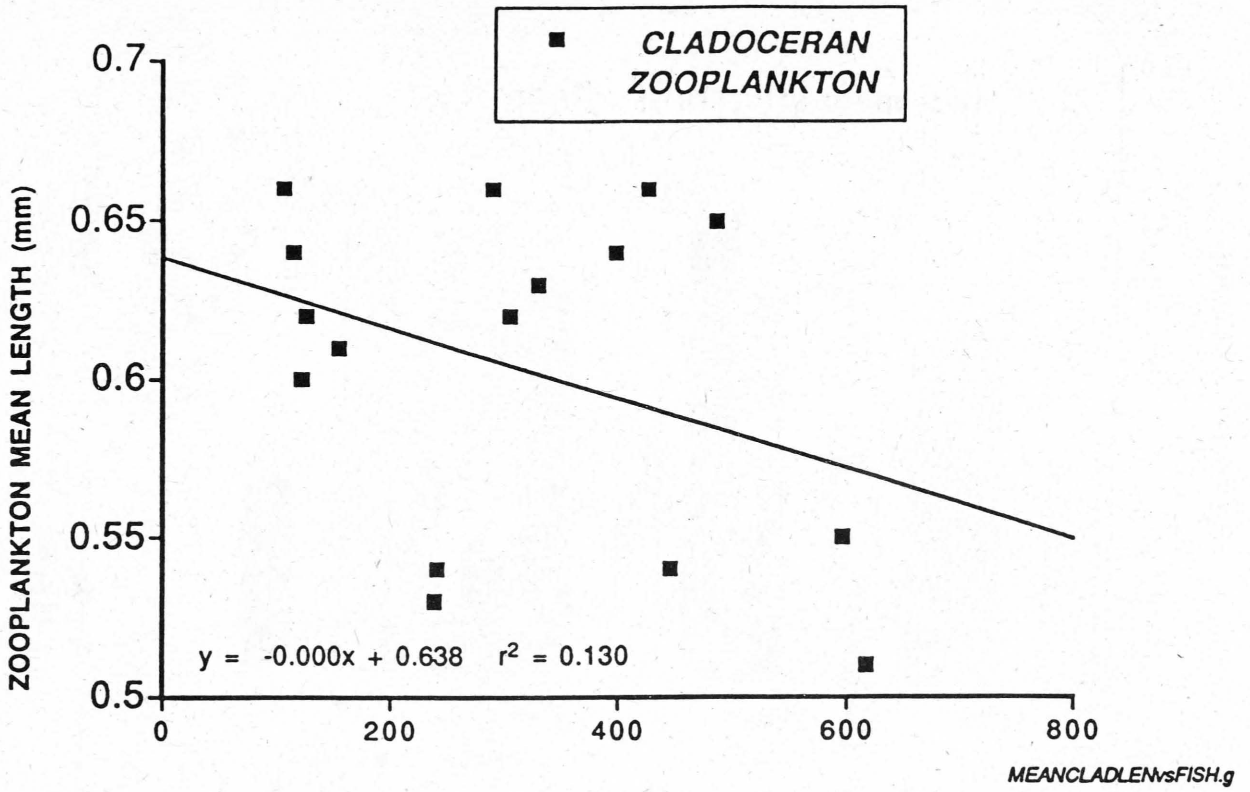


Figure 6. Fish density by lake area (#/ hectare) *(top)* and fish density by lake volume (#/ 1000m3) *(bottom)* from combined data from trawl and hydroacoustic estimates for Redfish, Alturas, Pettit, and Stanley Lakes in 1992, 1993, 1994, and 1995. Fish densities for 1994 and 1995 are from acoustic surveys (Taki et al. 1995, *pers. comm.*).



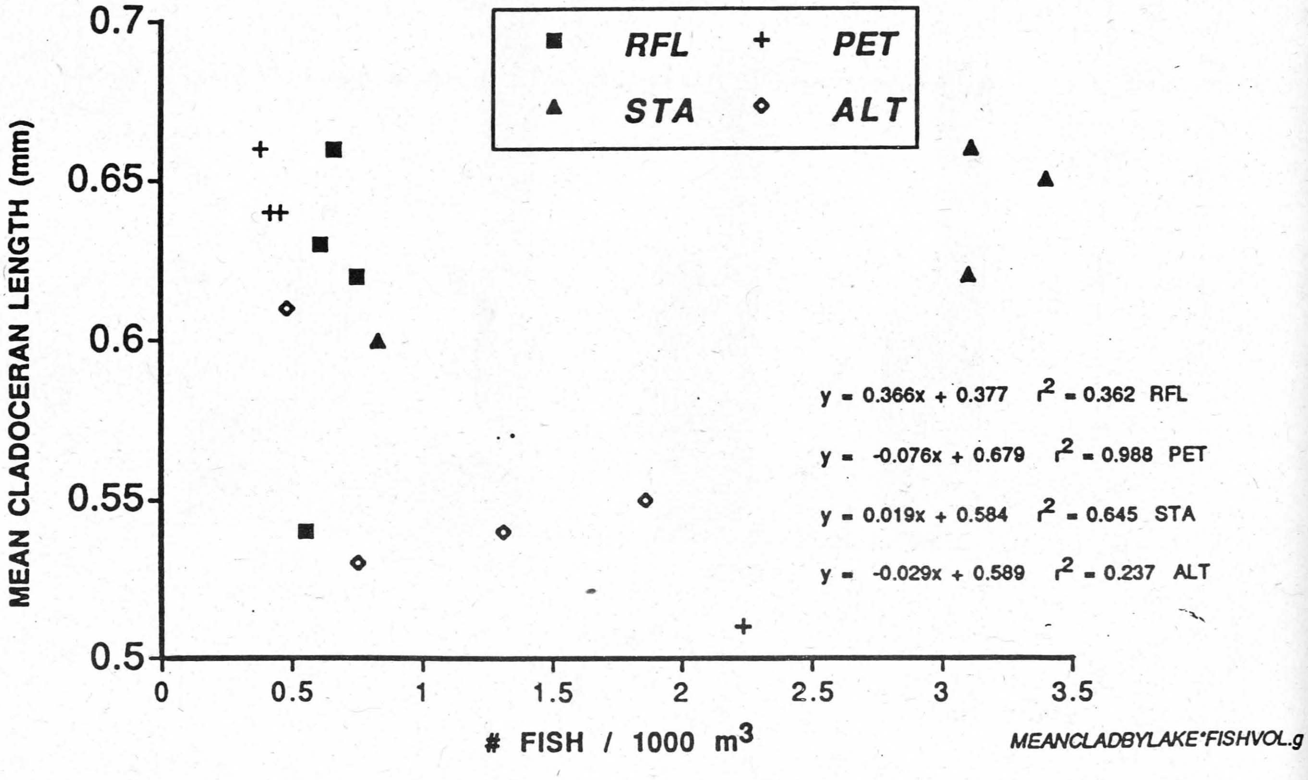


Figure 7. *Top:* Cladoceran zooplankton mean length (mm) as a function of fish density (# fish/ hectare) in Redfish, Alturas, Pettit, and Stanley Lakes in 1992, 1993, 1994, and 1995. A linear regression is shown for all lakes and years combined. *Bottom:* Cladoceran zooplankton mean length (mm) by lake as a function of fish density (# fish/ 1000 m^3) in Redfish, Alturas, Pettit, and Stanley Lakes in 1992, 1993, 1994, and 1995. Linear regressions are shown for each lake independently.

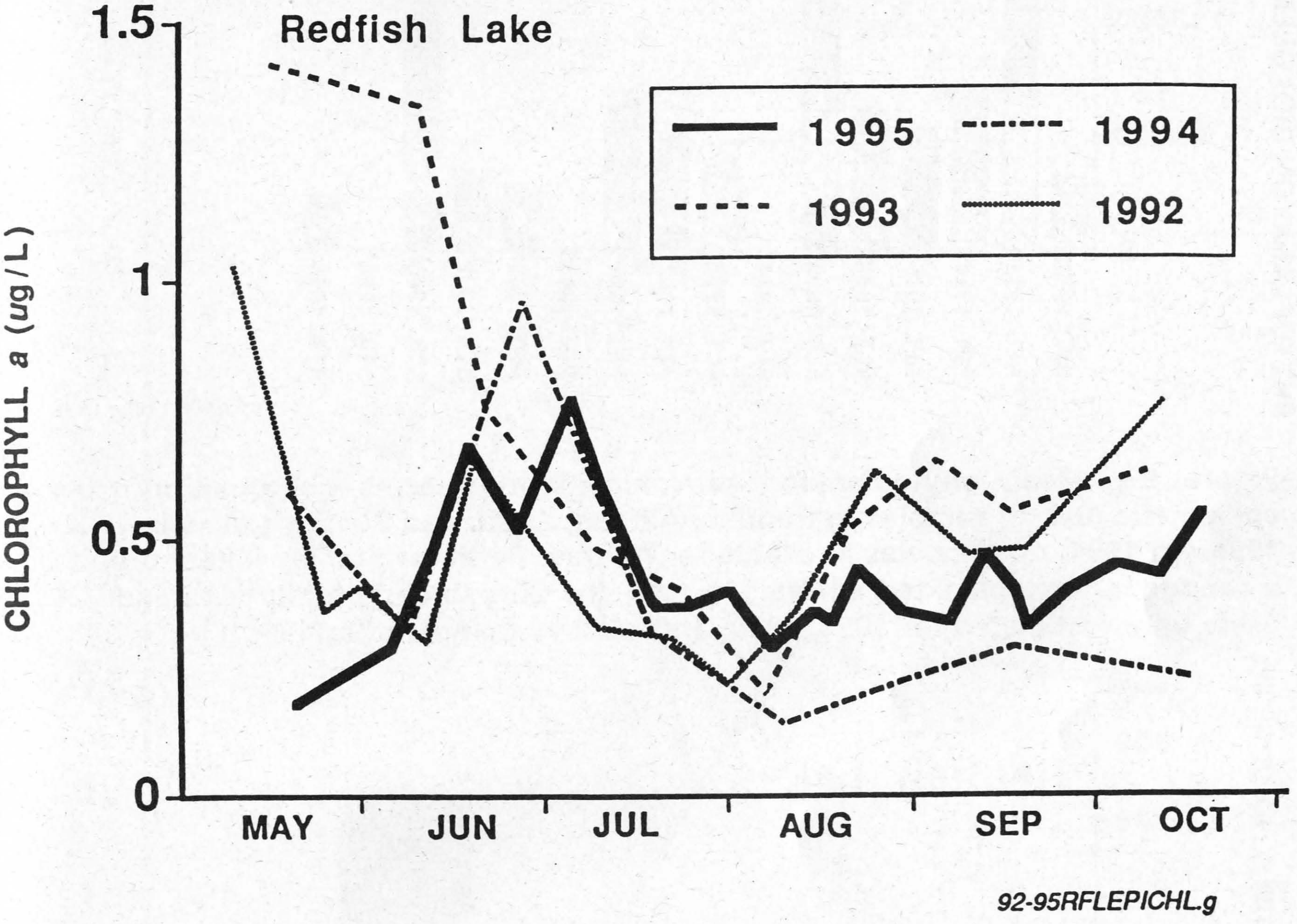


Figure 8. Epilimnetic (6-0 m) chlorophyll *a* (*u*g/L) in Redfish from May until October, 1992- 1995.

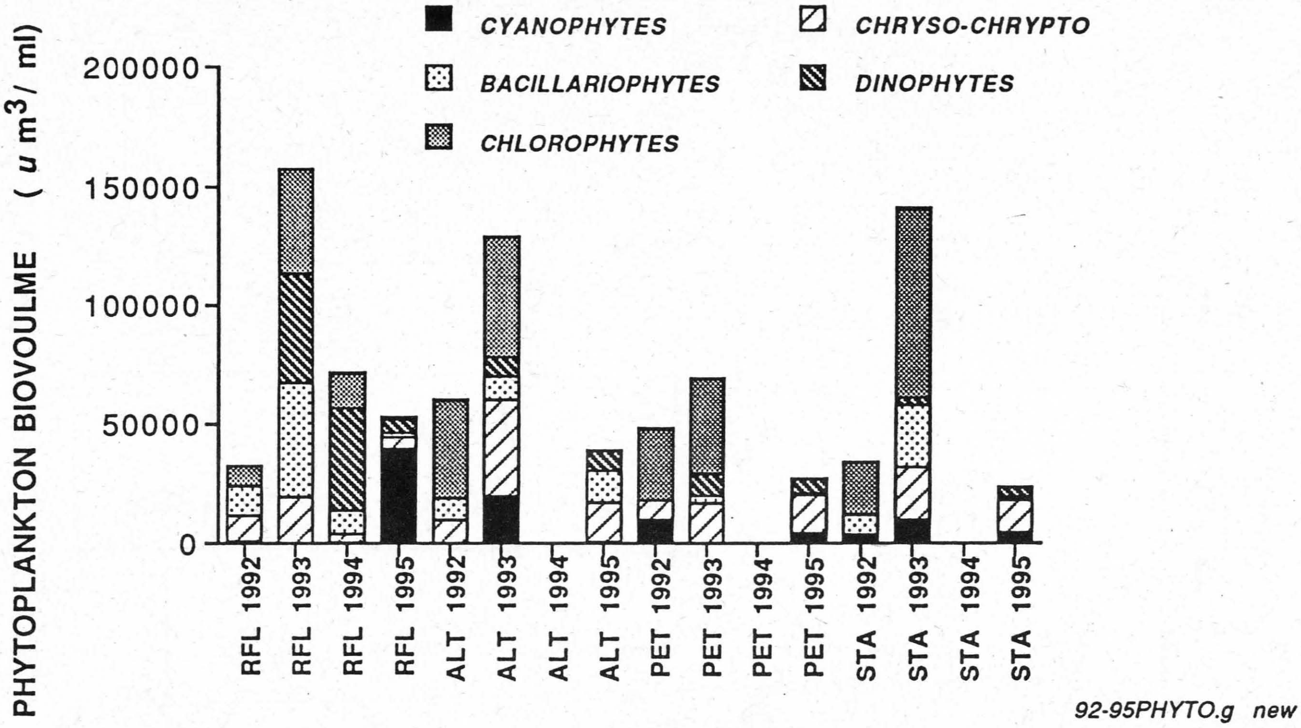


Figure 9 a. Mean phytoplankton biovolume and species composition from epilimnetic (6-0 m) samples in Redfish, Alturas, Pettit, and Stanley Lakes in 1992-1995. In 1994, no data was available for Alturas, Pettit, or Stanley lakes. Due to a change in phytoplankton analysis in 1995, the Chryso-chryptophyte category is likely underestimated for 1994, 1993, and 1992 as compared to 1995.

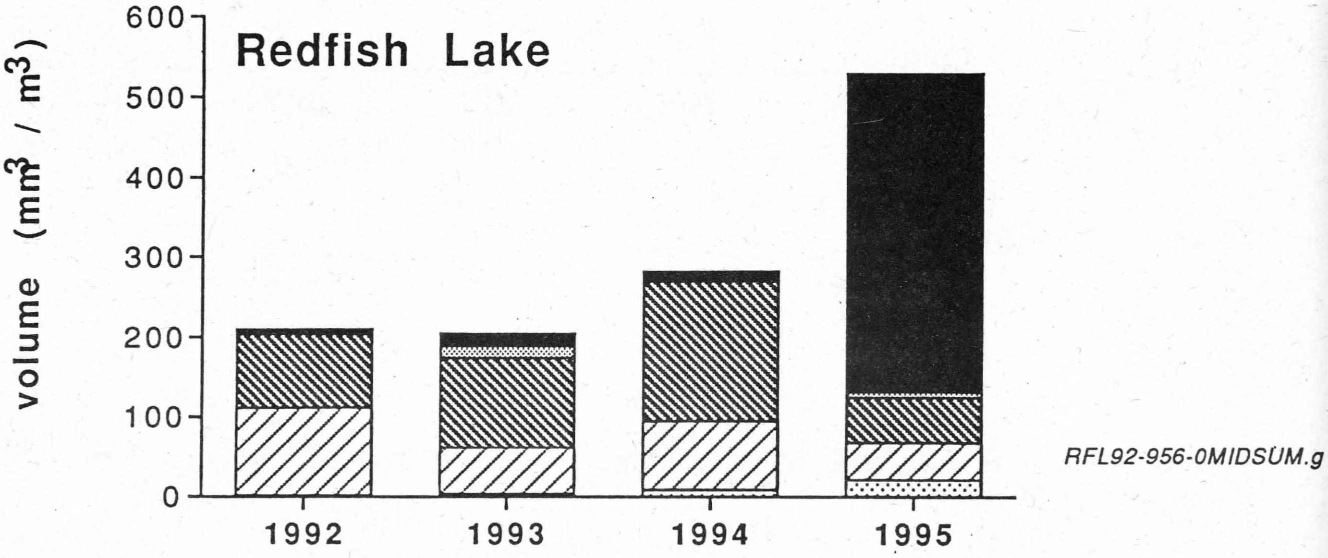


Figure 9 b. Epilimnetic phytoplankton (6-0 m) from late July, early August samples for Redfish Lake in 1992- 95. These samples (and all 1995 phytoplankton samples) were counted by EcoLogic Ltd. in contrast to those counted by Thorsten Blenckner, Utah State University above (9 b.) and in past reports.

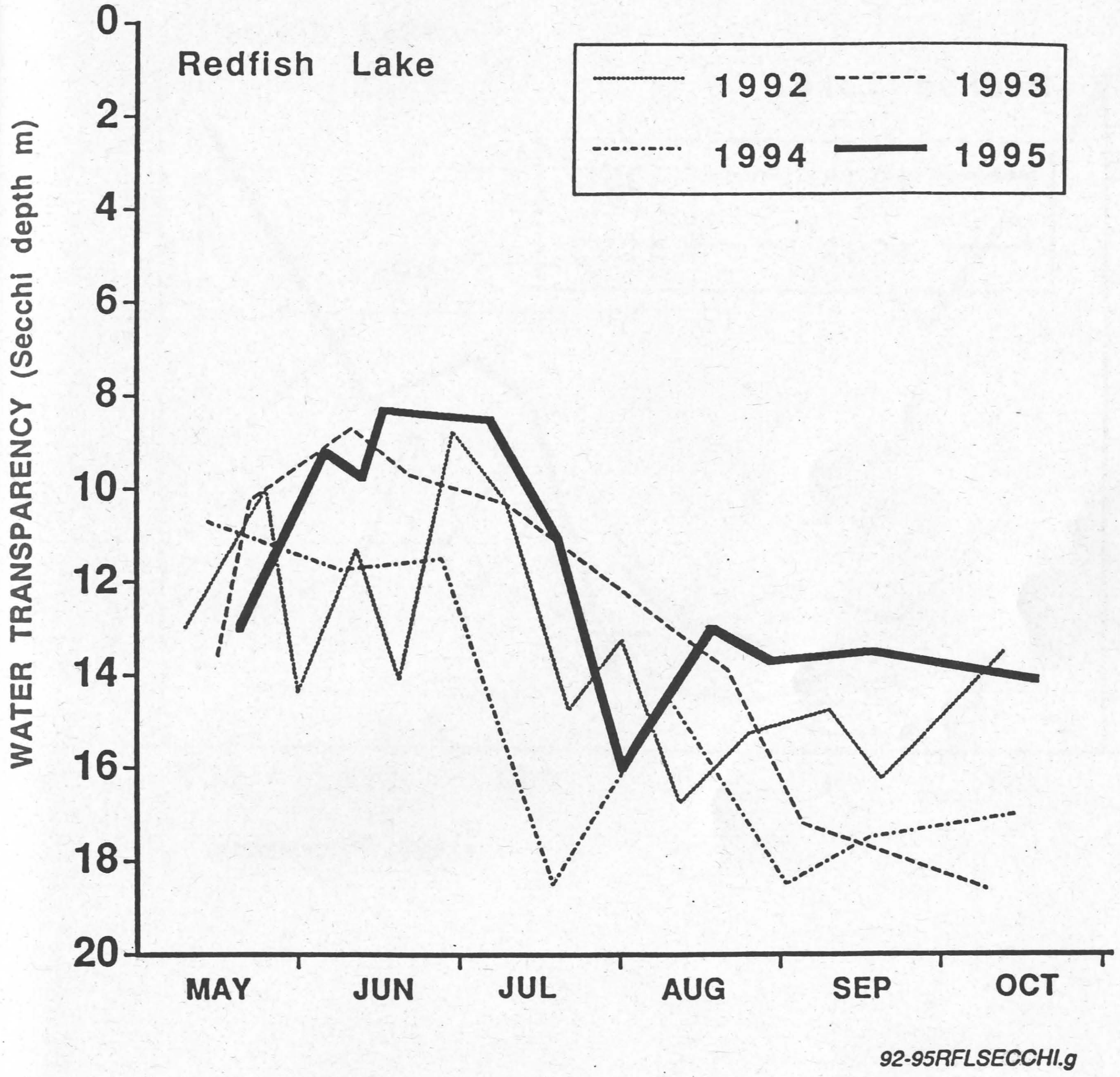
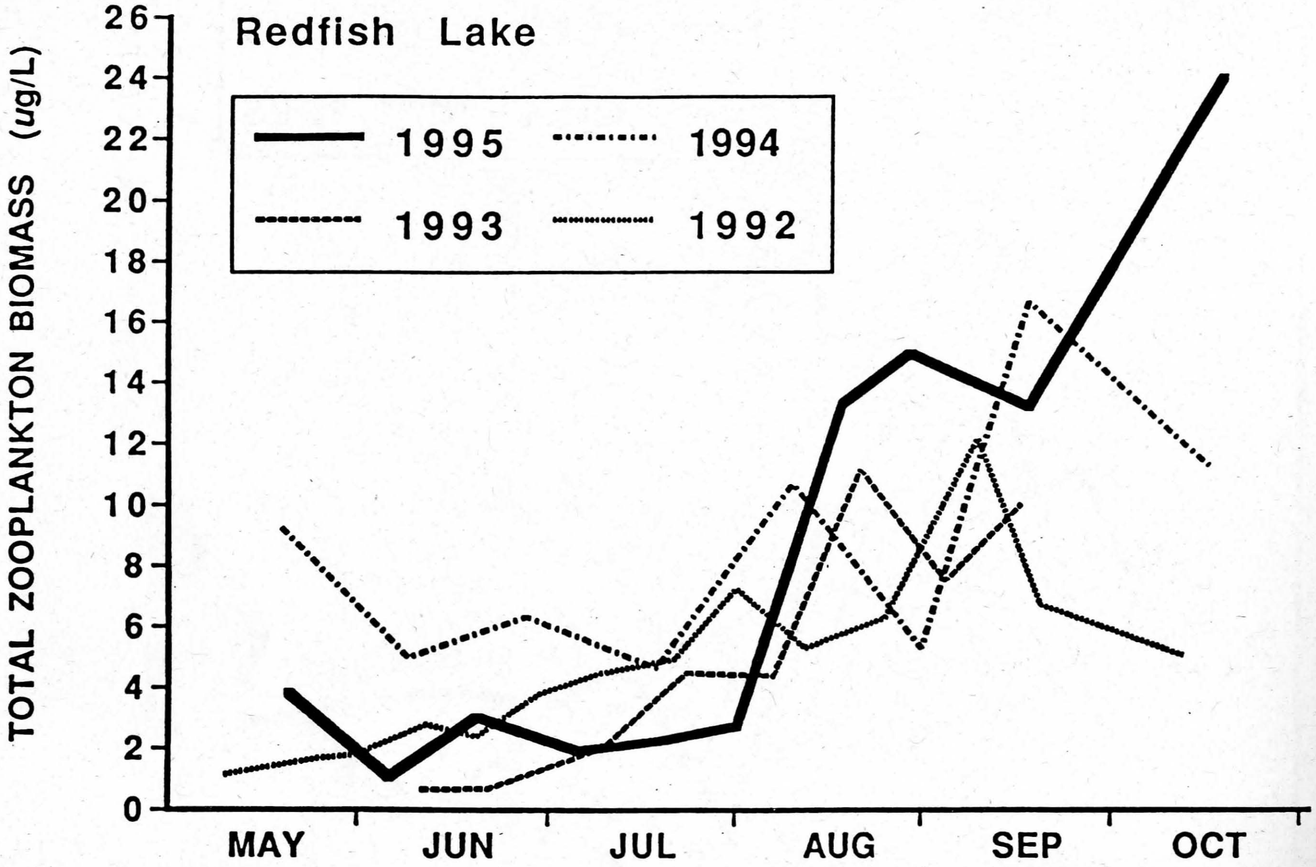


Figure 10. Secchi depth water transparency (m) in Redfish from May until October during 1992-1995.



92-95RFLTOTZOOPBIOM.g

Figure 11. Total crustacean zooplankton biomass (*ug*/L) in Redfish Lake from 1992 to 1995. Values shown are depth-weighted means for the entire water column.

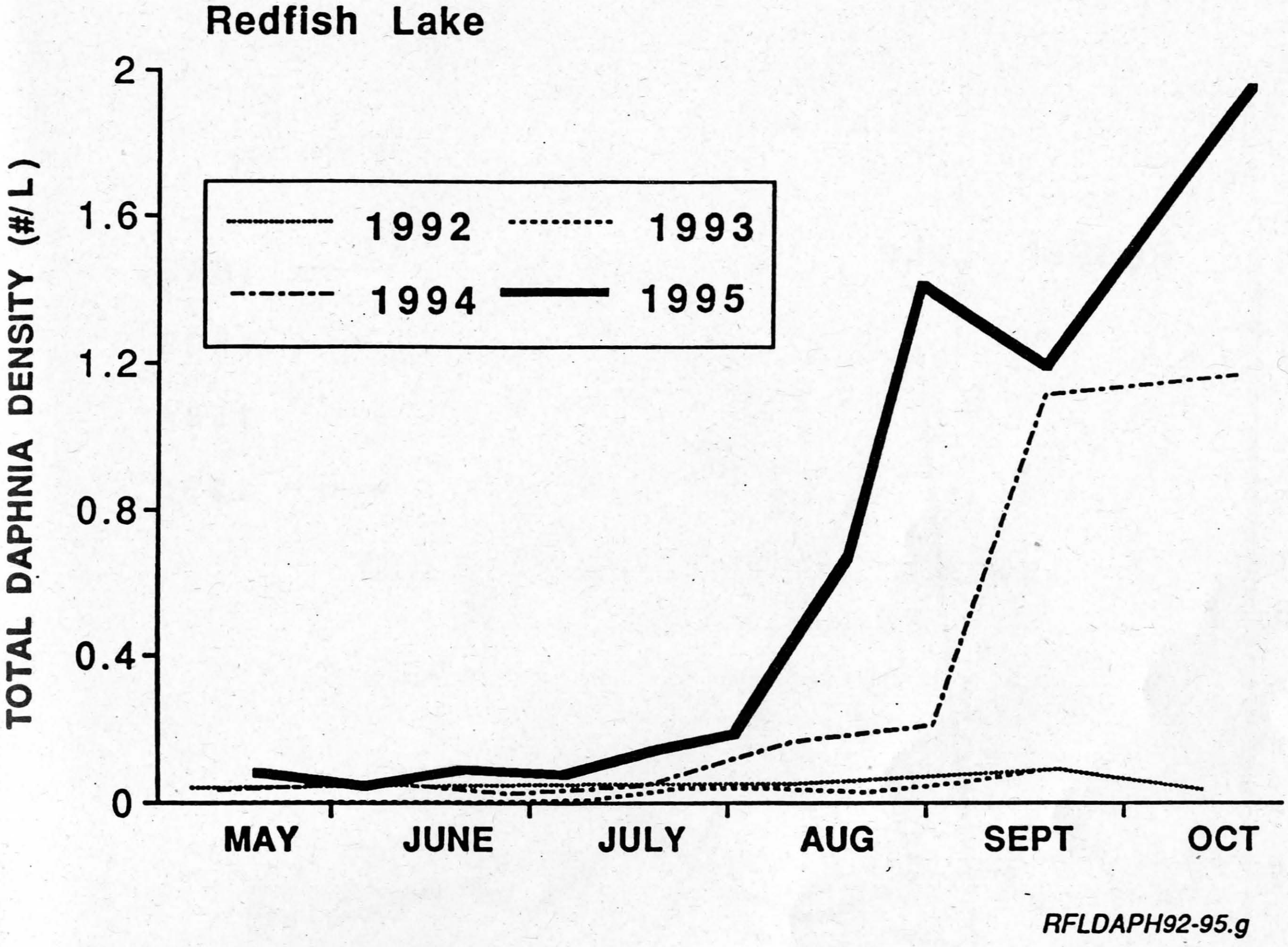
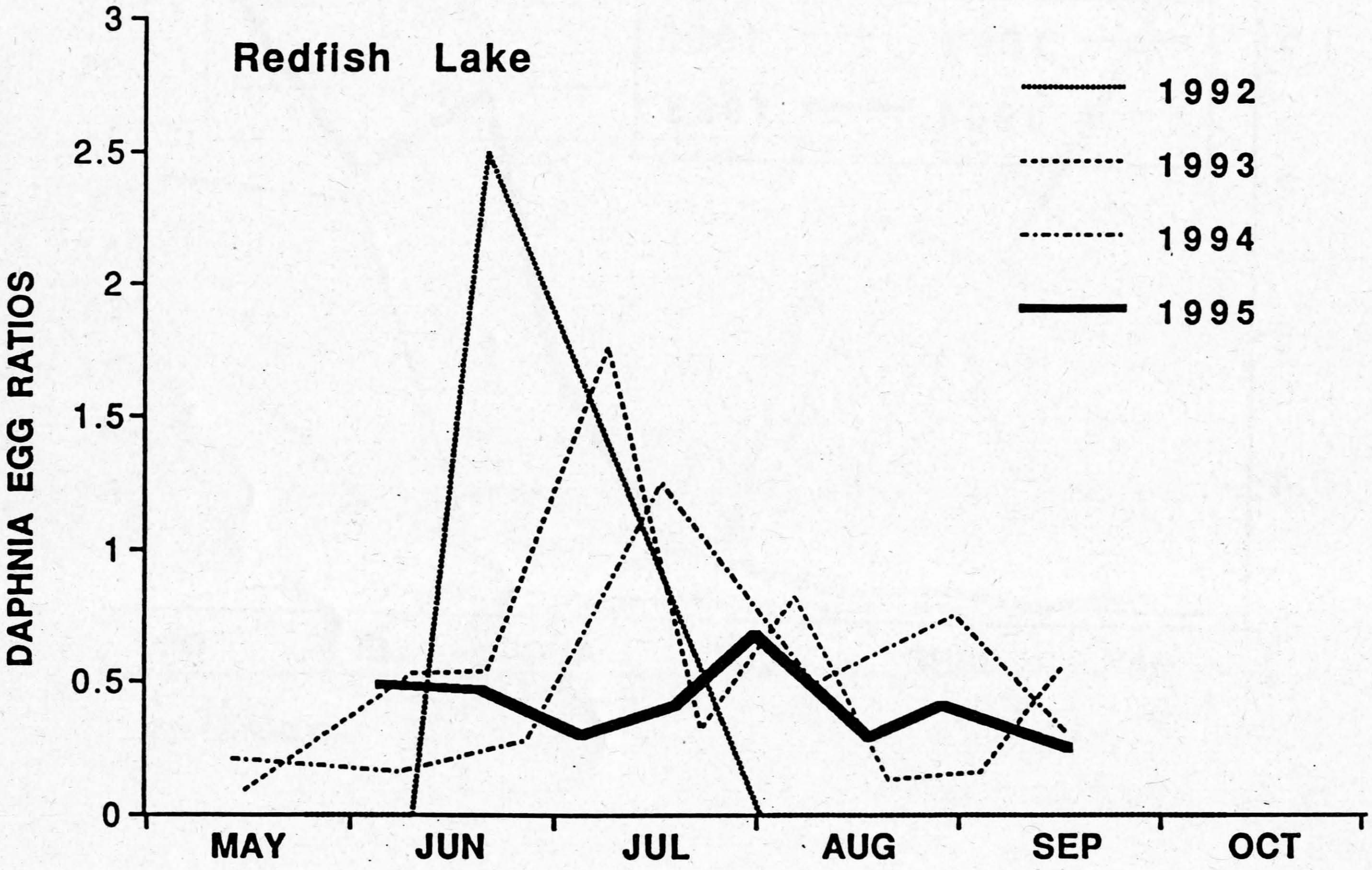


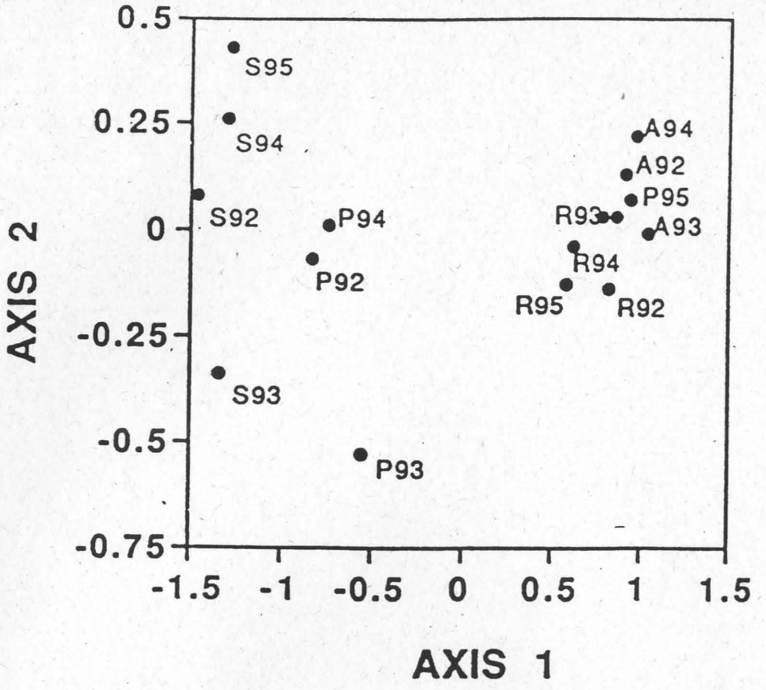
Figure 12. *Daphnia* denity (#/L) in Redfish from May until October, 1992-1995.



92-95RFLDAPHEGGRAT.g

Figure 13. *Daphnia* egg ratios (eggs/adult) in Redfish Lake from May until October, 1992-1995.

MULTIPLE DIMENSIONAL SCALING (ALL VARIABLES)



MULTIPLE DIMENSIONAL SCALING
(SELECTED VARIABLES)

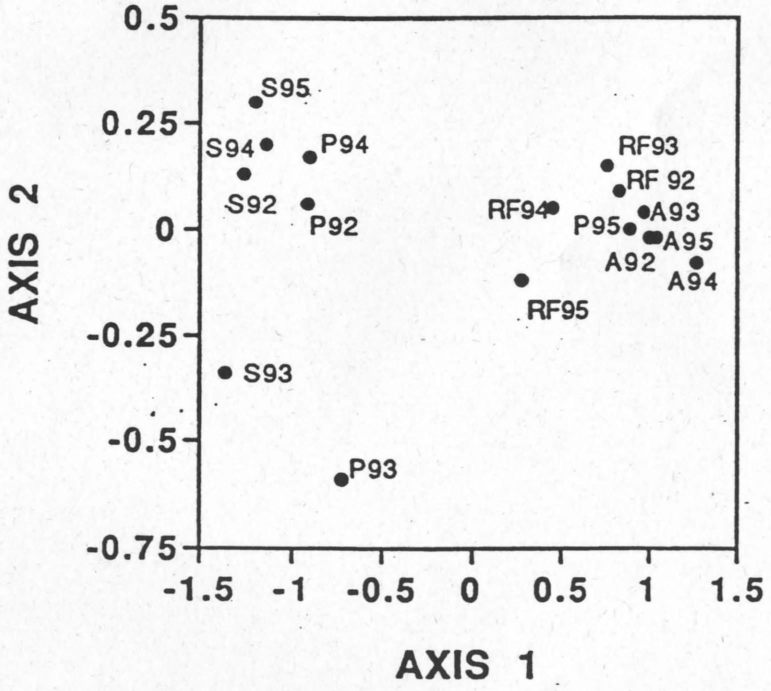


Figure 14. (Top) Multiple dimensional scaling (MDS) of limnological variables presented in Table 1. (Bottom) MDS of selected group of limnological variables (Secchi transparency, epilimnetic chlorophyll, crustacean zooplankton biomass, and daphnid biomass). Means of each parameter estimated from May - October data from 1992 to 1995 for Redfish (R), Alturas (A), Pettit (P), and Stanley (S) Lakes.

Bonneville Power Administration

PO Box 3621 Portland, Oregon 97208-3621

DOE/BP-22548-4 SEPTEMBER 1996 750