SNAKE RIVER SOCKEYE SALMON HABITAT AND LIMNOLOGICAL RESEARCH

ANNUAL REPORT 1994

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SNAKE RIVER SOCKEYE SALMON HABITAT AND LIMNOLOGICAL RESEARCH

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Abstract -Snake River sockeye salmon were listed as endangered in 199 1. Since then, the Shoshone-Bannock Tribes (SBT) have been involved in a multi-agency recovery effort. The purpose of this document is to report activities completed in the rearing environments of the Sawtooth Valley Lakes, central Idaho. SBT objectives for 1995 included continuing population monitoring and spawning habitat surveys, estimate smolt carrying capacity of the lakes, and supervise liiology and barrier modification studies.

Hydroacoustic estimates of 0. *nerka* densities in the Sawtooth Valley Lakes ranged from 32 to 339 fish/ha. Densities were greatest in Stanley followed by Redfish (217 fish/ha), Pettit (95 fish/ha), and Alturas. Except for Alturas, population abundance estimates were similar to 1993 results. In Alturas Lake, 0. *nerka* abundance declined by approximately 90%.

In 1994, about 142,000 kokanee fry recruited to Redfish Lake from Fishhook Creek. 0. *nerka* fry recruitment to Stanley and Ahuras lakes was 19,000 and 2,000 fiy, respectively. Egg to fry survival was 1 1%, 13%, and 7% in Fishhook, Alturas and Stanley Lake Creeks.

Kokanee spawning in Fishhook Creek was slightly lower than 1993 estimates but similar to the mean escapement since 1991. About 9,200 kokanee entered the creek in 1994 compared to 10,800 in 1993. Escapement for Stanley Lake Creek was only 200, a 68% reduction from 1993. Conversely, 0. *nerka* spawning densities increased to 3,200 in Alturas Lake Creek, up from 200 the previous year.

Approximately 92% of the 0. *nerka* production potential of Redfish Lake is being used by Fishhook Creek kokanee. Available space for sockeye production is 20,000 10-gram fish. Expected release totals for 1995 are 100,000 juvenile sockeye, well above safe levels of supplementation In the short term, lake fertilization is needed to meet production needs and prevent a forage crash during sockeye supplementation. For long term recovery, however, a trade-off between kokanee and sockeye production is required. We suggest limiting Fishhook Creek fry recruitment to 50,000 per year.

Spawning habitat is not limiting in Redfish Lake. The shoal spawning area on Sockeye Beach and in the South end of Redfish Lake should support up to 16,000 sockeye spawners. Percent fines in those areas were low (less than 35 % of the substrate was smaller than 6.3 mm), and should yield moderately high egg-to-fry survival.

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. Bjomn (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 abiity of these systems to produce sockeye smolts. Specific research objectives for 1994 were to (1) continue monitoring resident 0. *nerka* population dynamics (i.e., estimate in lake abundance, complete spawning surveys, and estimate tributary fiy recruitment), (2) describe fish foraging habits, (3) es&ate the current carrying capacity of Redfish Lake and compare it to production needs for 1995, and (4) complete spawning habitat surveys. Additionally, a feasibility study for barrier modifications (Montgomery Watson; Liming et al. '1994) and limnology investigations (Utah State University) were completed. Limnology results are included in this report.

This research should expedite sockeye recovery by helping to determine which of the Sawtooth Valley Lakes will provide the best spawning conditions, the safest rearing environment, the best growth rates, and ultimately the greatest numbers of adult returns.

STUDY AREA

Five lakes in the Sawtooth Valley are currently the focus of our habitat and limnology studies. Historically, the lakes and their inlet streams were used as spawning and rearing environments for endangered Snake River sockeye salmon (Bjomn et al. 1968; Evermann 1896). The lakes were glacially formed, range in elevation from 1985 to 2157 m, and are located in central Idaho (Figure 1). Specific qualities of the Sawtooth Valley 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. Secchi disk transparencies range from 8. - 16 m.

Native fish species found in the nursery lake system include sockeye/kokanee salmon 0. *nerka*, rainbow trout 0. *mykiss*, chinook salmon 0. *tshawytscha*, cutthroat trout 0. *clarki*, bull trout *Salvelinus confluentus*, sucker *Catastomus sp.*, redside shiner *Richardsonius balteatus*, dace *Rhinichthys*

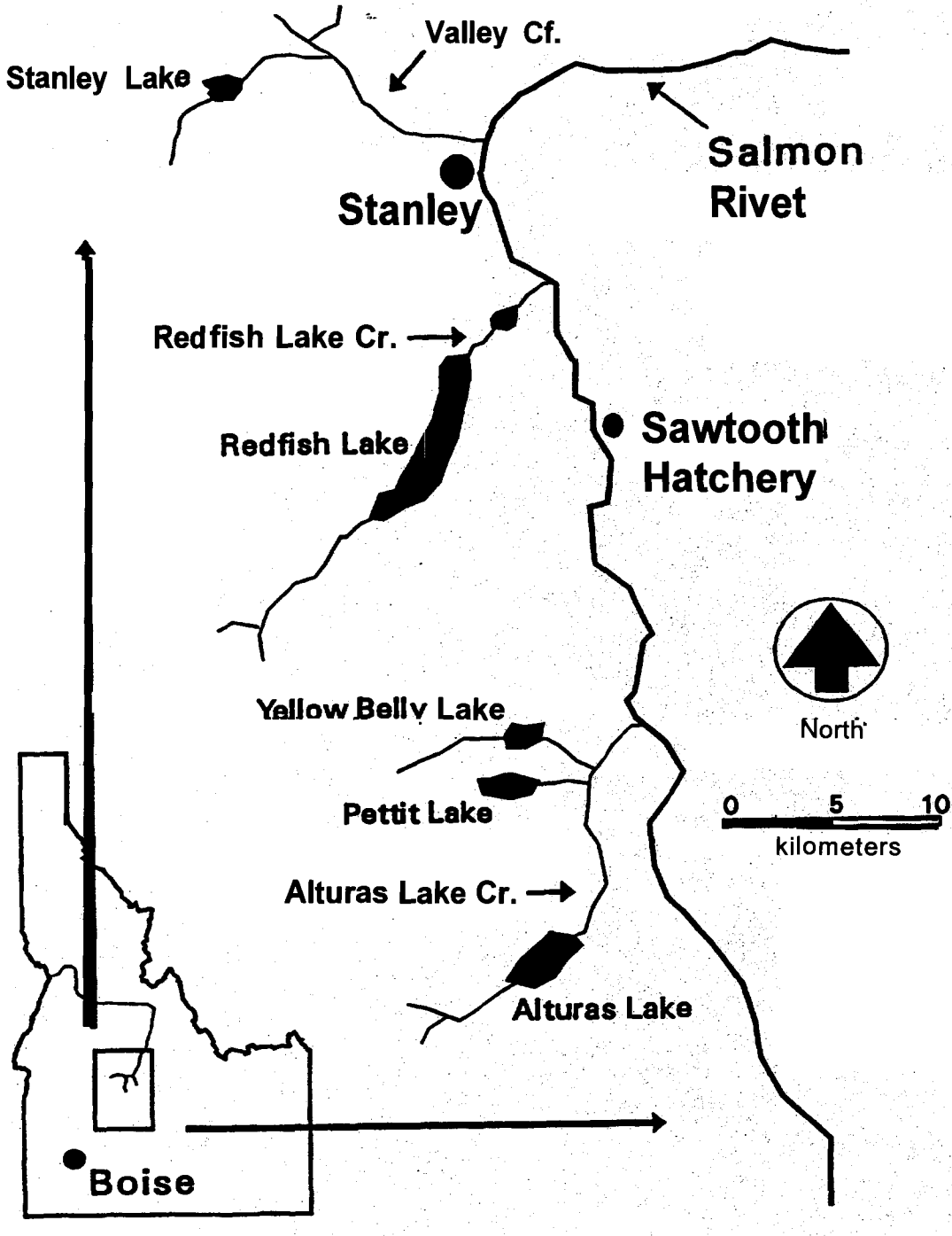

Figure 1. Stanley Basin study area map.

sp., northern squawfish ***Ptychocheilus oregonensis,*** and sculpin ***Cottus*** sp.. Non-native species include brook trout ***S. fontinalis,*** and lake trout S. ***namaycush.***

Table 1. Physical and Morphological features of the Sawtooth Valley Lakes.

Lake	Area (km^2)	Volume ($m^3 x 10^6$)	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

CHAPTER 1

POPULATION STUDIES

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, Pettit, and Redfish lakes, and five pings per second for Stanley Lake. A minimum of three (Stanley Lake) to four (ah other lakes) pings/target was necessary to qualii as a tracked target. Data were recorded on a Panasonic SV-3700 digital audio tape recorder.

Before sampling we set waypoints at each lake edge with a global positioning system (GPS). Waypoints were set to allow for sampling transects to run zigzag across all lakes except Pettit Lake, where we used parallel transects (Figure 2; Appendix 1). We sampled twelve, six, fourteen, and eight transects at Alturas, Pettit, Redfish, and Stanley lakes, respectively.

Sampling was conducted on two nights during the new moon phase in September, and 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 0. ***nerka***. Therefore, we employed vertical gillnets for this session to determine if other fish species were found in the pelagic areas during sampling.

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 = 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

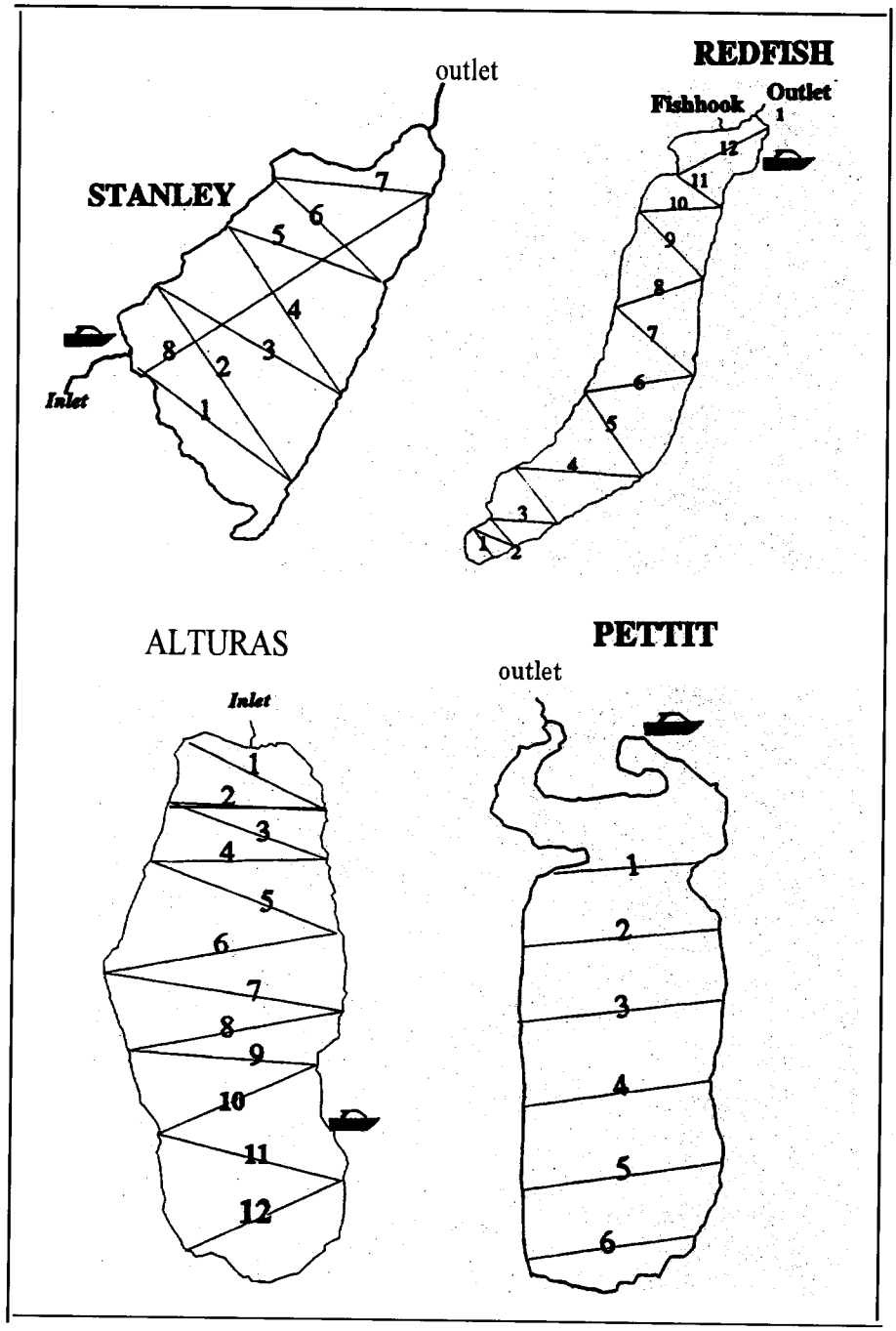

Figure 2. Transects used for hydroacoustic surveys.

length frequency distributions and age analyses. In Alturas Lake we were unable to capture enough fish in the vertical gill nets to partition fish targets by species. Instead, only fish within **100-200mm** were used to estimate 0. ***nerka*** densities. That range encompasses the variance observed in trawl catches. Five **different** size classes were used for Pettit, **Redfish,** and Stanley lakes. We also estimated vertical distributions for each lake.

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

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

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

$$\bar{D}_i = \frac{\sum_{j=1}^{4} L_j \bar{D}_{ij}}{\sum_{j=1}^{4} L_j} \qquad (3)$$

and variance was estimated by

$$var\bar{D}_i = \frac{T_i}{T_i - 1} \sum_{j=1}^{n} L_j^2 (\bar{D}_{ij} - \bar{D}_i)^2 / \left(\sum_{j=1}^{n} L_i\right) \qquad (4)$$

where D_i = mean density (number/stratum) in stratum I, D_{ij} = mean density for **the** j**th** 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

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

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

We estimated **fry** recruitment **from** Alturas, Stanley, and Fishhook Creeks. Fry were captured in small **drift** nets (30cm wide and 60cm deep) anchored to the substrate with rebar. In Fishhook Creek, three **fry** nets were located under the bridge 50 m above the confluence with the lake. Stanley and Alturas Lake Creeks were sampled with two **fry** traps. Each trap was located about 200' m above each lake. Nets were generally fished between 1800 and 0800 hrs. Fry captured in the traps were counted, a subsample measured for length and weight, and then released. Fry traps were operated daily **from** 4 April to 10 July 1994.

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

We attempted to monitor smolt migration from Alturas and Stanley Lakes. Alturas Lake Creek was sampled with a rotary screw trap. 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, The trap was deployed 18 May. We set the trap each night between 1800 and 2000 hrs. The trap was checked at 0000 hrs and pulled by 0800 hrs. To protect juvenile salmonids from being consumed by squawfish or large salmonids, the live box contained sage brush bundles for cover.

In Stanley Lake Creek, a fyke net (1.5 m wide and 1.5 m deep) was located on the outlet stream about 4.8 km below the lake. The net was deployed on 4 April and ran continuously through 14 June. No 0. *nerka* were collected in the Alturas or Stanley Lake Creek traps. Non-target species were identified and released.

O. nerka Spawning Surveys

Stream Spawning -Stream surveys were conducted *on* all Stanley Basin lake tributaries that contain 0. *nerka* populations. Surveys began in July and continued until spawning ceased. 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).

Stream life was estimated by marking kokanee with florescent yarn. To capture kokanee the day they entered the stream, either a fyke nets or wire weir was deployed at the creek mouth. Fish were anesthetized in MS222 and then tagged with florescent yarn just below the dorsal fin. A total of 67, 40, and 36 fish were tagged in Fishhook, Alturas, and Stanley Lake Creeks, respectively.

Beach Spawning -Since 1992, we have been monitoring beach spawning populations on Sockeye Beach, Redfish Lake. With exception to genetic research, information describing the beach spawning population (residual sockeye) is limited. In 1994, we estimated spawning densities *on* sockeye beach, the South end of Redfish Lake, and Point Campground. Historical spawning observations and or redd counts have been made in these areas (Spaulding 1993).

Snorkel surveys were used to estimate spawning densities. Each survey consisted of at least three observers swimming one or

two 20 min transects. Each transect covered approximately 300 m of shoal. Transects were completed parallel to shore at depths ranging from .5 m to 5 m. For Sockeye Beach, we estimated 0. *nerka* densities within the boundaries of Sockeye Beach as delineated by U.S.D.A. Forest Service signs. At the South end of the lake, surveys were conducted near the south-west inlet stream. Surveys were completed on Sockeye Beach every one to two weeks. Point Campground and the South end of the lake were surveyed at less frequent intervals.

Age Analysis

Fish samples were collected with vertical gillnets, fyke nets, and the IDFG's trawl boat. Prior to otolith removal, length and weight for each fish was recorded. Because IDFG and SBT shared samples, some of the age results overlap with those reported by Kline and Younk (1995).

In the lab, age was determined following the procedures outlined by Brothers (1987). At least two independent estimates of age were made for each otolith. If the observers did not agree, the sample was examined a second time. Additional verification of aging was determined during an exercise where IDFG and SBT biologists independently aged individuals from each size class of fish. Each otolith was analyzed until a super-majority (4 of 5 observers) agreed on its age. All samples used in the age analysis were collected in the fall.

0. *nerka* Diet Analysis

Stanley, Pettit and Alturas Lakes were sampled in June and September 1994. Single-paneled vertical gillnets of square mesh sizes 1.9, 2.5, 3.8, 5.08, and 6.35 centimeters were used. Each net was 3 m wide and 30 m deep. To sample the entire water column, a net location of about 30 m deep was selected. Nets were set between 1800 and 2000 hours and pulled by 0800 hours the next day. Depending on catch rates, lakes were sampled either one or two nights.

For diet analysis, fish stomachs were removed and placed in 70% ethanol. In the lab, prey items were sorted by order, blotted dry and weighed to the nearest 0.01 g. Zooplankton prey were enumerated and measured for length. Zooplankton lengths were converted to weight using the length weight regression equations reported in McCauley (1984). We calculated aggregate percent by weight (Swanson et al. 1974) for all species of fish sampled.

Fish collected in trawl surveys were also sampled for stomach contents. Trawling was completed concurrently with vertical gillnet and hydroacoustic surveys. Stomach contents were handled as described above.

Results

Hydroacoustics

The variance of *O. nerka* densities during 1994 between lakes was high, ranging from 32 to 339 0. ***nerka*/hectare** for Alturas and Stanley lakes, respectively (Figure 3). Volumetric densities were . 10, .32, .52, and *2.64 0. nerka*/1000 m^3 for **Alturas,** Pettit, **Redfish,** and Stanley lakes, respectively. Similar variances have been reported in the past (Teuscher et al. 1994).

Population estimates for 0. ***nerka*** in all four lakes showed a decrease in abundance in 1994 compared with previous years **(Figure** 4).

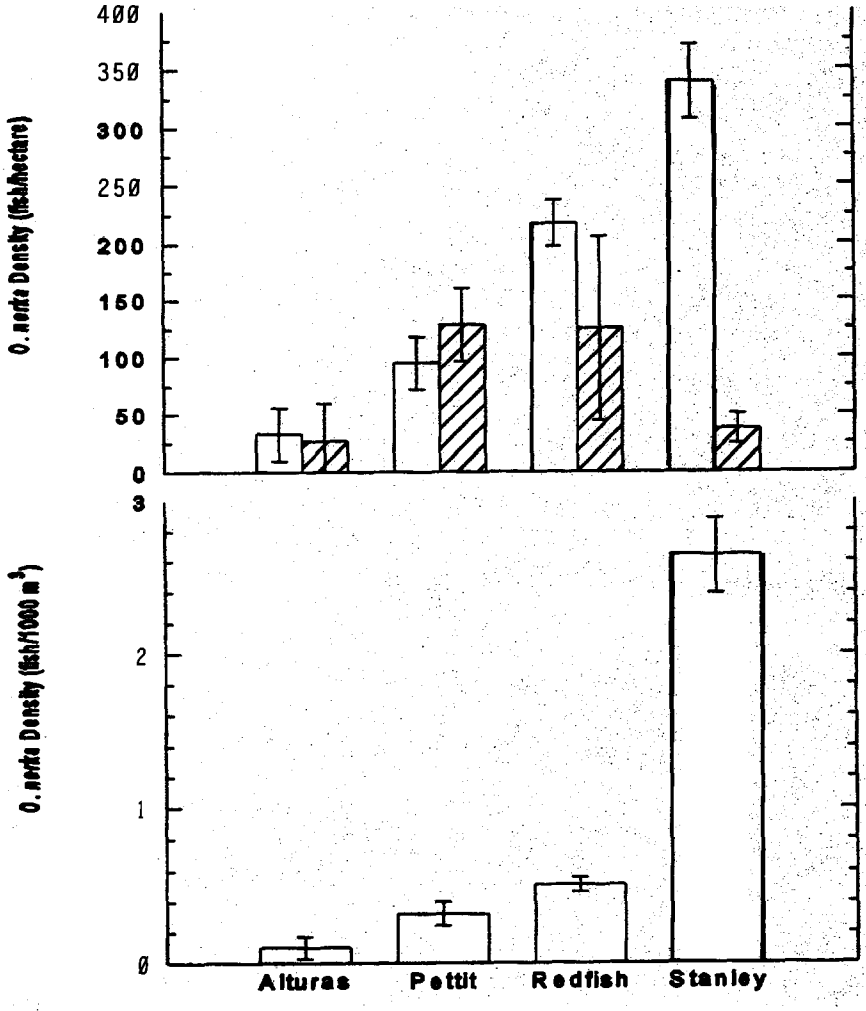

Figure 3. **Hydroacoustic** (open bars) and trawl (diagonal bars) estimates of *O. nerka* densities in four **Sawtooth** Valley Lakes during September 1994.

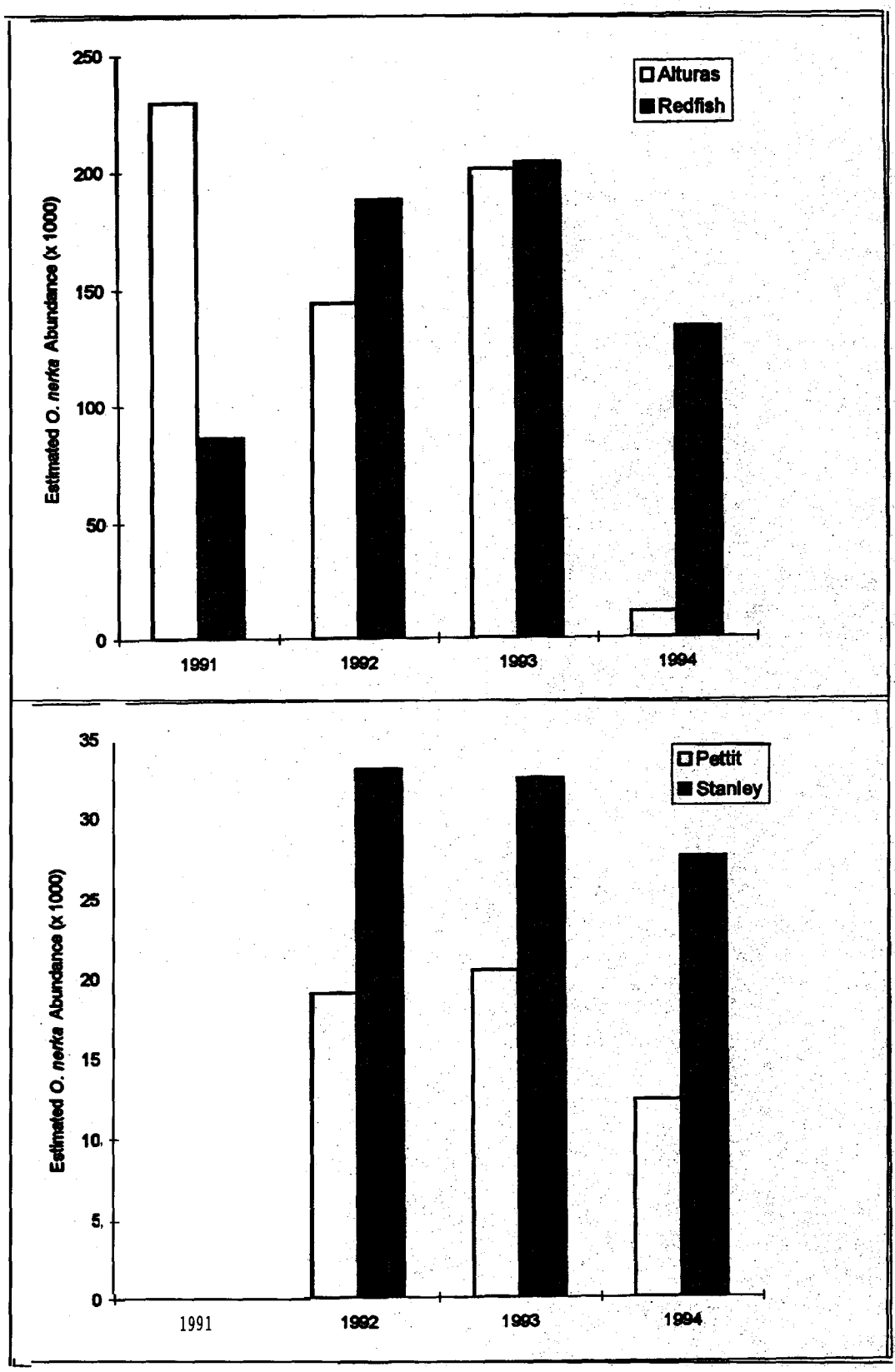

Figure 4. Annual 0. ***nerka*** abundance estimates for four Sawtooth Valley lakes. Error estimates were not available for all data, so were excluded from this figure

Declines were slight in all lakes except Alturas Lake which experienced a decline of approximately 90% of the 1993 estimate. Trawling efforts in Alturas showed a siilar decline starting in 1992 with another large decline occurring between 1993 and 1994 (Kline and Younk 1995). The decrease in trawling estimates, from 49,000 in 1993 to 5,700 in 1994, was similar in proportion to hydroacoustics. The hydroacoustic estimate in Alturas Lake including all targets was only ≈25% of 1993's estimate.

Alturas Lake - We estimated total fish abundance in Alturas Lake at 52,300 ± 8,000 during September of 1994 (Table 2). By partitioning targets into a conservative size range of 100 - 200 mm we estimated an in lake 0. *nerku* population of 11,000 ± 1,100. An estimated 2,000 *O. nerka fry* entered the lake in the spring of 1994 that would contribute to the *O. nerka* estimate. We also captured bull char and squawfish in our vertical gill nets with fork lengths >200 mm.

In Alturas Lake, the greatest proportion of targets were found between fifteen and twenty-five meters (Figure 5). Vertical distributions of *O. nerka* caught in the vertical gill nets were similar to hydroacoustic targets between fifteen and twenty meters, with a discrepancy in the ten to fifteen meter strata.

Pettit Lake - We estimated fish abundance in Pettit Lake during September at 17,200 ± 7,700 (Table 2) of which 12,400 were partitioned as *O. nerku (25 - 300 mm).* We captured redside shiners in the vertical gillnets that fell within that range, but they were all caught less than five meters deep and were not associated with other tracked targets. Pettit Lake supported a large population of three year old fish, estimated at 2,600 fish with a biomass greater than 546 kilograms.

Vertical distribution of 0. *nerka* in Pettit Lake was concentrated in the ten to twenty meter strata, with small numbers found in adjacent depths (Figure 6). Hatchery rainbow trout between 253 and 287 mm were caught in vertical gillnets, yet were found less than ten meters deep. Targets that size in that depth were excluded for population estimates for 0. *nerka.* Vertical gill net sampling did not match hydroacoustic estimates well for fish between *25* and 120 mm. Vertical gill net captures were predominant in the fifteen to twenty meter stratum with fifty percent of total 0. *nerku* captured compared with only thirteen percent for hydroacoustics. Sii-three percent of total *O. nerku* from hydroacoustic sampling was found in the ten to fifteen meter stratum, and only thirty-three percent of the vertical gill net sample was captured within that stratum. Vertical distribution for the larger size range matched well for the two methods (Figure 6), with greater than 60% of total 0. *nerka* for both found between ten and fifteen meters.

Redfish Lake - We estimated total lake abundance at 178,800 ± 33,000 fish during September of 1994 (Table 2). 0. *nerku (30 -* 200 mm) abundance was estimated at 133,500 ± 12,300. That figure did not include approximately 9,000 kokanee spawning in Fishhook Creek. The estimate of 76,600 ± 19,600 YOY fish compares with our emergent fry estimate of the same cohort of ≈ 142,000.

Table 2. Population estimates for four **Sawtooth** Valley Lakes **during** September of 1994. Length and weight values are represented in **millimeters** and grams. Underlined values were used in the 0. ***nerka*** population and biomass estimates.

SIZE CLASS	length	weight	Population Estimate	Biomass Estimate
		Redfish Lake		
<30	23	**.13**	27,500 ± 5,480	3,570 ± 710
30-70	47	1.2	76.600 ± 19.560	**95.000 ± 24.250**
71 - 130	97	10.3	36.000 ± 8.240	371.880 ± 85.130
131- 200	165	48.3	20.760 ± 7.470	1.003640 ± 361.340
>200	243	158.7	17,950 ± 7,030	**2,848,130 ± 1,116,080**
		Total Lake Estimate	178,830 ± 3 1,980	
		Pettit Lake		
25-60	43	**.90**	4.580 ± 2.260	4.070 ± 2.010
61 - 120	87	8.4	**1,980 ± 1.310**	16.580 ± 10.970
121 - 200	161	59.1	3.210 ± 1.530	189.700 ± 90.160
201 - 300	241	210.5	**2.600 ± 3.120**	546.170 ± 657.510
>300	374	**927**	4,870 ± 2,720	**4,517,270 ± 2,516,800**
		Total **Lake** Estimate	17,240 ± 7,650	
		Alturas Lake		
100-200	142	32.9	10.980 ± 1.090	361.180 ± 35.890
		Total Lake Estimate	**52,290** ± 7,950	
		Stanley Lake		
50 - 100	71	4.0	12.720 ± 1.540	50.880 ± 6.160
101 - 170	134	25.6	9.640 ± 580	246.780 ± 14.800
171 - 240	204	84.4	5.160 ± 810	435.500 ± 68.300
260 - 340			2,520 ± 330	
>360			4,190 ± 850	
		Total Lake Estimate	35,440 ± 3,440	

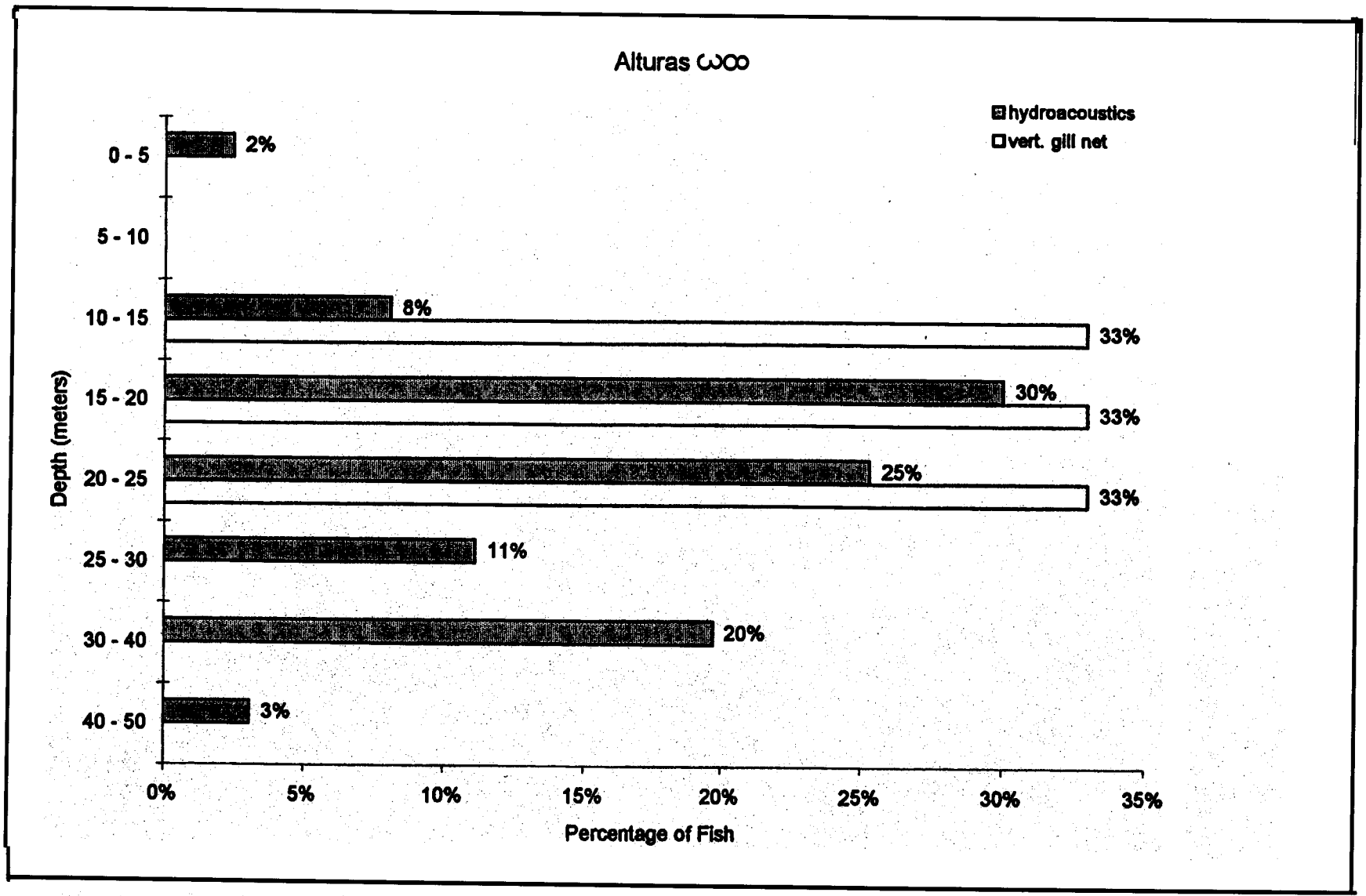

Figure 5. Vertical distribution of *O. nerka* in Alturas Lake during September of 1994.

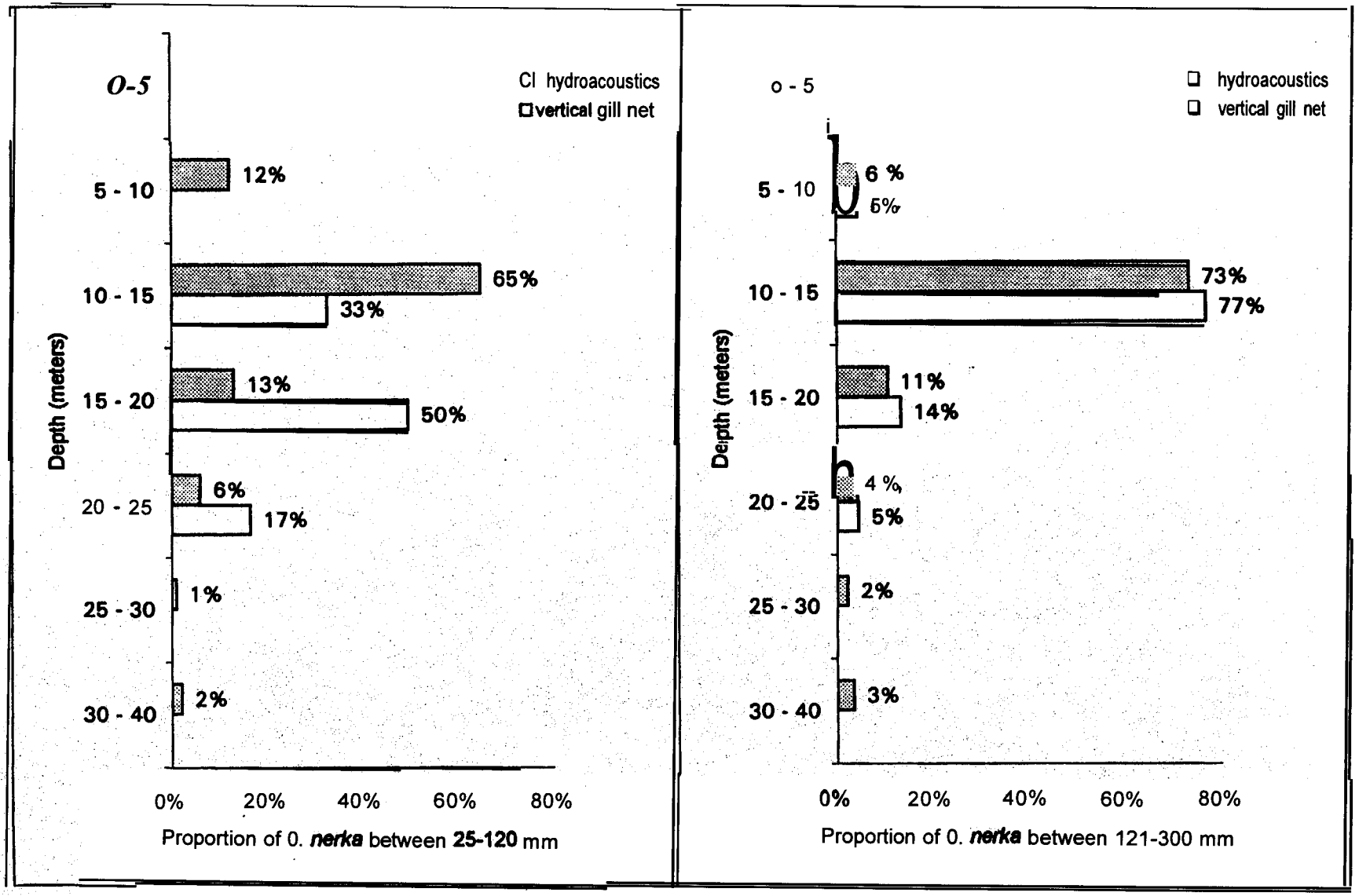

Figure 6. Vertical distribution of O. *nerka* during September 1994 in Pettit Lake.

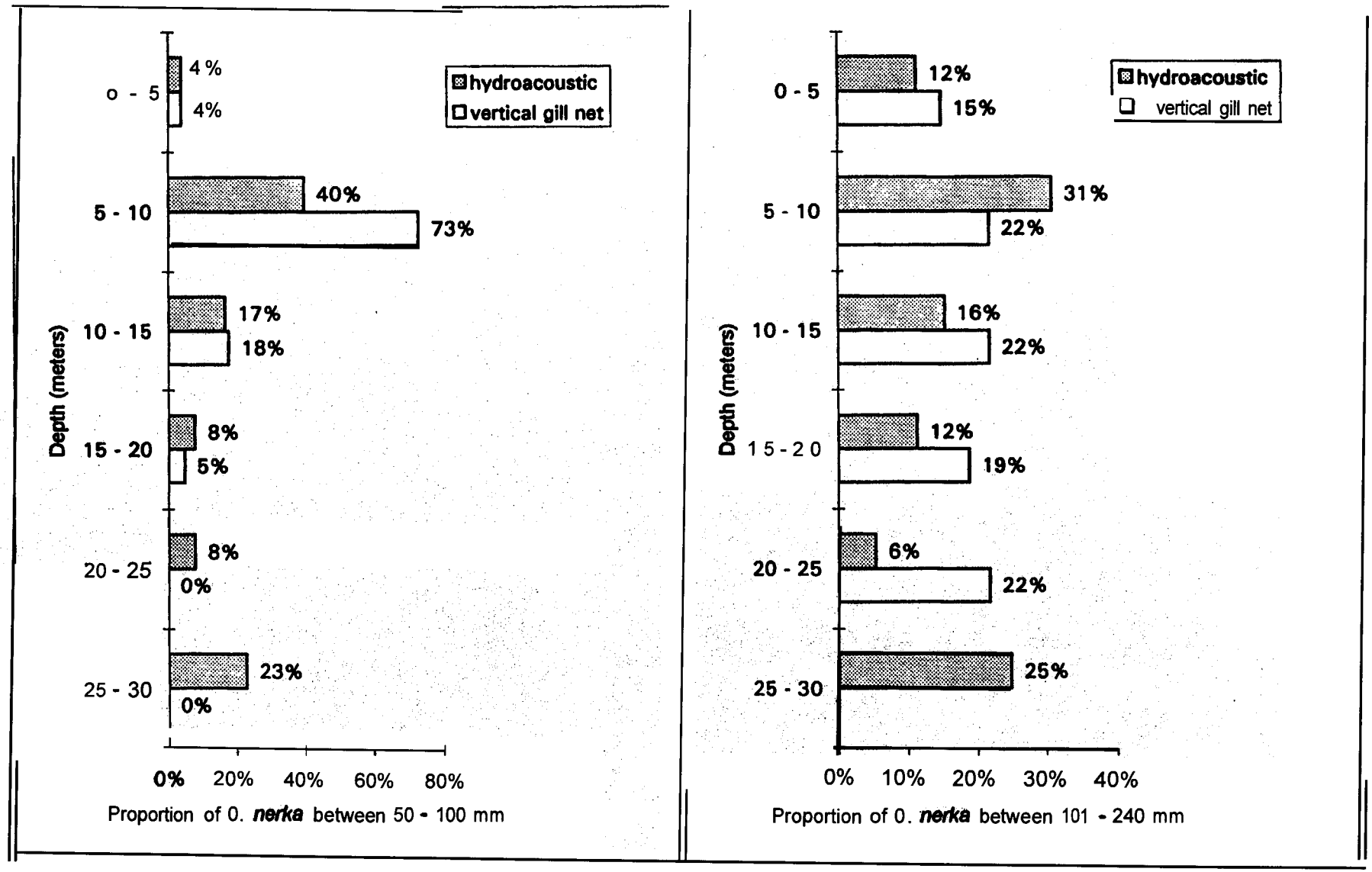

Figure 7. Vertical distribution of 0. ***nerka*** during September 1994 in Stanley Lake.

Most of the fish in **Redfish** Lake were found between ten and twenty meters (Appendix 1). The fifteen to twenty meter stratum contained the greatest proportion of all three 0. ***nerka*** **size** classes. Only small (30 - 70 mm) fish were found above five meters, and very few fish were targeted below twenty-five meters.

Stanley Lake - **In** lake 0. ***nerka*** abundance was 27,500 ± 2,560 during September 1994. Although this is a slight decrease from previous years, Stanley Lake had the highest density of fish out of the four lakes surveyed (Figure 3).

Fish were vertically distributed more evenly from bottom to top in Stanley Lake (Figure **7),** a shallower lake, than the other lakes surveyed. Small fish (50 - 100 mm) were more frequent between five and ten meters than at other depth strata Large fish (10 1 - 240 mm) were also more **frequent** at that depth range, but the proportion was much smaller.

Fry Recruitment

Approximately 142,000 **fry** entered **Redfish** Lake from Fishhook Creek. Emergence began on 14 April and continued through 11 July. Peak emergence occurred on 6 June, which was similar to the 1993 peak (Figure 8).

Assuming equal sex ratios and 230 eggs per female, egg to fry survival was 11%. **Egg-**to-fry **survival** and recruitment trends for Fishhook Creek are show in Table 3.

Stream recruitment of 0. ***nerka*** to Alturas and Stanley Lakes began earlier and was a small g-action of **Redfish** Lakes recruitment (Figure 9; Table 3).

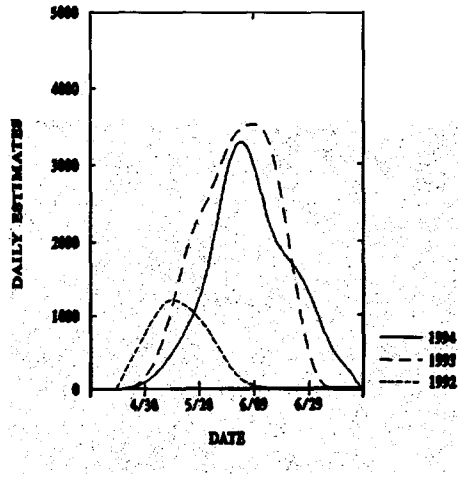

Figure 8. Daily **fry** emergence estimates in Fishhook Creek for 1992, 1993, and 1994. Lines are distance **weighted** least squares.

Table 3. Fry recruitment and egg to fiy survival in **Fishhook,** Alturas, and Stanley Lake Creek.

creek	year	recruits	egg/fry survival
Fishhook	1994	142,000	11%
Fishhook	1993	166,000	12%
Fishhook	1992	56,000	3%
Alturas	1994	2,000	13%
Alturas	1992	7,000	
Stanley	1994	19,000	7%

An estimated 2,000 **fry** recruited to Alturas Lake, compared to 19,000 for Stanley. Egg to fry survival was highest in Alturas Lake Creek (13%). Egg-to-fry survival in Stanley was 7%.

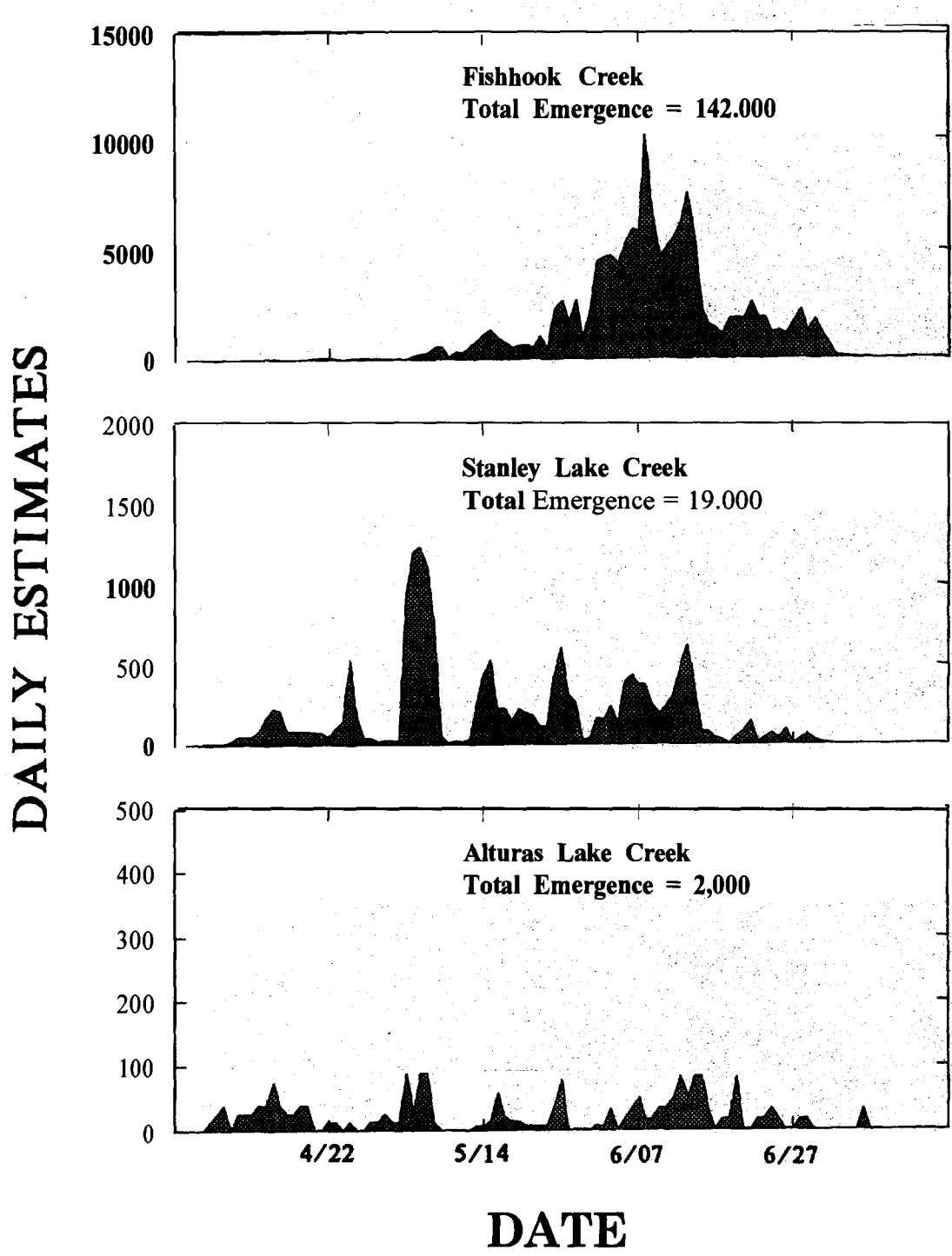

Figure 9. Daily fry emergence estimates in Fishhook, Stanley, and Alturas lake Creeks. Notice different y-axis.

Smolt Monitoring

No 0. ***nerka*** were collected during three nights of sampling Alturas Lake Creek (1 **8-** 20 May, 1994). On the third night of sampling we exceeded our incidental take for chinook salmon (23 fish). We initiated consultation with NMFS to increase the incidental take, but were not granted a permit modification before outmigration was over.

0. ***nerka*** smolts, however, did outmigrate from Alturas Lake. Several days after pulling our screw trap, 0. ***nerka*** smolts were collected at the Sawtooth Fish Hatchery. IDFG estimated that smolt production from Alturas Lake Creek was about 900 (Kline and Younk 1995)

In Stanley Lake Creek smolt monitoring began on 4 April and continued through 14 June, 1994. No 0. ***Nerka*** or chinook were captured in the Stanley Lake Creek trap. Speckled **dace** dominated the catch, followed by suckers, sculpin, and brook trout (total catch only 16 fish).

Suawninn Surveys

Stream Spawners -Based on an estimated stream life of 6 days, total escapement of kokanee spawners to Fishhook Creek was 9,200. Spawning densities were down from the 1993 estimate of 10,800, but similar to escapement from earlier years (Table 4).

Based on an estimated stream life of 11 days, total escapement to Alturas Lake Creek was 3,200. Alturas escapement increased by a factor of 16 over the 1993 estimates (Table 4). Conversely, Stanley Lake Creek escapement was sharply reduced. Approximately 600 kokanee spawned in Stanley Lake Creek. Stream life estimate for Stanley Creek was 7 days.

Spawn timing has been very consistent in Fishhook Creek. Fish entered the stream the first week of August and dropped out by the end of September (Figure 10). Spawning in Stanley Lake Creek begins in mid-July, peaks several weeks later, and a small group of fish enter the creek in October. Spawning in Alturas Lake Creek mirrors temporal trends of Fishhook Creek (Figure 10).

Table 4. Escapement estimates for Fishhook, Alturas, and Stanley Creeks.

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

Spawning size varied among the lakes. Stanley Lake Creek kokanee were the largest stream spawning fish (mean fork length = 237mm). Fishhook fish were slightly smaller at 221 mm, followed by Alturas at 179 mm (Figure 10). Spawning weights and ranges are reported in Table 5.

Variation ***in size was*** greatest in Alturas, ranging from **130** to 290 mm fork length. The siie **frequency** distribution (Figure 10) shows a dominant cohort in the 170 mm range, and a few much larger fish around 280 mm. Age is similar between the two groups indicating that the larger fish are

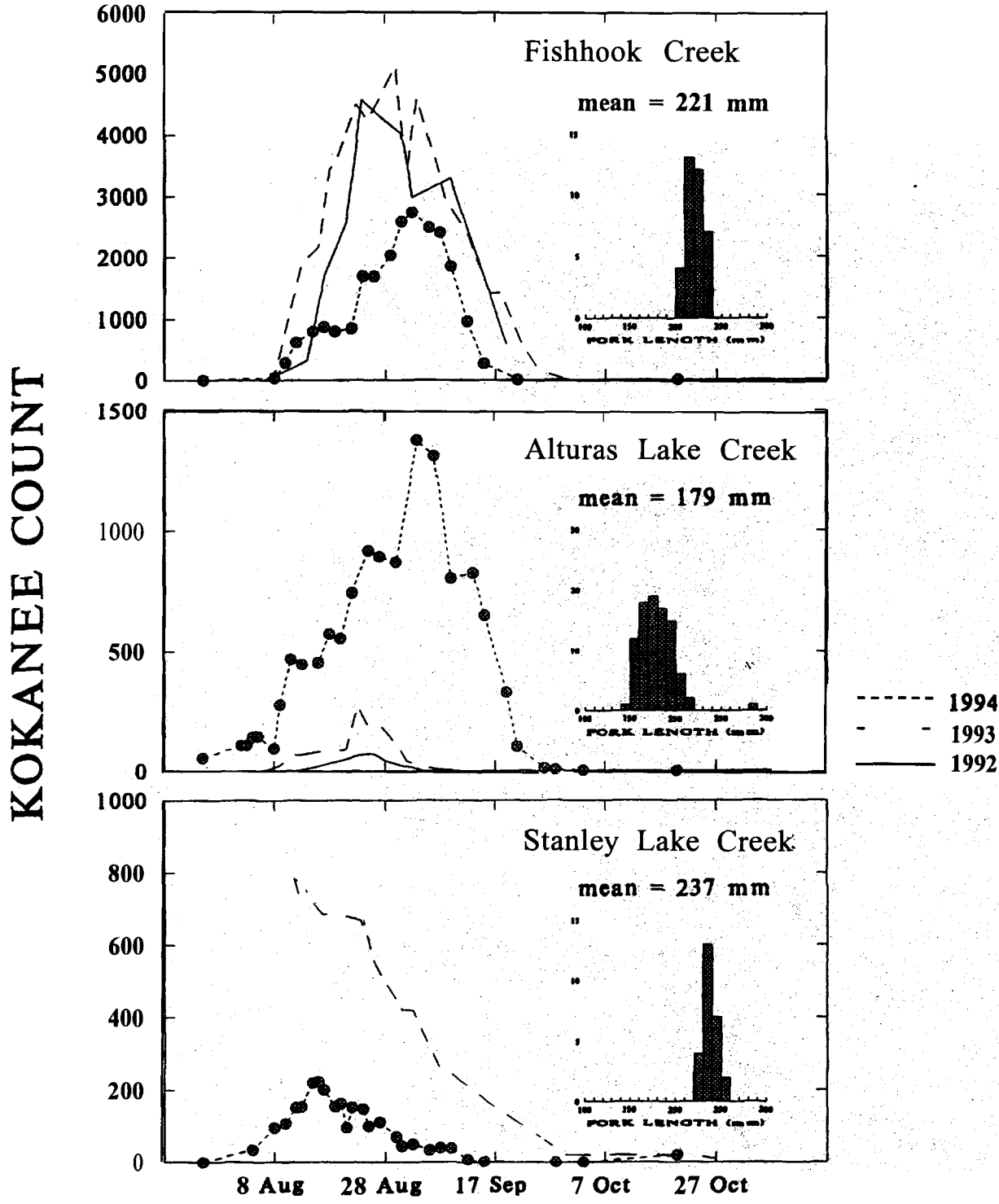

Figure 10. Kokanee escapement in Fishhook, Stanley, and Alturas Lake Creeks.

utilizing a more profitable food resource or foraging environment.

For stream spawning populations, fecundity was highest in Fishhook Creek (330 eggs per female), followed by Stanley at 270, and was lowest in Alturas Lake Creek (150). Fecundity of the shore spawning kokanee in Pettit Lake is much higher at 670 . The higher fecundity from Pettit may be due to their larger siie (Table 5)

Table 5. Spawning characteristics of 0. *nerka in the* Sawtooth Valley Lakes. Mean (range) values are reported. Redfish, Stanley and Alturas fish spawn in tributary streams. Pettit kokanee spawn on shoal habitat.

lake	length mm	weight g	fecundity
Redfish	221 (204-235)	118 (90-151)	230 (202-283)
Stanley	238 (226-255)	131 (105-169)	270 (186-361)
Alturas	179 (149-280)	55 (29-204)	150 (92-186)
Pettit*	257 (240-275)	245 (197-306)	670 (559-1056)

*1993 data

Beach Spawners -Snorkel surveys were completed in three areas of Redfish Lake (Sockeye Beach, Point Campground, and the South end of the lake). Surveys began on 3 October and continued through 9 November, 1994. Results from the surveys are shown in Table 6. Unfortunately, the point estimates preclude a determination of total residence time and make it diicult to predict total escapement of the shore-line populations in Redfish lake. However, some qualitative estimates of population size can be made. For the stream populations, escapement is roughly equal to 2 times the peak count. Applying that assumption yields an escapement estimate for Sockeye Beach of about 50.

Table 6. Results from beach spawning surveys in Redfish lake.

location	method	date	count
SB	Snorkel	3oct	8
SB	Snorkel	11 Oct	22
SB	Snorkel	19 Oct	3
SB	Snorkel	2 Nov	19
SB	Boat	16 Nov	0
PCG	Snorkel	2 Nov	0
PCG	Snorkel	8 Nov	0
SE	Snorkel	19 Oct	40
S E	Boat	9 Nov	0

SB Sockeye Beach
PCG Point Campground
SE the south end of Redfish Lake

In the south end of the lake, spawning redds were visible and provided a rough estimate of spawning densities. On 9 November, we counted about 15 redds. Based on redd counts, at least 30 residuals spawned there. However, on 19 October we observed 40 fish, indicating that more than two fish share a redd. Again, we can not precisely estimate the population size in the south end, but the limited data we have indicates a modest escapement. No fish were observed spawning in the Point Campground area.

Age Analysis

September length and weight at age varied greatly among the four lakes. Age 0 0. *nerka* in Pettit Lake were small (0.7 g) compared to similar aged fish from the

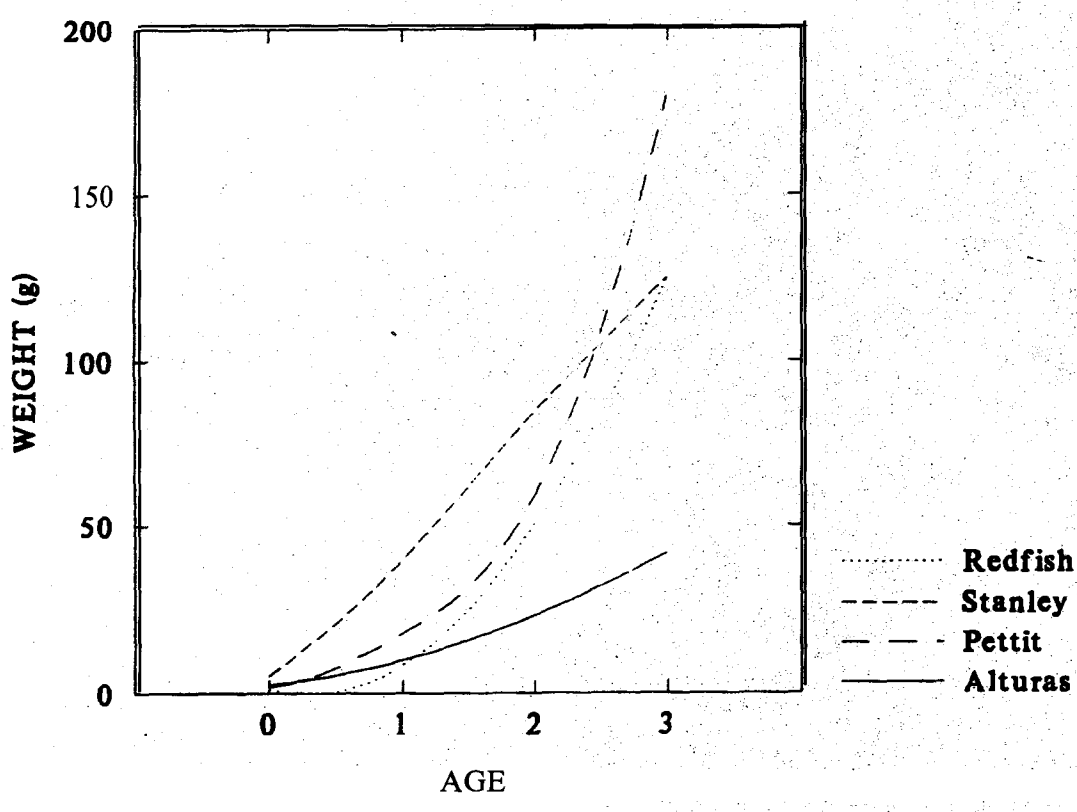

Figure 11. Growth patterns for 0. *nerka* populations in the Sawtooth Valley Lakes. Lines represent distance weighted least squares.

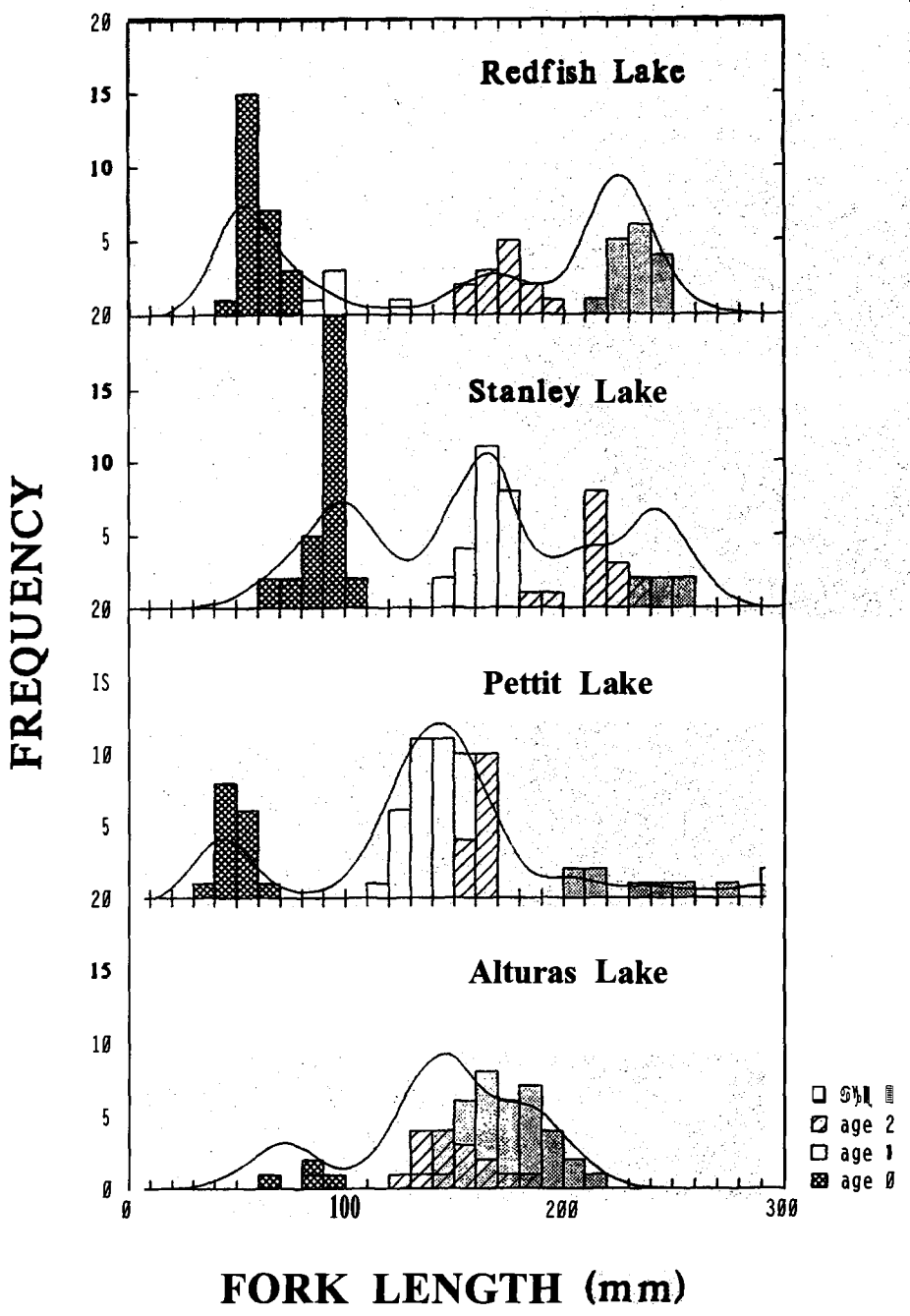

Figure 12. Length frequency distributions of 0. *nerka* collected in September 1994 in four Sawtooth Valley Lakes. The smoothed lines that overlay the bar plots are pooled length data from trawl, vertical gillnet, and a sample of lengths from ~ 30 spawning fish collected in the creeks. Therfore, the lines represent our best depiction of length frequencies for all cohorts. The lines are distance weighted least squares functions. Bars represent aged fish.

other lakes (Table 7). The early emergence of Stanley and Alturas *O.nerka* appeared to provide an early growth advantage (2.7 g and 5.6 g for age 0 fish from Alturas and Stanley Lakes).

Growth patterns for the Sawtooth Valley 0. *nerka* populations varied greatly. Growth in Stanley and Alturas appeared linear (Figure 11). More conventional patterns were observed for Redfish and Pettit Lake populations.

Length frequency distributions for 0. *nerka* age classes are shown in Figure 12. Overlap among age classes was limited in Redfish, Stanley and Pettit Lakes. Conversely, a high degree of overlap occurred between 2 and 3 year old fish in Alturas Lake (Figure 12).

O. nerka Diet Analysis

Zooplankton and chironomids are the dominant prey of 0. *nerka in* the Stanley Basin Lakes. In general, chironomids are the preferred prey item early in the year. Zooplankton are selected more frequently in September (Figure 13 and 14). However, the temporal shift was not as pronounced in Alturas Lake and was probable due to the extremely low zooplankton population (Figure 14).

To summarize, 0. *nerka* diet was siiar to that reported in 1993. No significant diet overlap occurred between rainbow trout and 0. *nerka*. In Stanley Lake, lake trout feed extensively of 0. *nerka*. Squawfish in Alturas lake consumed a variety of fish prey that included 0. *nerka*

A complete review of diet results can be found in Appendix 2. Rainbow and lake trout diets are also reported in Appendix 2.

Table 7. Mean weight and length in September of different age classes of 0. nerka from Redfish (R), Stanley (S), Pettit (P), and Alturas (A) Lakes.

		age			
Lake	N	0	1	2	3
			g		
R	65	1.3	8.3	51.0	126.0
S	75	5.6	37.7	86.9	125.3
P	81	0.7	28.4	49.2	183.2
A	65	2.7	9.8	23.8	42.1
			mm		
R	65	49	90	163	223
S	75	82	156	207	235
P	81	41	131	156	229
A	65	73	107	**141**	165

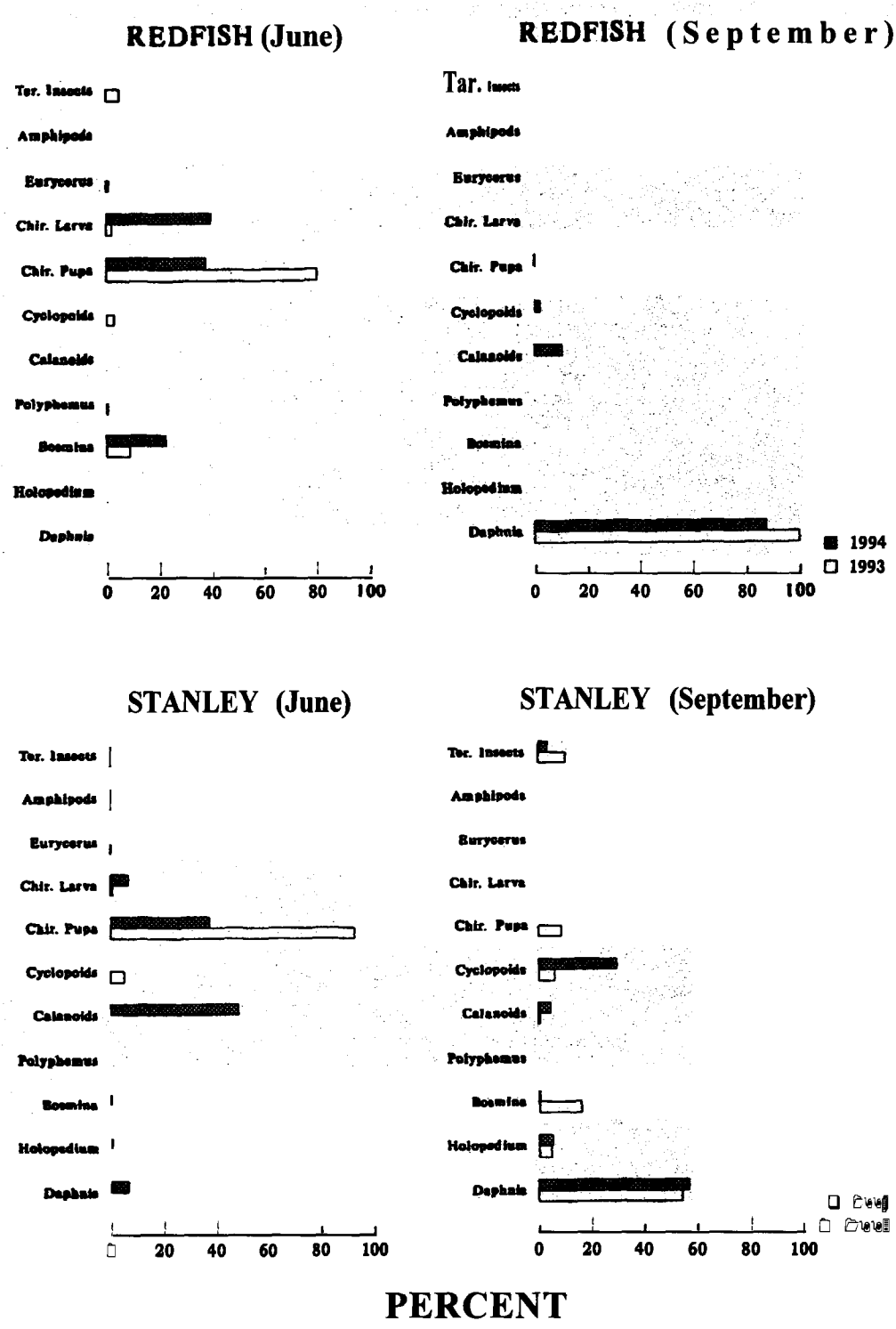

Figure 13. Diet analysis for 0. *nerka* in Redfish and Stanley Lakes. Data are percent by weight.

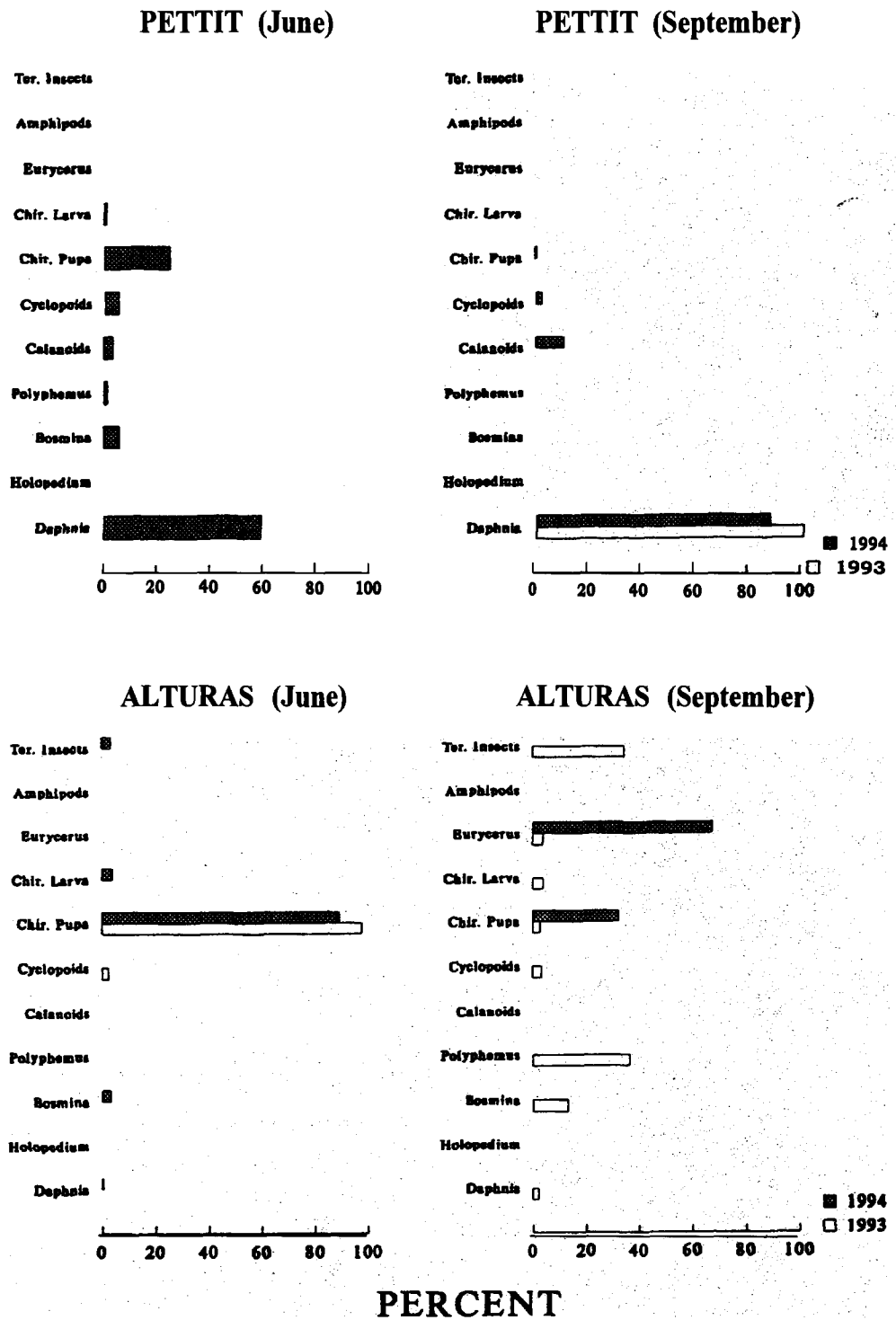

Figure 14. Diet analysis for 0. *nerka* in Pettit and Alturas Lakes. Data are percent by weight.

Redfish Lake -Redfish Lake supports the largest 0. ***nerku*** population in the Stanley Basin. Our 1994 pelagic estimate of 0. ***nerku*** abundance was 133,000. Since we began hydroacoustic population estimates in 199 1, no clear signs of increase or decrease have occurred (Figure 4). The range of estimates varied between 100,000 and 200,000 fish, but the variance on those estimates was high. Also, we used a different echosounder in 1994. The new split beam system should help reduce the variance around the population estimates and describe fish size more accurately, but it also complicates comparisons with earlier data.

Trawl estimates of 0. ***nerku*** abundance were lower than our acoustic results. Acoustic estimates for Redfish Lake in 1994, 1993, and 1992 were 2.6, 4.1, and 4.8 times greater than trawl estimates. Parkinson et al. (1994), reported a similar magnitude of differences between trawl and acoustic. In Stanley Lake, however, trawl and acoustic estimates are extremely diierent (Table 8). The 'cause of the difference may be due to Stanley Lake being too small to sample effectively with the trawl (Kline 1995, personal communication). Despite the Stanley Lake results, if we combine all years and lakes, the correlation coefficient for trawl and acoustic abundance estimates was significant (r-0.86, $p < 0.01$).

Since 1991, 36,800 kokanee have spawned in Fishhook Creek. The lowest escapement occurred in 1991 (7,200); the highest was in 1993 (10,800). Most of the spawners that enter the creek are age 3 and weigh about 120 g. Spawning begins in August and continues through September. Spawnersremain in the steam about 6 days, where each female deposits approximately 330 eggs.

Table 9. Comparison between trawl and acoustic estimates of 0. ***nerka*** abundance in the Sawtooth Valley Lakes. Trawl data is from Kline and Younk (1995).

Lake	Year	Acoustic	Trawl	Ratio A/T
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	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	1994	12,400	14,700	0.8
Pettit	1993	20,400	10,500	1.9
Pettit	1992	19,000	3,000	6.3
Stanley	1994	27,500	2,700	10.2
Stanley	1993	32,400	1,300	24.9
Stanley	1992	33,000	2,100	15.7

Mean egg to fry survival in Fishhook Creek is about 9%. Fry weigh 0.26 g on emergence and reach 1.3 g by September. Migration to the lake coincides with a spring chironomid hatch. Young of the year captured in June 1993 fed almost exclusively on them. By September, kokanee shift their diet to zooplankton.

Alturas Lake -Unlike **Redfish,** the 0. ***nerku*** population is crashing. Hydroacoustic estimates of abundance dropped by 90% since 1993 (Table 9). A similar decline was observed with the trawl (88%; Kline and Younk 1995). The decline in numbers was not a surprise. In 1992, only 60 kokanee were observed spawning in Alturas Lake Creek (Spaulding 1993). Escapement in 1993 was also extremely low (200). Additionally, no young of the year have been sampled in the trawl since 1992 (Kline and Younk 1995).

A number of hypotheses have been forwarded to explain the population crash. Some of the possibilities are listed here; (1) 0. ***nerka*** densities in **1990, 1991,** and 1992 reached levels that crashed the population, (2) beaver dams prevented fish **from** reaching spawning grounds, (3) **squawfish** predators controlled them, and (4) nutrient inputs are limiting fish production.

Determining which possibility or **combination** thereof that best describes the population crash is diicult. Some of the possibilities are reviewed below. First, compared to the other Stawtooth Valley Lakes, zooplankton biomass is exceptionally low in Alturas. Conversely, nutrient loading is very high, suggesting that the zooplankton are influenced more by predators than by nutrient limitation. If predation did collapse the zooplankton population, we should see an increase in zooplankton biomass because fish densities are now very low. To date, that increase has not occurred. However, it may take several years before the zooplankton population can rebound.

Secondly, beaver dams did not impede spawning in 1992 or 1993. The high numbers of fish observed in trawl surveys in 1990 and 1991 never entered the stream to spawn. Those strong cohorts should have contributed significantly to recent spawning populations. Conversely, very few spawners were observed, and barriers. did not stop migration (SBT; spawning observations).

Finally, **squawfish** are abundant and prey on 0. ***nerka*** in Alturas Lake (Teuscher et al. 1994). To what extent they contributed to the population decline is unknown. Additionally, we do not know to what extent juvenile squawfish compete with 0. ***nerka*** for zooplankton prey.

Spawning size is another interesting characteristic of Ahuras Lake. The mean spawning size in Alturas Lake Creek is 166 mm (42.1 **g).** However, a few extremely large kokanee (290 mm, -300 g) enter the creek. Age results indicated that most of the large and small spawners are age 3. Therefore, the large fish must either have foraged **differently** within Ahuras Lake or migrated to a **different** system. When comparing age 3 fish from both size groups, otolith examination showed similar first year growth patterns. Differences occurred in the second growing season, or similar to what is expected when comparing resident to sea-run samples.

The large fish may be outmigrating to Perkins Lake. Perkins Lake is a small system located just below Alturas. Zooplankton biomass in Perkins Lake is 300 times greater than in Alturas Lake (Teuscher 1993, unpublished data), and should support excellent **growth.** Interestingly, **Evermann** (1896) reported the mean size of what he termed small "redfish" in Alturas Lake Creek were 290

mm and 227 g. About 1,000 fish of that description were observed in the creek on September 7, 1895. **Evermann** (1896) reported that "the individuals of the small **form** vary but slightly in length and scarcely at all in weight. Hundreds ofindiiduals were weighed by us, and except near the end of the spawning season, the weight was always half a pound". So the average residual in 1895 was similar to the few large spawners that we observed in 1995, but much larger than our average of 55 g.

A final note, the 0. ***nerka*** population in **Alturas** Lake should be protected. Alturas and **Redfish** 0. ***nerka are*** the only population in the Stanley Basin that have maintained anadromous characteristics. Management activities in the area should reflect their importance. For example, we suggest that the fishing in Alturas Lake Creek be restricted. During our 1993 and 1994 spawning surveys, we observed 0. ***nerku*** being harvested from the stream. Illegal harvest of bull char -another sensitive species in the area- was also observed. Closing the stream to fishing from mid-July through the end of September would eliminate both problems. Additionally, resurrecting the broodstock program for the Alturas stock is warranted and necessary to meet the **NMFS** recovery goal that states three lakes must be used in recovery (Schmitten et al. 1995).

CHAPTER 2

LAKE CARRYING CAPACITY AND PRODUCTION **ESTIMATES**

Methods

We are anticipating that the broodstock program will provide substantial numbers of sockeye to be released in **Redfish** Lake in 1995. To ensure that the forage resources (primarily zooplankton) **of Redfish** Lake are not overtaxed by supplemented sockeye, we compared carrying capacity to estimated demand. The comparison allowed us to determine if lake fertilization or kokanee management programs should be implemented in 1995.

Lake Carrying Capacity

To estimate the current production potential of **Redfish** Lake, we reviewed methods reported in the literature. Most of the methods relate fish production to measures of productivity at a lower **trophic** level. The models used in our analysis included a chlorophyll **a** model (Chl **a;** Rieman and Meyers **1992),** a **euphotic** volume **(EV)** model **(Koenings** and Burkett **1987),** a total phosphorus (TP) method **(Plante** and Downing **1992),** and phytoplankton production **(PR)** measures **(Stockner** et al. 1995). A stock recruitment model (SR) was also evaluated **(Bowles** and Cochnauer 1984). Results from each model were converted to smolt carrying capacity in 10 gram equivalents. Converting output to 10 g equivalents was done to facilitate comparisons between models and to replicate historical smolt size observed for **Redfish** Lake.

Specific limnology data are required for each model. The limnology data was collected by Utah State University and is reviewed in a later section of this report. Equations for each production method are shown below:

Chla	log(y+1) = 1.188 + 0.586*chl a
TP	log(kg/ha) = -0.47 + 0.95(log TP)
EV	smolt biom. = -130 + 81.22(EV)
PR	PR units = (((daily PR * 0.0583) + 3.25); lake area); 1 PR unit = 9,200 10 g smolts
SR	production = 1,090 smolts / ha

A production workshop was completed to determine which technique best describes the carrying capacity of Redfish Lake. Personnel from IDFG, SBT, Bonneville Power Administration, Idaho Department of Environmental Quality, and John Stockner, an independent sockeye expert working for the Government of Canada Fisheries and Oceans participated.

Current Production

Current production was estimated using bioenergetics. Model parameters specific to sockeye are reported in Beauchamp et al. (1989). Model inputs include fish abundance, water temperature, diet, growth, and mortality rates. These parameters were estimated using hydroacoustics, temperature profiles, gillnet data, age analysis, adult escapement counts, and fry recruitment estimates.

A few of the initial fish parameters are shown in Table 10. Fish weight was estimated using trawl and age data. Abundance of each cohort mirrored hydroacoustic results, but we also considered fry recruitment and spawning counts. The model simulation ran from September 1993 to September 1994.

Table 10. Inputs to the bioenergetics model. September 15 is day 1 of the simulation.

Cohort	Day	Weight	Abundance
1	270	0.3	150,000
1	365	1.3	80,000
2	1	1.3	80,000
2	365	8.0	36,000
3	1	8.0	36,000
3	365	51.0	18,000
4	1	51.0	18,000
4	365	118.0	9,000

Results

Production Potential

Results from the production estimates varied by an order of magnitude. The EV model predicted that 1.2 million 10 g smolts can be produced by Redfish Lake. The chl a model provided a much lower result (140 thousand 10 g smolts). -The TP, chl a, and PR production estimates were similar (Table 10). Based on the recent success of the PR model, and the similarity in results between three of the models, we decided to compare the current production of kokanee in Redfish Lake to the production potential estimate using the PR model.

Current Production

0. *nerka* production for Redfish Lake between September 1993 and September

1994 was 2.41 million grams. This value is very similar to total biomass estimate of 0. *nerka* (2.56 million grams; based on hydroacoustic estimate of pelagic biomass + biomass of 9,200 creek spawners). The P/B ratio for Redfish Lake is 0.94.

The older age classes contributed the largest fraction of total production. The cohort contributing the most production (45%) was the year class beginning as age 1 and ending as age 2. Each cohort increased in age because day 1 in the model was 15 September. September was used as the start date because that is when abundance and growth information was collected. If we add production from the cohort of fish that emerge in May to the production of age' O-l cohort, combined production was only 19% of the total. Age 2-3 cohort made up 36% of the total.

Discussion

The Stockner model (EV model) indicated that 260,000 smolts could safely be produced by Redfish Lake. Most of the production (92%; 240,000 10 g smolt equivalents) is being used by kokanee. Available space for stocking sockeye is only 20,000 10 g smolts. If we exceed that number, the likely result would be poor growth and survival. Edmundson and Peltz (1990) and Zadina et al. (1993) reported declines in sockeye growth and smolt survival after supplementation exceeded lake carrying capacity. Fortunately, about 100,000 sockeye are available for release in Redfish Lake in 1995; well above lake carrying capacity.

Options for solving the shortfall in productivity include rearing 80,000 sockeye to smolt size or making room for them in the lake. Rearing to smolt stage would require hatchery space that is not available. More importantly, smolt releases preclude necessary development that occurs in the rearing environment (i.e., imprinting foraging, and predator avoidance). Instead, we suggest that space within Redfish Lake be made available. Freeing space for sockeye can be accomplished by increasing production potential with lake fertilization or replacing a portion of the kokanee biomass with sockeye. Based on estimates from the PR model, lake fertilization would stimulate zooplankton production sufficient to meet 1995 productivity needs. However, lake fertilization should not be considered a long term solution because the kokanee and sockeye will continue to compete for forage resources.

Generally, kokanee are rare in sockeye rearing lakes (Goodland et al. 1974; Ricker 193 8). The current scenario in Redfish Lake is different. Since 1991, for every sockeye that returned to Redfish Lake, 2,300 kokanee spawned in Fishhook Creek. To help restore a balance, some of the kokanee biomass should be replaced with sockeye. A 50% reduction in kokanee production would make space for over 100,000 juvenile sockeye (2.41 million g of kokanee / 2). Kokanee reduction can be accomplished by limiting fry recruitment from Fishhook Creek. Limiting fry recruitment to 50,000 a year would meet our goal.

Recovery Implications

The Snake River sockeye recovery goal is 2,000 adult returns to the Stanley Basin Lakes. Can we accomplish that goal by

maximizing smolt production in the rearing environment -everything else remaining constant? No.

The production potential of all the Stanley Basin Lakes is approximately 5 16,000 smolts. Under the current smolt/adult survival, 5 16,000 smolts could produce approximately 1000 sockeye (5 16,000 * 0.002 smolt/adult survival; assumes recent smolt to adult survival observed for Redfish Lake). Also, the proposed fertilization plans for Redfish Lake will increase production potential by an estimated 22%. Even if we could maximize smolt production and fertilize every Sawtooth Valley Lake, we could not achieve NMFS's modest recovery goal of 2,000 adults.

Given smolt carrying capacity limitation in the Sawtooth Valley Lakes, increasing smolt/adult survival is needed to meet recovery goals. In Bjornn et al. (1968), average smolt to adult survival between 1955 and 1964 for Redfish Lake was 0.008. Bowles and Cochnauer (1984) estimated that prior to dam construction, smolt to adult survival for Alturas Lake was 0.04. Returning smolt to adult survival to that measured by Bjornn et al. (1968) would increase adult returns by a factor of 4. More importantly, the recovery goal of 2,000 adult returns could be reached by producing a realistic 250,000 smolts. The goal of the research presented here is to ensure the best rearing environment possible. However, the burden of recovery depends on increasing migration survival. Without meaningful changes in migration survival, recovery is impossible.

Table 10. Yearly production potential estimates for 0. *nerka* in the Sawtooth Valley Lakes. The Chl a model is an estimate of total biomass. We converted biomass to production assuming a P/B ratio of .94.

Lake	Chl a	TP	PR	SR	EV
Redfish	**199,000**	156,000	260,000	670,000	**1,200,000**
Alturas	119,000	82,000	147,000	368,000	539,000
Stanley	33,000	23,000	46,000	88,000	84,000
Pettit	5 1.000	20.000	63.000	137.000	287.000
TOTALS	402,000	281,000	516,000	1,263,000	2,110,000

Chla Chlorophyll a model
TP Total Phosphorus model
PR Primary production rate model
SR Stock recruitment model
EV Euphotic volume model

CHAPTER 3

HABITAT RESEARCH

Methods

Spawning Habitat Survey

The purpose of this study was to determine the quality of habitat available to beach spawning 0. ***nerka*** populations in the Stanley Basin Lakes. In 1993, we completed a substrate survey of known spawning areas in **Redfish** Lake. Methods used to complete the visual surveys were reported in Teuscher et al. 1994. In 1994, we collected substrate using a Ponar dredge. The dredge samples were taken to collaborate the visual data collected in 1993, and to provide a more quantitative review of substrate composition.

The Ponar dredge sampled a 225 cm^2 area. Sample depth varied with substrate composition. In areas with abundant fines, the dredge went deep (15 cm). The dredge did not penetrate gravel substrate as well. However, the dredge sampled deep enough to replicated the depth of a redd built by a residual sized 0. ***nerka (240 mm).***

In the lab, samples were filtered through a standard set of screens. Volume measurements were recorded for each particle size. For analysis, substrate sizes were recorded as percent of total volume sampled.

Dredge samples were collected in **Redfish,** Stanley, and Pettit Lakes. Sample areas for each lake are shown in Figure 13. In each area, six sites were sampled at a depth of 1 m. Sample sites were separated by a distance of at least 20 m. To accurately represent substrate composition at each site, three psuedoreplicates were collected. We used a one way **ANOVA** to compare substrate composition among sample areas.

Results and Discussion

In **Redfish** Lake, one of the Sockeye Beach sites and the south end of **Redfish** Lake had the lowest values for percent fines < 6.3 mm. Egg to survival should be good (30 to ***70%)*** in those areas based on values reported by Bjornn and Reiser (1991). Interestingly, those same areas are where sockeye redds have been observed and residual spawners concentrate.

For dredge samples, percent fines ranged from 30 to 84%. Visual estimates of percent fines **< 6.3mm** were consistently higher and more variable (range 2.7 to 89.6 **%;** Table 12). The dredge may slightly underestimate percent fines due to small particles escaping during dredge retrieval. Conversely, visual estimates only describe surface particles and may overestimate percent fines.

Spawning habitat in **Redfish** Lake is not limiting Recommended space requirement per spawning pair is 6.7 m (Burner 1951). If we sum the area at depths between 0.3 and 3 m from sites with low percent fines (sites 1, 3, and 8; Figure 15) and divide by 6.7, there is ideal spawning space for approximately 15,700 sockeye in **Redfish** Lake. Less productive spawning sites can be used if Sockeye Beach and the south end of the lake become fully seeded. In the near future, however, we do not anticipate escapement values any where near what ideal spawning areas can support.

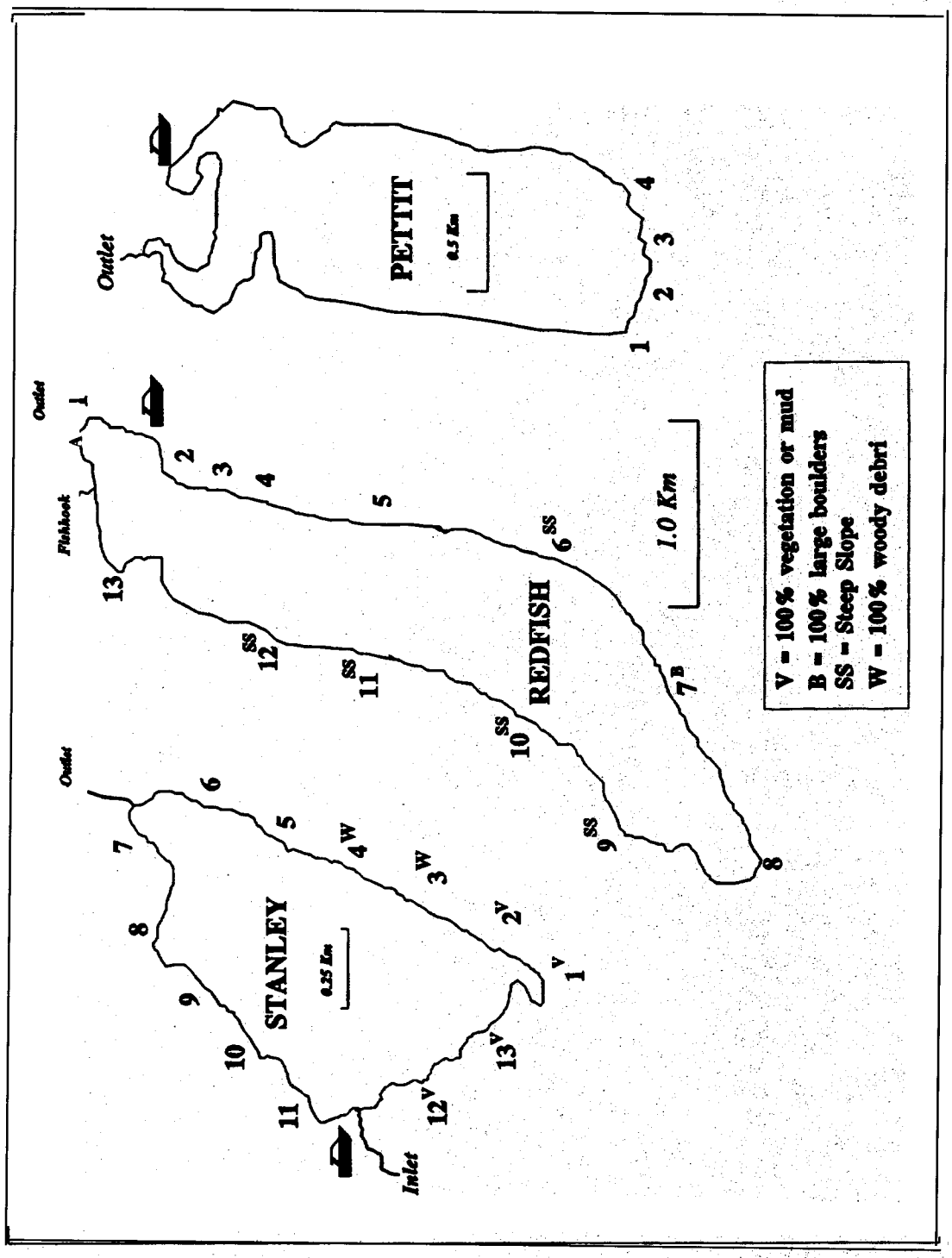

Figure 15. Sample locations for spawning habitat surveys in three Sawtooth Valley Lakes.

Table 12. Estimates of percent fines < 6.3 mm (s.d.) in Redfish Lake. See Figure 15 for site locations. * Indicates areas where slope was to steep to sample with Ponar dredge. Sites 2-4 are in the Sockeye Beach area.

Site	Dredge	Snorkel	%Slope
1	30.3 (19.3)	74.9 (31.8)	2.3
2	59.9 (3.1)	82.4 (25.3)	0.8
3	30.5 (11.4)	67.3 (27.4)	5.0
4	84.2 (17.9)	89.6 (21.1)	12.5
5	44.6 (33.3)	79.2 (17.9)	5.7
6	*	2.7 (6.6)	30.9
7	*	16.7 (20.4)	19.6
8	33.3 (19.1)	39.0 (33.6)	13.2
9	*	46.7 (3 1.4)	11.3
10	*	17.5 (18.9)	19.0
11	*	34.1 (29.2)	30.0
12	*	44.7 (24.2)	16.7
13	70.6 (14.9)	86.5 (16.9)	11.1

Redfish Lake stocks may be introduced into Pettit Lake in 1995. If the releases are successful, shoal spawning habitat should be available. Results in Pettit Lake varied, but the average percent fines were low (Figure 13).

Vegetation and woody debri covered a large portion of the shoal area in Stanley Lake. Site 1 - 4, 12 and 13 were judged as unsuitable for spawning (Figure 13). The areas void of woody debris or vegetation may support spawning. Percent fines in site 5 and 8 were low (Figure 13).

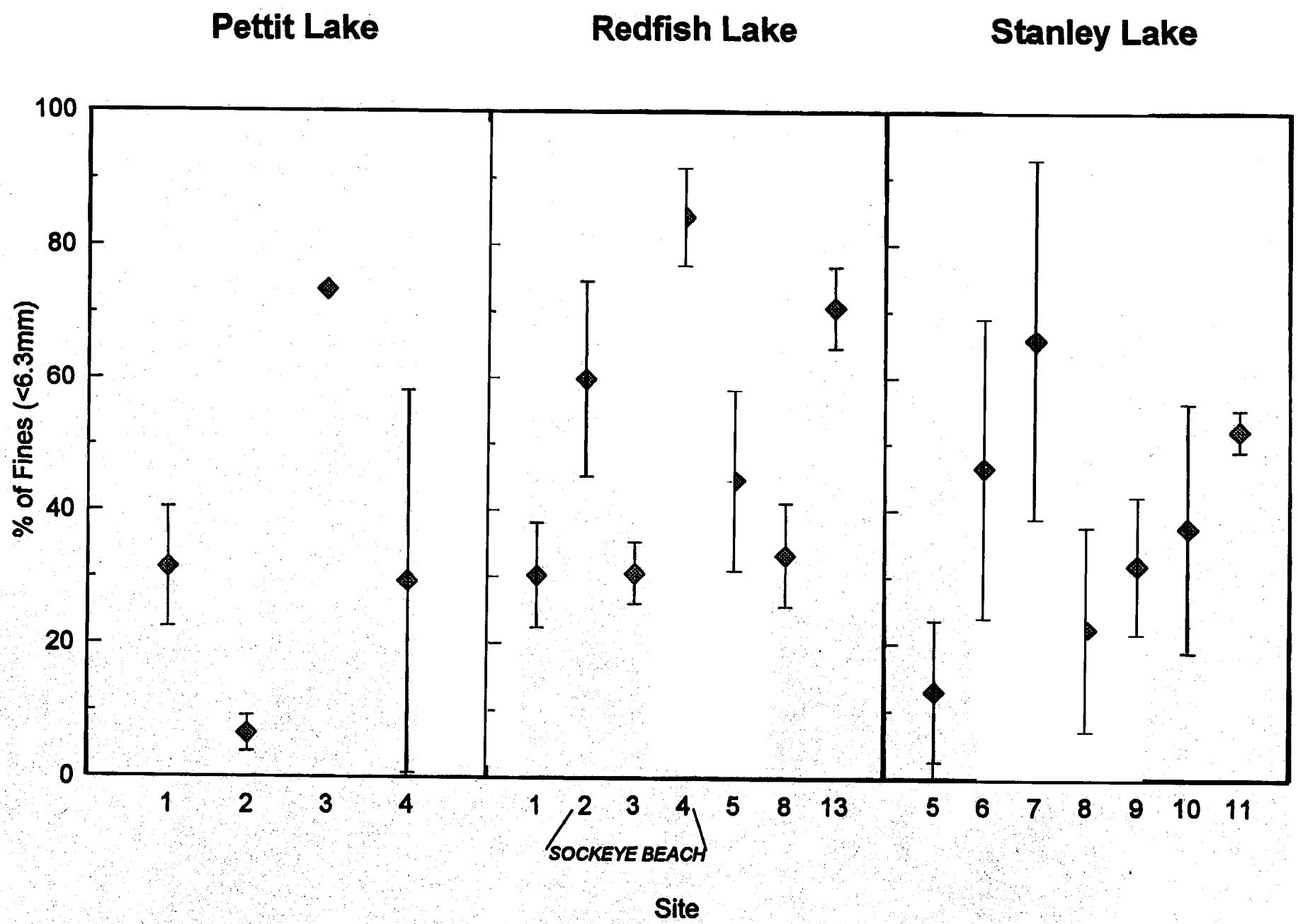

Figure 16. Percentage of fines < 6.3 mm from Ponar dredge samples.

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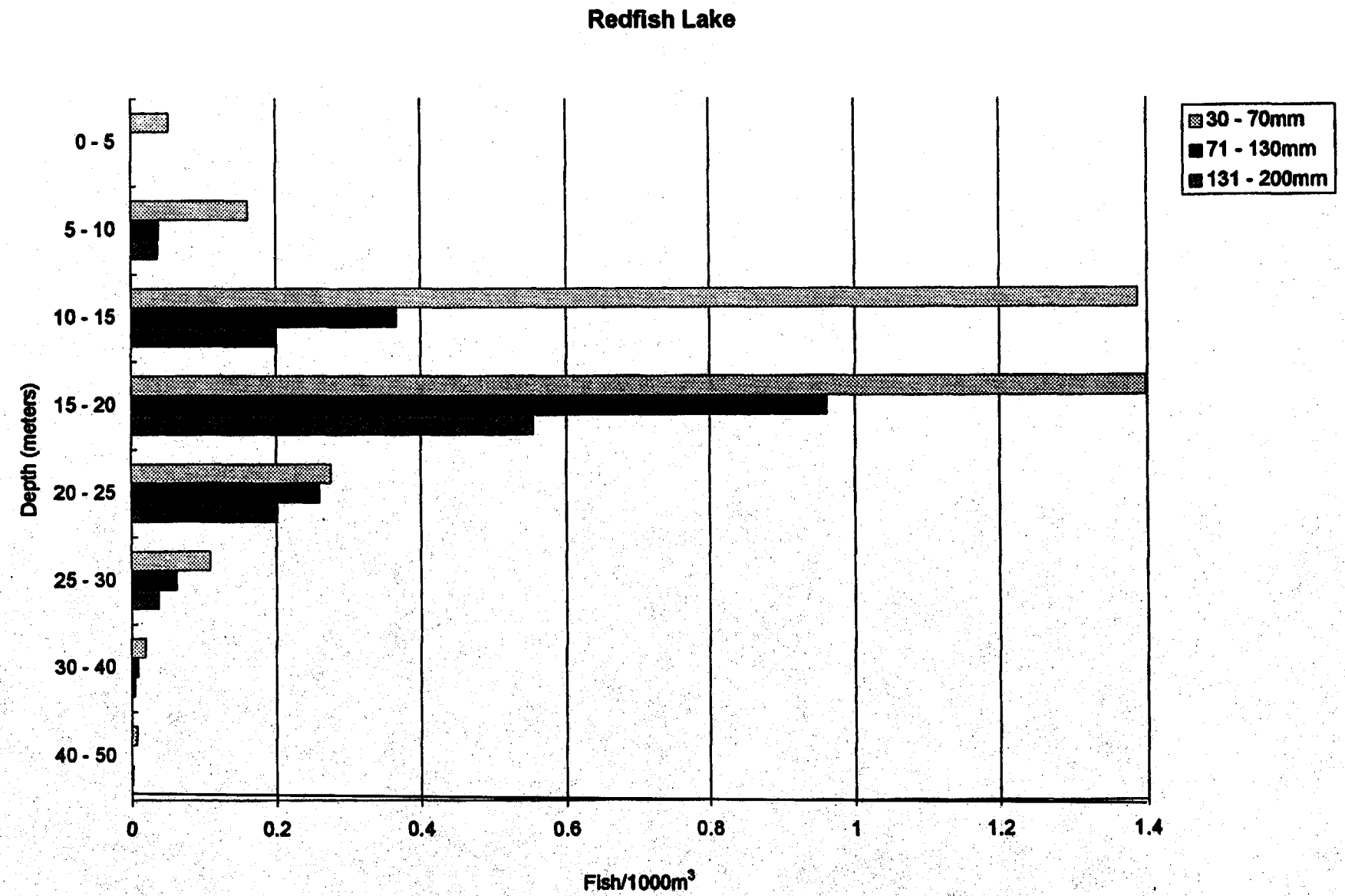
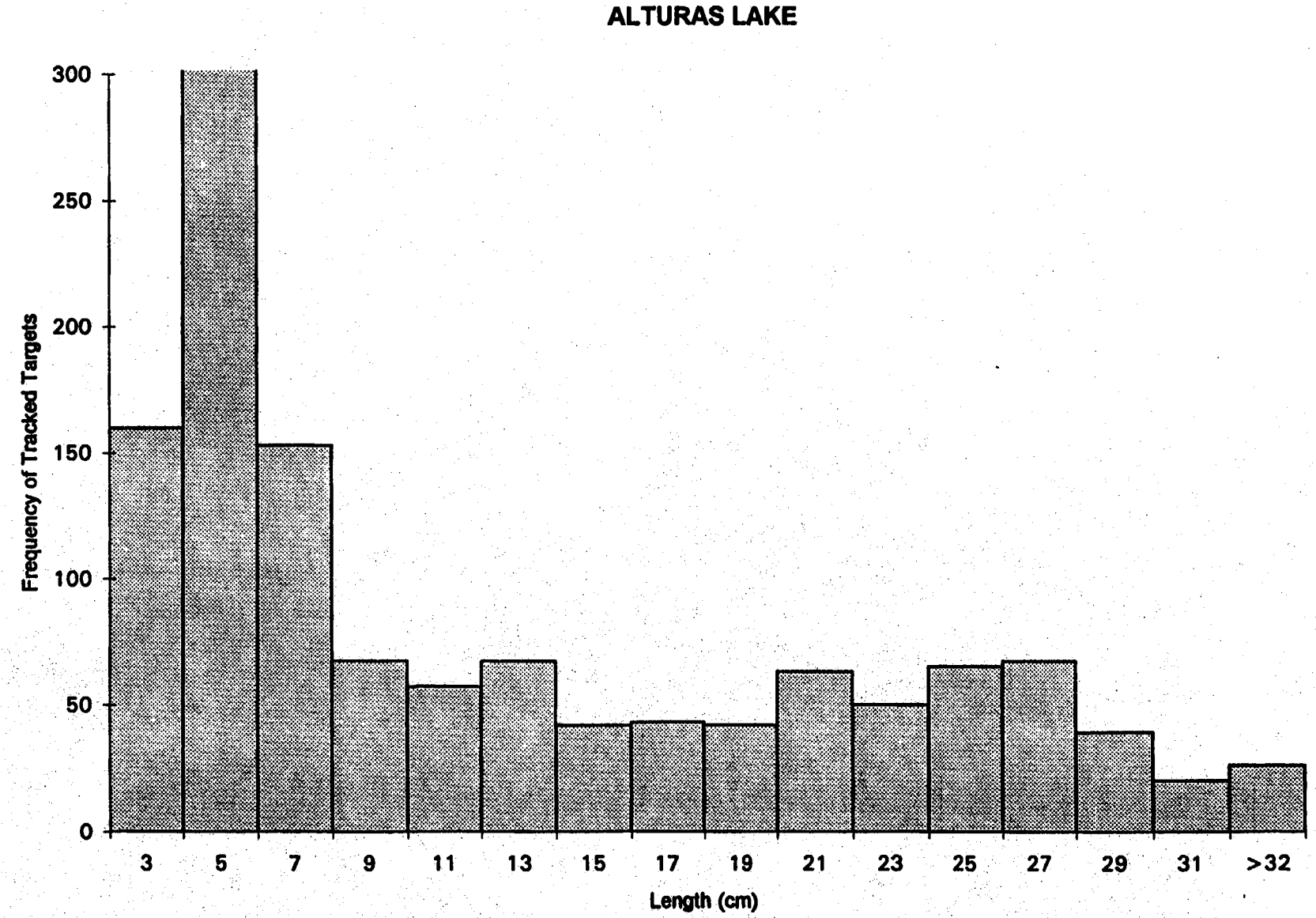
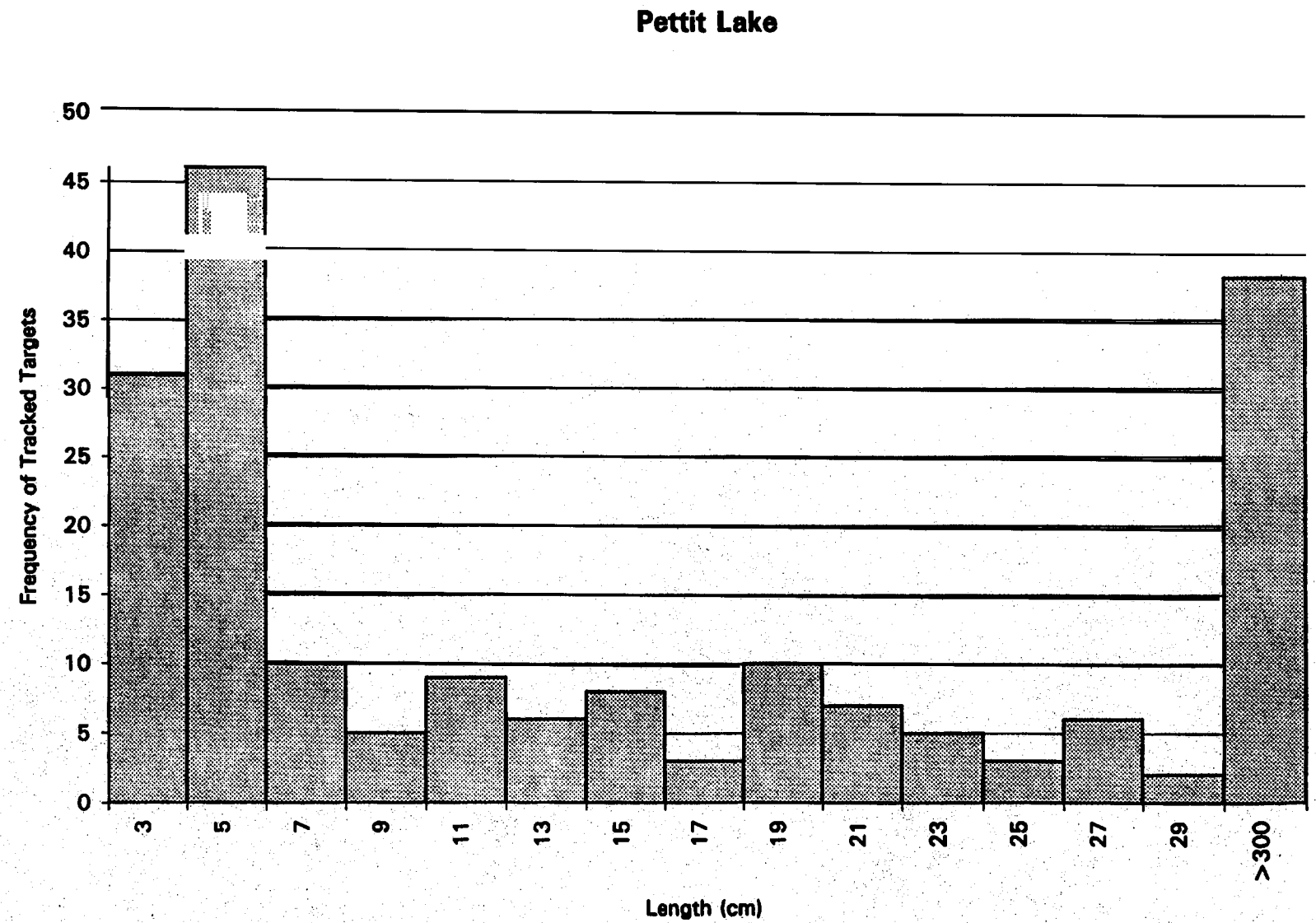
Appendix 1

Additional vertical distribution and length **frequency** information.

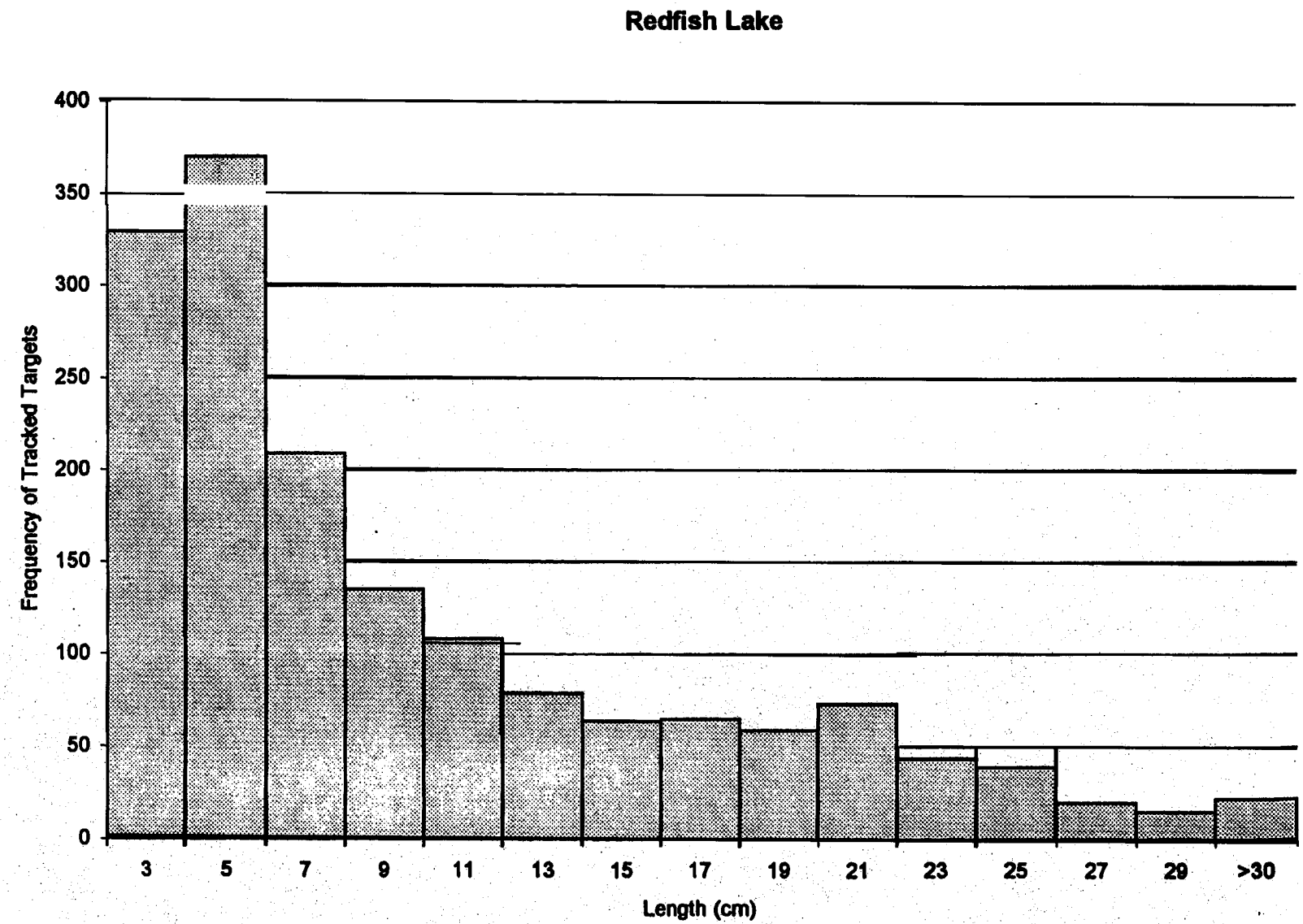

Redfish Lake
30 - 70mm
71 - 130mm
131 - 200mm
Depth (meters)
0 - 5
5 - 10
10 - 15
15 - 20
20 - 25
25 - 30
30 - 40
40 - 50
0
0.2
0.4
0.6
0.8
1
1.2
1.4
Fish/1000m3



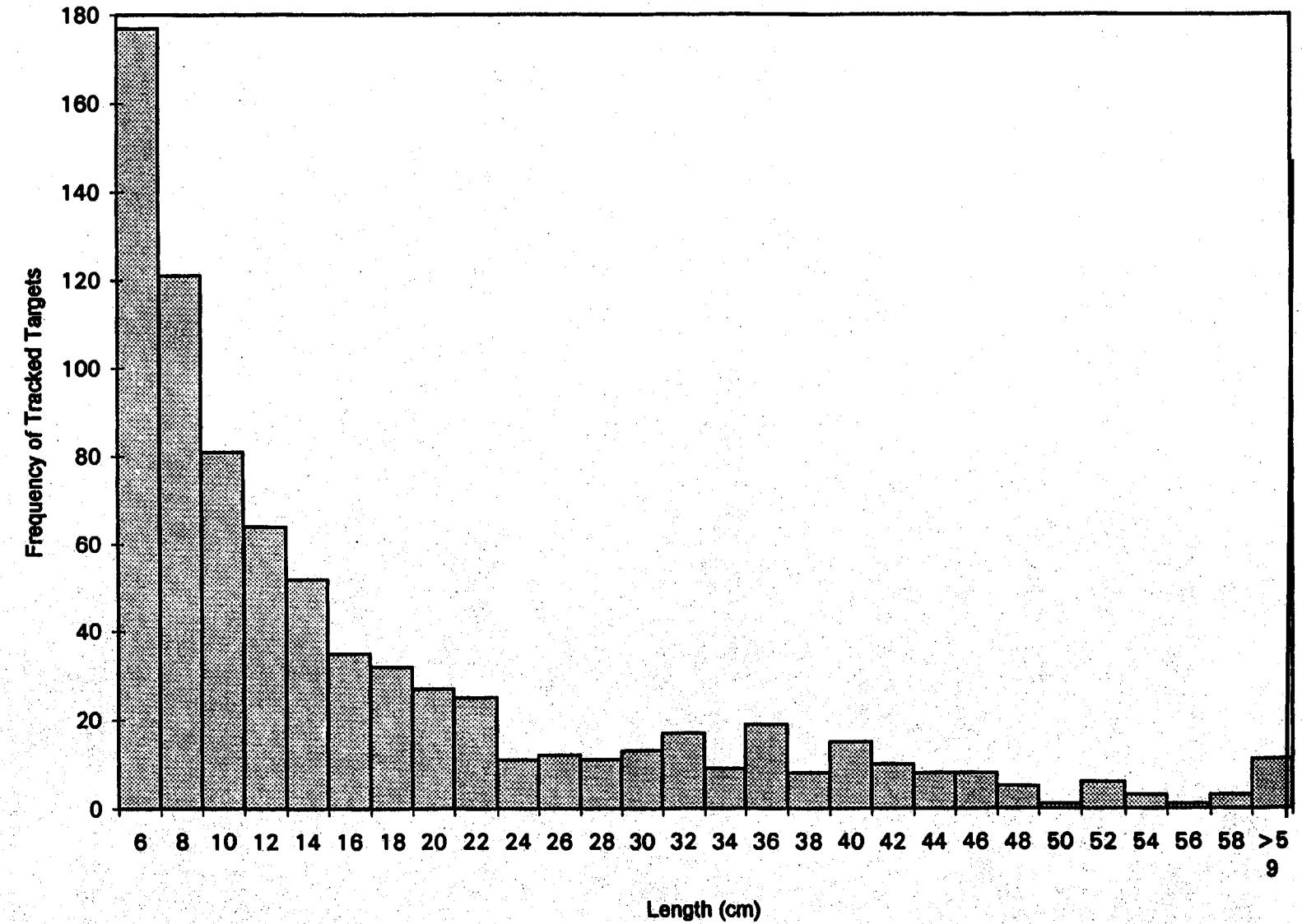
Length distribution of tracked targets during September 1994 hydroacoustic surveys.



Length distribution of tracked targets during September 1994 hydroacoustic surveys.



Length distribution of tracked targets during September 1994 hydroacoustic surveys.



Length distributionof tracked targets during September 1994 hydroacoustic surveys.

Key:

daph = **Daphnia**	**holo=Holopedium**	**bosm=Bosmina**	**poly=Polyphemus**
cala=calanoid	cycl=cyclopoid	**chirP=chironomid** pupa	**chirL=chironomid** larva
euri=Euricyrus	amp=amphipod	**TI=terestial insects**	**AI=Aquatic Insects**
mol=moluska	**odo=odonata**	**plt=plant** material	**scl=sculpin**
rss=redside shiners	**misc=miscellaneous**	**O.n=*O. nerka***	**suk=sucker**
udf=unidentified fish	**olg=oligocheat**	**dip=dipteria**	**zoo=zooplankton**

Appendix 2A. 0. ***nerka*** diet analysis in the Sawtooth Valley Lakes for 1993 and 1994. Values are **percent** by weight.

Lake	Date	N	size mm	daph	holo	bosm	poly	cala	cycl	chirP	chirL	euri	amp	TI
Redfish	June 1993	33	23-157	Ø.Ø	Ø.Ø	8.8	Ø.2	Ø.Ø	2.8	80.1	2.2	1.0	0.0	4.9
Redfish	June 1994	5	69-182	0.0	0.0	22.6	Ø.Ø	Ø.Ø	Ø.Ø	37.7	39.7	Ø.Ø	Ø.Ø	Ø.Ø
Redfish	Sep 1993	20	20-200	41.Ø	9.1	3.7	7.7	Ø.3	8.8	18.7	5.8	Ø.Ø	Ø.Ø	4.9
Redfish	**Sep 1994**	**12**	**-**	**10.2**	**54.4**	**2.0**		**2.0**	**0.0**	**8.1**	**0.0**	**7.1**	**0.0**	**0.0 16.2**
Stanley	June 1993	26	170-229	0.0	0.0	0.0	0.0	0.0	5.2	92.7	1.0	0.6	0.1	0.4
Stanley	June 1994	10	1 **10-232**	6.4	Ø.4	Ø.3	Ø.Ø	48.6	Ø.Ø	37.5	6.8	Ø.Ø	Ø.Ø	Ø.Ø
Stanley	Sep 1993	12	70-220	54.4	4.7	15.9	0.0	0.4	6.0	8.4	0.0	0.0	0.0	10.2
Stanley	Sep 1994	21	85-207	57.1	5.Ø	Ø.2	Ø.Ø	4.5	29.9	Ø.Ø	Ø.Ø	Ø.Ø	Ø.Ø	3.3
Stanley	**Dec** 1993	5	94-197	6.6	Ø.Ø	1.3	Ø.5	71.5	20.1	Ø.Ø	Ø.Ø	Ø.Ø	Ø.Ø	Ø.Ø
Stanley	Jan 1994	7	112-184	Ø.Ø	0.0	Ø.2	Ø.Ø	33.7	66.1	Ø.Ø	Ø.Ø	Ø.Ø	Ø.Ø	Ø.Ø
Stanley	**Oct 1994**	**13**	**159-247**	**78.0**	**10.3**	**0.5**	**0.0**	**11.2**	**0.0**	**0.0**	**0.0**	**0.0**	**0.0**	**0.0**
Alturas	June 1993	18	66-134	Ø.Ø	Ø.Ø	Ø.Ø	Ø.Ø	Ø.Ø	2.3	97.6	0.0	0.0	0.0	0.1
Alturas	June 1994	29	**100-145**	Ø.3	Ø.Ø	3.2	Ø.Ø	Ø.Ø	Ø.Ø	89.3	3.8	Ø.Ø	Ø.Ø	3.4
Alturas	Sep 1993	20	-	2.1	Ø.Ø	13.1	36.6	Ø.Ø	3.3	2.8	3.9	3.7	Ø.Ø	34.5
Alturas	Sep.1994	**3**	**122-154**	**0.0**	**0.0**	**0.0**	**0.0**	**0.0**	**0.0**	**32.5**	0.0	67.4	**0.1**	**0.0**
Pettit	June 1993	0	-											
Pettit	June 1994	22	89-268	59.1	0.0	5.7	1.2	3.4	5.1	24.6	1.0	0.0	0.0	0.0
Pettit	sep 1993	15	240-275	100.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Pettit	**Sep** 1994	22	1 **10-288**	87.5	Ø.Ø	Ø.Ø	Ø.Ø	10.3	1.9	Ø.3	Ø.Ø	Ø.Ø	Ø.Ø	Ø.Ø

Appendix 2B. Rainbow trout diet in the Sawtooth Valley Lakes in 1993 and 1994. The 1993 samples were collected near shore wi
samples were collected with vertical gillnets set away from shore. Values are percent by weight. See first page of appendix for abb

Lake	Date	N	size range mm	zoo	chirP	chirL	TI	AI	mol	odo	plt	scl
Alturas	June 1993	6	220-271	0.0	15.4	0.0	71.4	5.8	1.1	0.0	6.3	0.0
Alturas	June 1994	4	230-261	0.0	99)2	0.0	0.0	0.0	0.0	0.0	0.8	
Alturas	Sep 1992	17	220-300	0.0	7.8	0.2	16.3	0.4	17.7	1.7	45.1	10.8
Alturas	Sep 1993	13	209-289	0.0	0.4	6.5	2.6	26.7	0.0	0.0	57.3	4.7
Alturas	Sep 1994	3	249-286	0.0	19.0	0.0	17.9	0.6	0.0	0.0	47.5	0.0
Alturas	Dec 1993	9	241-295	0.0	0.0	0.0	0.0	8.6	0.1	0.0	84.1	0.0
Stanley	June 1993	14	230-278	0.0	71.1	0.0	10.3	0.6	0.0	0.0	13.4	0.0
Stanley	Sep 1992	19	234-295	11.4	0.7	0.0	28.0	2.9	10.3	0.8	45.9	0.0
Stanley	Sep 1993	23	220-285	17.2	2.9	3.9	34.1	2.7	22.8	0.7	3.3	0.0
Stanley	Sep 1994	8	230-296	2.9	0.0	0.4	6.2	6.5	28.1	0.0	55.9	0.0
Pettit	Sep 1992	10	249-300	0.0	20.9	0.0	32.8	2.2	17.3	20.5	6.3	0.0
Pettit	Sep 1993	20	210-333	0.0	31.0	0.0	10.8	0.6	18.5	21.0	8.1	0.0
Pettit	Sep 1994	5	254-278	0.0	0.0	0.0	57.9	1.0	0.0	20.9	20.2	0.0

Appendix 2C. Squawfish (sqw), bull trout (bult), lake trout (LT), and redside shiner (rss) diets in Alturas, Stanley, and Pettit Lakes. Data are percent by weight.

Lake	Sp.	Date	size range	N	O.n	suk	scl	rss	sdf	mol	odo	cbrP	cbrL	Al	Tl	olg	dip	zoo	plt	misc
Alt	sqw	6193	172-278	50	16.2	2.6	11.4	0.0	2.4	0.0	1.4	0.4	0.0	3.8	19.5	27.7	12.4	0.0	1.3	0.9
Alt	sqw	9192	230-439	20	20.0	10.0	0.0	30,0	20.0	20.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Alt	sqw	9193	134-480	62	16.1	23.1	0.0	0.0	16.0	33.7	0.0	0.2	0.0	0.4	0.0	0.0	3.2	0.0	4.3	3.0
Alt	sqw	12/93	94-235	26	0.0	7,7	38.3	0.0	0.0	7.7	0.0	0.0	7.7	30.8	0.0	0.0	0.0	0.1	7.7	0.0
Alt	bult	9/93	329-540	19	70.8	12.4	0.0	0.0	16.8	0.0	0,0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Alt	bult	1993	297-545	10	88.5	0.0	0.0	0.0	11.1	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Stan	LT	6/93	253-654	12	3.2	0.0	0.0	19.2	8.9	0.0	0.0	66.4	0.0	0.0	0.0	0.8	0.0	0.0	0.0	1.5
Stan	LT	9/92	220-660	13	90.5	0.0	0.0	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	8.6
Stan	LT	9/93	295-699	7	83.1	0.0	0.0	0.0	7.1	0.0	0.0	0.1	9.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Stan	LT	9/94	288-618	14	89.8	0.0	0.0	0.0	0.0	0.0	0.0	10.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Pett	rss	6/93	127-145	10	0.0	0.0	0.0	0.0	0.0	9.8	69.5	0.0	0.0	5.2	0.0	0.0	0.0	0.0	0.0	15.5
Pett	rss	9/93	119-148	16	0.0	0.0	0.0	0.0	0.0	9.0	91.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Pett	rss	9/94	76-89	6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	83.3	16.7	0.0	0.0	0.0	0.0

LIMNOLOGY RESEARCH

Prepared for

The Shoshone-Bannock Tribes

Prepared by

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CHAPTER 1

Limnological Investigations of Sawtooth Valley Lakes, 1994

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Chris Luecke

Wayne A. Wurtsbaugh

INTRODUCTION

In 1994 we continued monitoring the limnological characteristics of Redfish, Alturas, Pettit and Stanley lakes in the Sawtooth Valley of central Idaho. This marked the third consecutive year of monitoring under natural conditions, in anticipation of large scale manipulations of the lakes in succeeding years. In 1995 a lake fertilization program in Redfish Lake is planned to coincide with releases of juvenile sockeye salmon from the endangered-fish broodstock program. Lower densities of juvenile sockeye salmon may also be released into Pettit Lake in 1995. The limnological information collected on the four lakes in 1992-94 will be essential to assess effects of lake fertilization and fish stocking on the structure and function of the lake food web.

Limnological sampling conducted in 1992 and 1993 indicated that the rankings of abundances of phytoplankton and zooplankton among the lakes was similar (Steinhart et al. 1994). Redfish and Pettit lakes had the lowest mean summer concentrations of chlorophyll in surface waters, Stanley Lake had the highest concentration, and Alturas was intermediate. This ranking of phytoplankton abundance was associated with similar rankings of nutrient concentrations, indicating that phytoplankton abundance was closely linked to bottom-up processes in these lakes. The ranking of zooplankton biomass in the four lakes in 1992-93 was also consistent among years, although rankings were different for zooplankton compared to phytoplankton characteristics. Highest mean summer zooplankton biomasses were present in Pettit, followed by Stanley, Redfish and Alturas Lakes (Steinhart et al. 1994). This ranking of zooplankton biomasses was associated with kokanee densities, suggesting that the top-down effects of planktivorous fish had greater effects on zooplankton biomass than did bottom-up effects of phytoplankton food resources.

In spite of the similar rankings of trophic level biomasses among, the lakes in diierent years, differences in seasonal development and mean concentration of organisms within a given lake was apparent between 1992 and 1993. The warm, dry summer of 1992 resulted in lower concentrations of chlorophyll compared to the cool, wet summer of 1993. The early summer increase in zooplankton biomass was delayed in most lakes in 1993 compared to 1992, possibly due to lower water temperatures. Biomass of zooplankton in late summer was similar between years in Redfish, Pettit and Stanley Lakes (Steinhart et al. 1994).

In 1994 we are particularly interested in determining whether these trends in inter-lake and inter-annual limnological characteristics persist. Observations from '1992-93 indicated that inter-take patterns ofvariation were greater than inter-annual patterns. If these patterns hold, then management plans for enhancement of juvenile sockeye salmon can be formulated on a lake by lake basis. Less emphasis would be needed concerning the vagaries of annual weather patterns and lake trophic structure.

Methods

During 1994 we sampled the Sawtooth Valley Lakes approximately once a month. Sampling occurred at a deep station in each lake. During the ice-free periods we sampled from a boat; however, from January through April, and in December, the lakes were ice covered so sampling was performed through a hole cut in the ice using a hand-powered auger.

Temperature, Oxygen, Conductivity and Light

Temperature, dissolved oxygen 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). During ice-covered periods, the underwater light probe was extended horizontally away from the hole by a 3.7-m long PVC pole. This pole was used to keep the light sensor away from light that penetrated through the hole. Water transparency was measured using a 25-cm black and white Secchi disk.

Water Chemistry

Water was collected for water chemistry at all sample dates; however, on some dates not all parameters were analyzed for all nutrient components. The epilimnion was sampled with a depth-integrating 6-m Tygon tube. Two to four additional depths were sampled using a 4-L Van Dom bottle. We normally 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 reactive phosphorous (SRP)] were filtered before freezing with a 0.45 μm acetate 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,-N. TP and SRP were analyzed calorimetrically in our laboratory (Utah State University LIMNOLOGY Laboratory) using the molybdate - ascorbic acid method (Steinhart et al. 1993). Nitrogen analyses were conducted by the University of California, Davis using the Kjeldahl digestion method for TKN, the hydrazine method for NO,, and the indophenol method for NH_4.

Chlorophyll

Epilimnetic chlorophyll samples were taken from the 6-O m depth-integrated tube samples and deeper samples with the Van Dom bottle. Epilimnetic and discrete depth samples were taken at the central, index station, and epilimnetic samples were also taken at two additional stations 200-500 m fiom 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 accorrlmg to (Steinhart et al. 1993, Holm-Hansen and Riemann 1978).

Phytoolankton

Samples for phytoplankton enumeration and identification were also taken using the 6-O m Tygon tube above: and the Van Dom water bottle described above. Water samples for phytoplankton analyses were preserved using Lugol's iodine solution. A 100 ml aliquot from each sample was filtered through a 0.45 pm tied-ester filter (Millipore HAWP). The filters were cleared and permanently mounted, according to the method of Crumpton (1987). Cells were counted and a minimum of 10 individuals in each taxa were measured to calculate biovolume (Wetzel and Likens 1991; Steinhart et al. 1993).

Zooplankton

Zooplankton were sampled at three stations in each lake on most dates. By sampling several stations, we hoped to improve minimize variance due to patchiness of the zooplankton. Zooplankton were collected using a standard closing zooplankton net (35.5-cm diameter by 158-cm long with 80 pm mesh). A General Oceanics flow meter, modiied 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 shallow Stanley Lake, zooplankton tows were only taken from 10-O m and from the bottom to 10 m. Zooplankton samples were preserved in a buffered 5% 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 (see Steinhart et al. 1993); however, in this report we refer to each species by genus. Eggs were counted, and egg ratios (eggs/female) calculated for each cladoceran species to identify periods of reproduction, and as a surrogate measure of surplus energy.

Stream discharge and chemistry

We measured discharges in the two tributaries of Redfish Lake, Fishhook Creek and Redfish Creek. Discharge was measured using a Marsh-McBirney Flo-Mate 2000 electromagnetic flow meter, or estimated using gage readings and a relationship *between gage* height and discharge (Gross and Wurtsbaugh 1994). Depth-integrated samples for nutrient analyses were stored in Nalgene polyethylene bottles. Bottles were rinsed first with 0.1 N HCI and then three times with aliquots of the actual sample. Samples were stored in an ice cooler, frozen upon return to our field laboratory, and subsequently analyzed for total phosphorus and total Kjeldahl nitrogen as described previously.

Results

Temperature, Oxygen, and Conductivity

The Sawtooth Valley Lakes sampled in this study have diierent morphometries (Table 1). In general, Redfish Lake is the

largest, deepest and clearest lake. Alturas and Pettit Lakes are intermediate in size, depth, and clarity. Stanley Lake is the smallest, shallowest, and least transparent of the sample lakes.

Seasonal and yearly differences in epilimnetic and hypolimnetic temperatures are shown in (Figure 1). In January, under the ice, there was an inverse temperature stratification in the lakes (Figure 2). The lakes thawed in early May and *remained* ice-free until late November. From June-October there was a well developed thermocline was present in all the lakes (Figure 2). The thermocline developed very quickly after ice-out in May, and was well developed by the end of June. In contrast, during the cool and wet summer of 1993, the thermocline did not become well developed until August. Stanley Lake had already turned-over by November 4; however, the other lakes had not turned-over by that date.

Epilimnetic temperatures in 1994 were comparable to 1992, but 1993 was clearly a colder summer (Figure 1). Maximum surface temperatures in 1992 and 1994 were around 18°C. In 1993, however, maximum surface temperatures barely reached 16°C. There was little change in the hypolimnetic temperatures in the lakes during the course of this study.

Dissolved oxygen levels were similar in 1992, 1993 and 1994, although values were slightly lower this year. During ice-covered periods, the lakes were super-saturated immediately below the ice (Figure 2). For much of the water column, dissolved oxygen levels were sufficient (> 5 mg/L) for salmonids during all times of the year. In Pettit and Stanley Lakes, however, dissolved oxygen levels were less than 5 mg/L near the bottom (Figure 2).

Year-round conductivity sampling revealed that Pettit, and to a less degree, Stanley Lake are meromictic, and consequently are not mixing to the bottom with each overturn. Conductivities rapidly increase around 35 m in Pettit, and around 17 m in Stanley, indicating that lakes are salt-stratified. In general, water column conductivities in Redfish, Pettit, and Stanley Lakes are less than 35 µS. Alturas lake has slightly more salts, and a conductivity above 55 µS.

Water Transparency

Light profile measurements indicated that the photic zone in the clearest lakes (Redfish and Pettit) extended to 20-35 m, while in the more productive Stanley Lake it varied between 12-18 m. The seasonal variation in the compensation point for photosynthesis (nominally 1% of surface light intensity) was similar between 1993 and 1994 (Figure 3). The compensation point in all lakes gradually increased as summer progressed with the deepest compensation point observed in October in 1993 and in July in 1994. The compensation point then began to decrease in winter culminating in levels immediately below the ice in January and December of both years. Mean annual extinction coefficients in the lakes during 1994 varied from 0.16 to 0.32 (Table 1).

Secchi depths, another measure of water clarity, were seasonally variable in the Sawtooth Valley Lakes (Figure 4). Water transparency was lowest after ice-out in May. and increased until the fall turnover. Stanley Lake had. the lowest mean Secchi depth, while Redfish, Alturas and Pettit

Lakes had somewhat similar transparencies (Table 1). Secchi depths in 1994 were slightly shallower than in 1993 (Figure 4). Secchi depths (m) were related to the extinction coefficient (v) by the following equation (also see Figure 5):

$v = 0.639 - 0.404 \text{ LOG (Secchi depth)}$
$R^2 = 0.764; n = 29$

This equation will allow us to use the easily-measured Secchi depths to predict the extinction coefficient, and hence the depth of the photic zone, ligbt available for fish feeding, etc.

Stream Discharge and Nutrients

Runoff in the Sawtooth Basin was extremely low in 1994. As indicated by the hydrograph of Fishhook Creek (Figure 6), discharges in 1994 were slightly lower than those in 1992, which was previously the most severe drought year recorded for the upper Salmon River drainage. Integrated annual discharge indicates that the 1994 flow was only 87% of that in 1992, and only 42% of the: flow in 1993, which was a near-normal water year.

Total phosphorus and nitrogen concentrations in the inflows were extremely low, and except for periods of high discharge, similar to those in 1992 and 1993 (Appendix 3). In 1994 mean annual total phosphorus and total nitrogen (TKN) concentrations were 10 and 81 µg/L (Table 2). The resulting N:P ratio for the runoff was 8: 1.

Because nutrient concentrations in 1994 were similar to other years, but discharges were extremely low, nutrient loading to Redfish Lake this year, was also low. We are unable to calculate the exact load because nutrient samples were not collected during the first 8-days of the rising hydrograph in May. Exclusion of this period would result in a significant underestimate of the total annual loading. In 1993, for example, 39% of both the nitrogen and the phosphorus was loaded during the first S-days of the spring rising hydrograph. Comparison of the 1992 and 1993 data for Redfish Lake indicates that nutrient loading was approximately proportional to annual discharge (Gross and Wurtsbaugh 1994). Consequently, our data suggest that nutrient loading to Redfish Lake in 1994 was approximately 40% of that measured in the normal water year in 1993.

Lake Water Chemistry

Epiietic total phosphorous varied little among the four lakes throughout 1994, and there was little seasonality (Figure 7, top). Total phosphorous (TP) values generally ranged from a low of 3 µg/l in Pettit Lake in April to a high of ~15 µg/l in Alturas Lake in September. A high peak was also observed in Stanley Lake in November, shortly after fall turnover. This peek was not evident in Redfish Lake, but it had not mixed completely by November (Pettit and Alturas were not sampled on this date). No increase in total phosphorous was apparent at the time of spring overturn (May).

Total nitrogen (TN) ranged from a high of ~ 180 µg/l in early June to a low of -50 g/l in Redfish Lake in early July (Figure '7, bottom). TN in Alturas and Pettit was slightly higher than in Redfish and Stanley Lakes with the exception of a high value in Stanley in Late September.

Vertical depth profiles of soluble nutrients (NO_3, NH_4, SRP) in Redfish Lake showed an increase with depth occurring primarily at sample depths deeper than 25 m (Figure 8). This pattern became more pronounced as the summer progressed. Missing values in the figure occur because SRP (soluble reactive phosphorous) values were often below our detection limit of 2 µg/l. Similar profiles of soluble nutrients are available for the other three lakes in Appendix 2.

Vertical depth profiles of TN and TP in Redfish Lake showed that these nutrients were usually lowest in the epilimnion and increased slightly in the metalimnion (Figure 9). TP concentrations in the hypolimnion were often only slightly higher than those in the metaliion. Vertical profiles of TN were highly variable with either no difference in the metalimnion or a decrease in TN at the metalimnion. Hypolimnetic values again paralleled the high values observed in the epilimnion and metalimnion. Similar vertical total nutrient graphs are available for the other three lakes in Appendix 1.

Chlorophyll

Epilimnetic chlorophyll *a* concentrations during spring overturn (May-June) were considerably lower in 1994 than in 1993 (Figure 10), and than in 1992 (Budy et al. 1993). In 1994, spring concentrations for most of the lakes were near 0.5 µg/L, whereas in 1992 and 1993 they were generally above 0.75 and 1.0 µg/L, respectively. As in 1992, chlorophyll concentrations in 1993 decreased throughout most of the summer, and increased only slightly in the late fall. This contrasts with pattern in 1993 when epilimnetic chlorophyll increased markedly in the fall. In part this may be due to the fact that the deep chlorophyll maxima were less-well developed in 1994 (Figure 11; Appendix 4) than in 1993 (Gross et al. 1994), and when mixing occurred, there was less chlorophyll to be mixed into the surface waters (Fee 1976). Stanley Lake had the highest chlorophyll concentrations in fall and early winter in 1994 but did not have a pronounced spring. bloom as observed in 1993.

Phytoplankton

Epilimnetic phytoplankton biovolume and species composition in Redfish Lake has varied substantially between years (Figure 12). Biovolumes were very low in the drought year of 1992, and dominated by diatoms, chrysophytes, and chlorophytes. Biovolumes in 1993 were markedly high than in 1992 and Dinophytes became a dominant group in the epilimnetic phytoplankton. The total phytoplankton biovolumes in 1994 were intermediate between the low values in I992 and the high values in 1993.

In Redfish Lake the biovolume of phytoplankton increased in the upper metalimnion (15 m), and still more near the bottom of the photic zone (compensation point, 24-40 m) (Figure 13). The phytoplankton in the deep chlorophyll layer were dominated by diatoms and dinophytes. A substantial bloom in diiophytes occurred in the epilimnion in early June, and gradually shifted to deeper layers through July, creating the deep chlorophyll maxima noted in other years. The chrysophytes were a more significant member of the species composition in the 24-40 m samples than in the dinophyte-dominated 15- m samples. Redfish lake phytoplankton biovolumes by taxa and by

depth strata at each date are also shown in Appendix 5.

A comparison of chlorophyll *a* concentrations and phytoplankton biovolumes along a depth profile in Redfish Lake on 30 Aug 1994, indicated a strong correspondence between the two parameters (Figure 14). The highest chlorophyll a concentration and the highest biovolumes were observed in the metalimnion at 35-40 m. A strong (R^2 = 0.898) relationship between chlorophyll a concentration and phytoplankton biovolume suggests that chlorophyll serves as a good surrogate for the direct measure of algal abundance.

Zooplankton

Epilimnetic crustacean zooplankton dynamics in the Sawtooth1 Valley Lakes tied both seasonally and annually as well as among lakes (Figure 15). Zooplankton populations for all of the lakes was reduced in 1994 compared to 1992 and 1993. In general, Pettit and Stanley Lakes had the highest zooplankton biomass per liter, while Redfish Lake was intermediate, and Alturas Lake had the lowest biomass.

In all of the lakes except Alturas, zooplankton biomass was much lower during the winter than in the summer (Figure 16). Zooplankton biomass in Alturas Lake, on the other hand, increased after the fall, although zooplankton biomass tended to be highly variable throughout the year.

Similar patterns to the biomass data were observed in whole-lake total crustacean densities (Figure 17). Crustacean densities in all of the lakes were ahnost always less than 10/L. Pettit and Stanley had the highest densities of total zooplankton, whereas Redfish and Alturas Lakes usually had summer densities less than 3/L. Zooplankton densities in Alturas Lake remained low throughout the year with a slight increase in January and February.

Consideration of areal zooplankton biomass (g/m^2) allows us to compare the total amount of plankton in the lake, regardless of the lake's depth (Figure 18). Whereas biomass expressed in volumetric units gives us an idea of the biomass of plankton available to a single foraging fish the depth-integrated areal biomass should be a better measure of the total amount of plankton in the lake, and hence it's abiity to support a 0. nerka population. When expressed in areal units, the biomass of zooplankton in Pettit Lake is considerably higher than in the other lakes. The areal biomass in Alturas Lake was higher than the other lakes in the winter, but considerably lower in the summer. Redfish Lake, which had low *volumetric* biomasses, nevertheless had *areal* biomasses similar to Stanley Lake during the summer, and even higher biomasses than Stanley during the winter. For Redfish Lake the difference between the areal and volumetric expressions results because the plankton in this lake is spread out over a 90-m water column, whereas in the other lakes the zooplankton (and phytoplankton) are more compressed.

Seasonal zooplankton species composition was quite variable among lakes in 1994 (Figure 19). The wntribution of the major zooplankton taxa in Redfish Lake was near equal until June (Figure 19A), when *Daphnia* became dominant in addition to increased biomass of *Holopedium*. Alturas lake, however, was completely dominated

by ***Bosmina*** for the entire year (Figure **19B).** Pettit lake was dominated by both ***Daphnia*** and cyclopoids for most of the year with a slight increase in calanoids over the summer months (Figure 19C). Similar to Pettit Lake, Stanley Lake was dominated by ***Daphnia*** and cyclopoids, but calanoids and ***Holopedium*** were also composed a major portion of the summer biomass **(Figure** 19, D).

Cladoceran egg ratios were variable for 1994 among the four lakes (Figure **20),** in part because the low densities of mature animals was low, and we were unable to make precise estimates of fecundity. Overall, the egg ratios were very low, generally less than 1 egg/female. In **Redfish** Lake, the egg ratios of ***Bosmina*** were generally highest during spring overturn and the period of highest chlorophyll **(cf. 10),** and lower in the late fall and winter. ***Hoiopedium egg*** ratios also peaked in mid-summer. Alturas egg ratios were low all year with the exception of isolated peaks of ***Daphnia*** and ***Holopedium*** in July. The higher production for Pettit Lake was apparent in the ***Daphnia*** egg ratios, but smaller peaks of ***Holopedium*** and ***Bosmina occurred in*** early June. In Stanley Lake ***Bosmina*** production was high for most of the year until a crash in mid-July corresponding with a dramatic increase in ***Daphnia*** production and ***Holopedium*** production to a lesser degree.

The size-frequency distributions of crustacean zooplankton diiered markedly between lakes (Figure 21). In Pettit and Stanley Lakes, large **zooplankton** were moderately abundant in both winter and summer. In contrast, both **Redfish,** and particularly Alturas Lake were dominated by small species of zooplankton. In **Alturas** Lake, the zooplankton are not only scarce (Figure **17),** but extremely small.

Discussion

The warm dry summer of 1994 was similar to 1992 in lake water temperatures, but seasonal patterns of phytoplankton and zooplankton abundances were generally intermediate between 1992 and 1993 observations. Chlorophyll concentrations and phytoplankton biomass were higher in 1993 and 1994 compared to 1992. Mean summer zooplankton biomass, however was similar among the years. Zooplankton biomass in **Redfish** Lake remained higher during the autumn of 1994 compared to previous years.

Although phytoplankton biomass as measured in terms of chlorophyll concentration was closely related to nutrient loadiig in the four lakes (Steinhart et al. **1994),** these bottom-up effects did not explain the pattern of zooplankton biomass in the lakes (Figure 10). The relationship of mean TP concentrations to zooplankton biomass among lakes was weak in each of the years. If anything, the relationship of TP to zooplankton biomass was negative. Low levels of phosphorus produced high zooplankton biomass in Pettit Lake in all years. (Figure 13). Conversely, high concentrations of TP were associated with extremely low **biomasses** of zooplankton in **Alturas** Lake in 1994.

Top-down effects related to variation in planktivorous fish density may explain this pattern of zooplankton biomass. High densities of fish estimated in Alturas Lake in 1991-1993 could have exerted sufficient

predation pressure on zooplankton populations to reduce them to very low levels (Steinhart et al. 1994). In Pettit Lake, the low density of planktivorous fish estimated in 1992-1994 may have allowed crustacean zooplankton to thrive under low predation conditions. Results of these comparisons indicated that top-down effects of fish predation likely exerted more control over zooplankton biomass than did bottom-up effects of food resources.

Comparison of the limnological characteristics of the four lakes from 1992-1994 indicated that the trophic structure of each lake has remained relatively stable. Pettit and Redfish Lake had the lowest, Alturas was intermediate, and Stanley had the highest phytoplankton abundances in each year. The ranking of summer zooplankton biomass among years was also consistent, with Alturas, Redfish, Stanley and Pettit exhibiting lowest to highest volumetric zooplankton biomass each year. The consistency of these ranking suggests that inter-annual variation in climate and fish recruitment processes had less effect on food web function than did inter-lake differences in species composition and lake physics. This conclusion implies that lake management strategies for enhancing sockeye salmon production should vary among the lakes, but that only minor adjustments in the management of a given lake would have to be made on an annual basis.

The limnological information collected from 19921994 indicates that a variety of lake management approaches could be tried to enhance production of juvenile sockeye salmon. The low concentrations of phytoplankton in Redfish and Pettit lakes make them candidates for lake fertilization. The low abundances of zooplankton in Redfish and Alturas lake suggests that efforts to reduce planktivorous competitors with juvenile sockeye salmon would be beneficial. The program of lake fertilization and kokanee reduction planned for Redfish Lake in 1995 will allow us to evaluate the effectiveness of these techniques at providing appropriate conditions for growth and survival of juvenile sockeye salmon.

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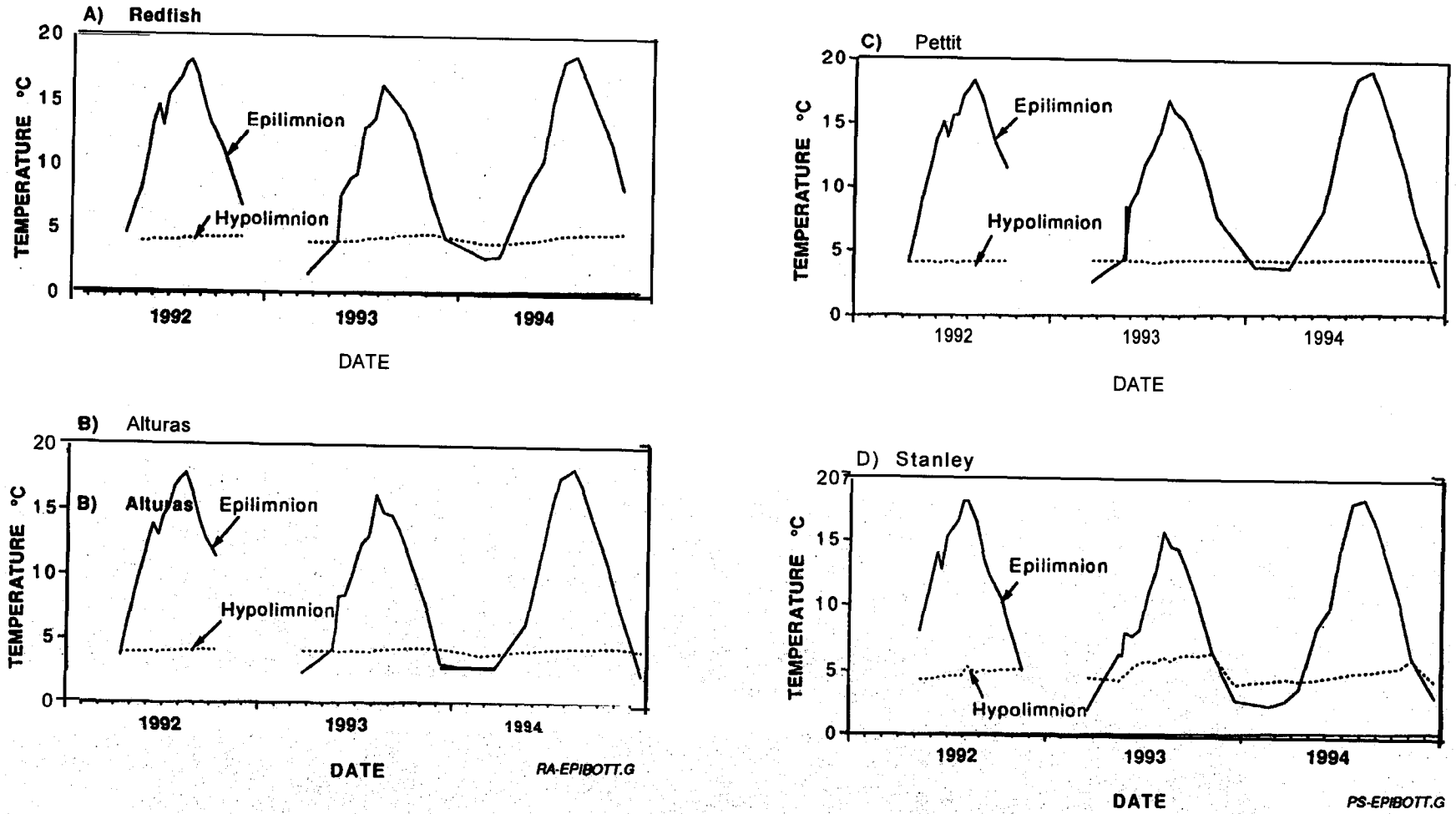


Figure 1. Epilimnetic (2 m) and hypolimnetic temperatures for the Sawtooth Valley Lakes from 1992-1993. A) Redfish B) Alturas C) Pettit D) Stanley.

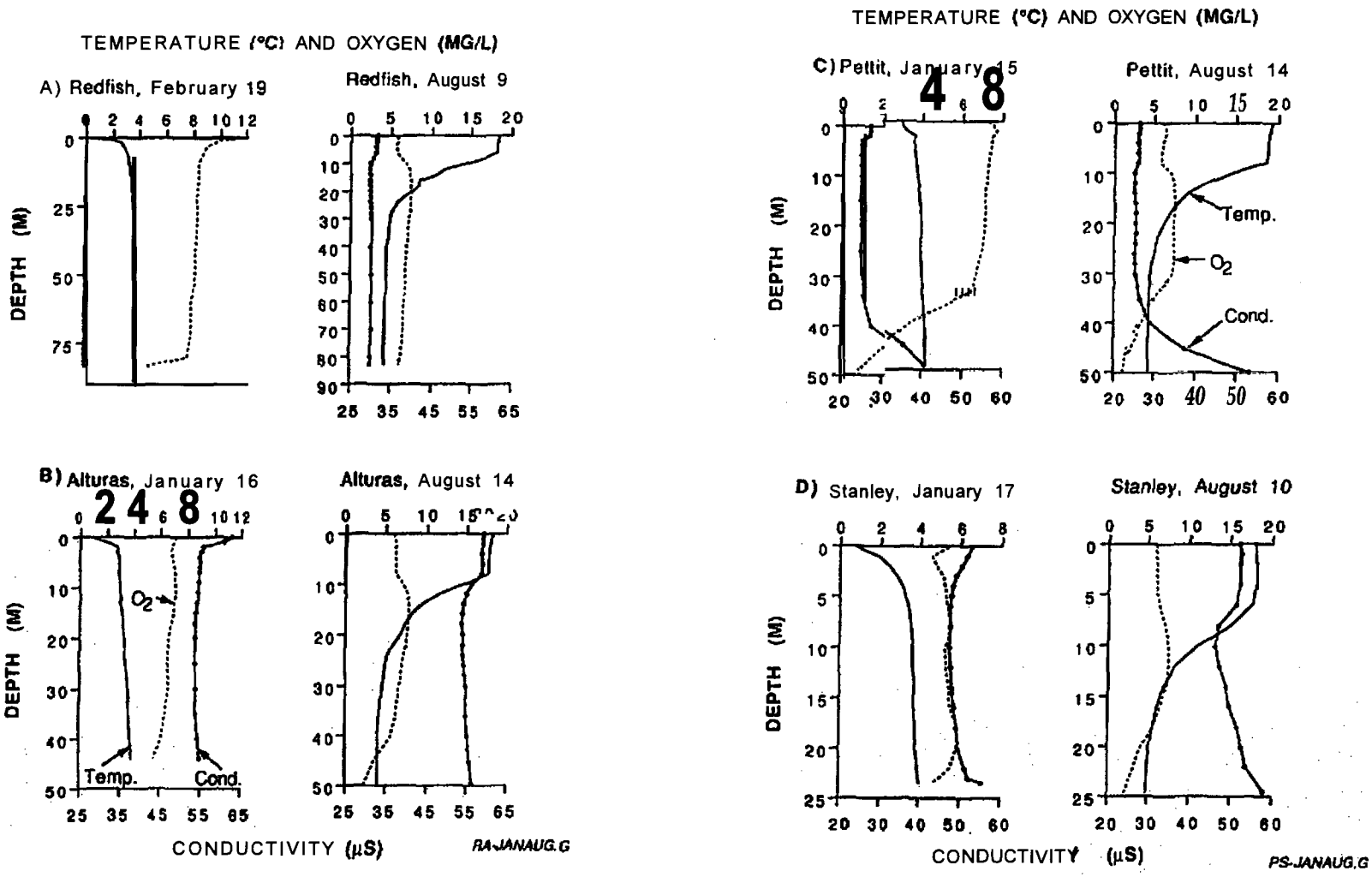


Figure 2. Temperature, oxygen, and conductivity profiles for the Sawtooth Valley Lakes during ice-cover in January and during mid-summer in August. A) Redfish B) Alturas C) Pettit D) Stanley.

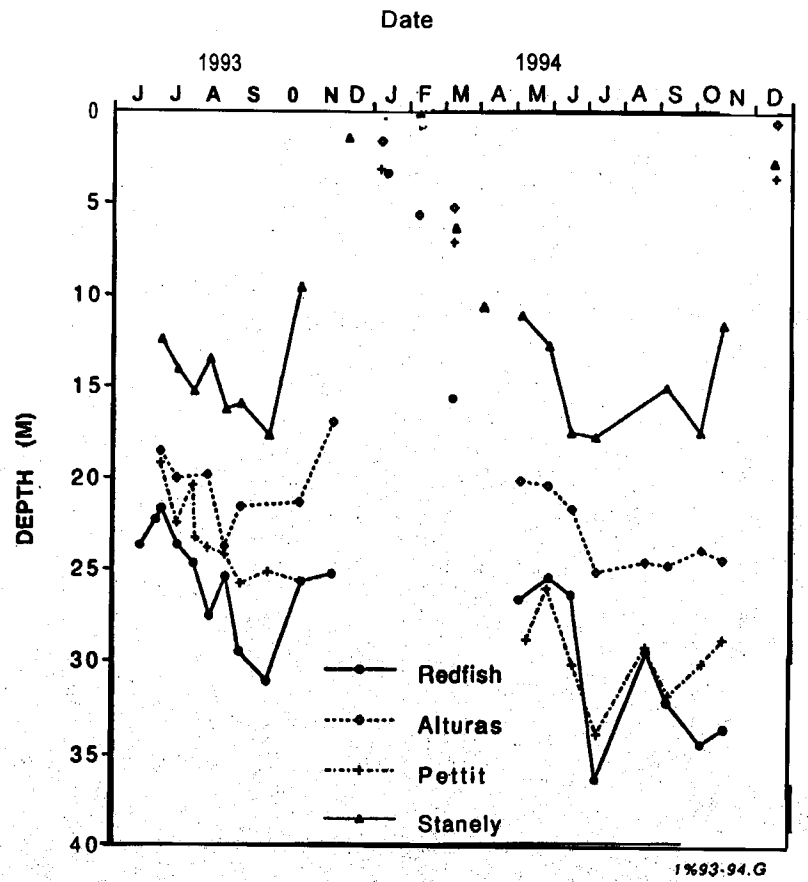


Figure 3. Depth of the compensation point (1% surface light intensity) in the Sawtooth Valley Lakes in 1993-1994. The points not connected by the line were measured under-ice.

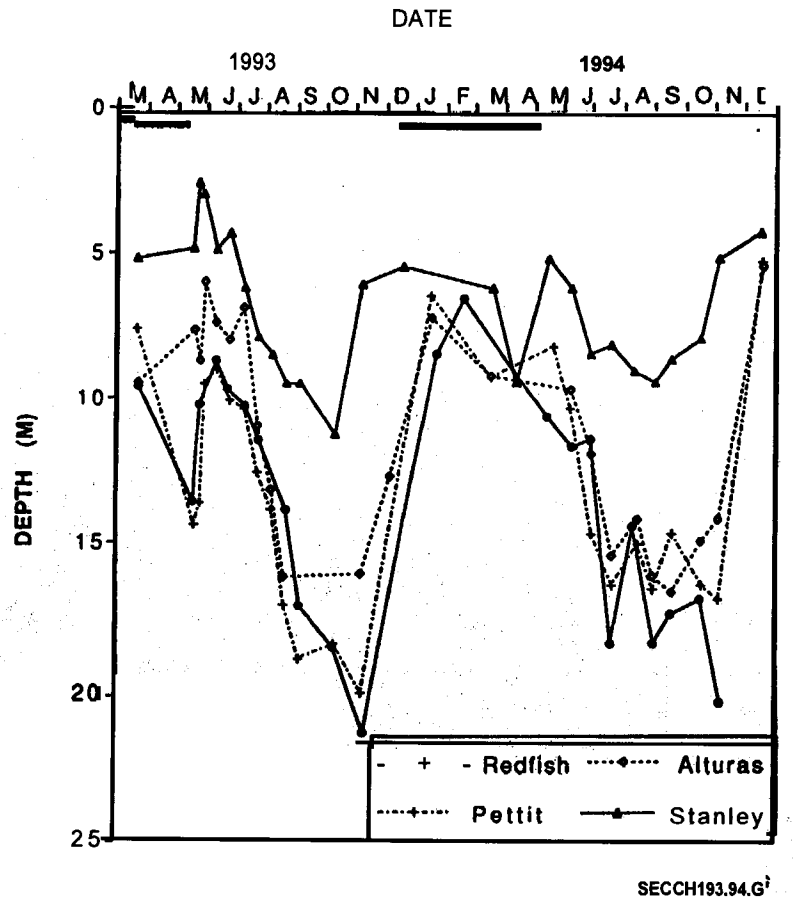


Figure 4. aécchi depths for the Sawtooth Valley Lakes in 1993-1994. The bars denote Periods of ice-cover orthe lakes.

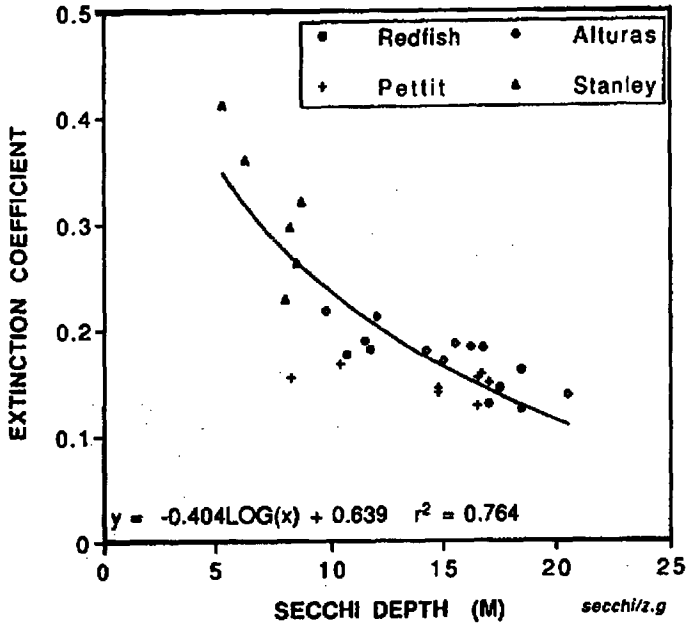


Figure 5. Relationship between Secchi depth and extinction coefficient for the Sawtooth Valley Lakes in 1994. The regression is calculated using points from all four lakes.

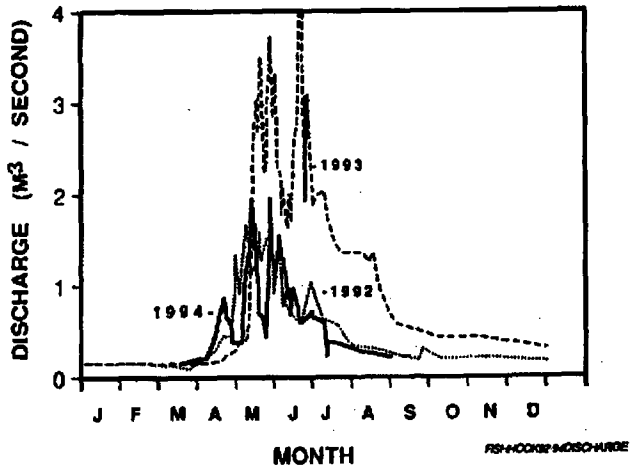


Figure 6. Hydrographs for Fishhook Creek in 1992, 1993, and 1994. Discharge and gauge readings were taken approximately 100, and 300 m above the lake respectively.

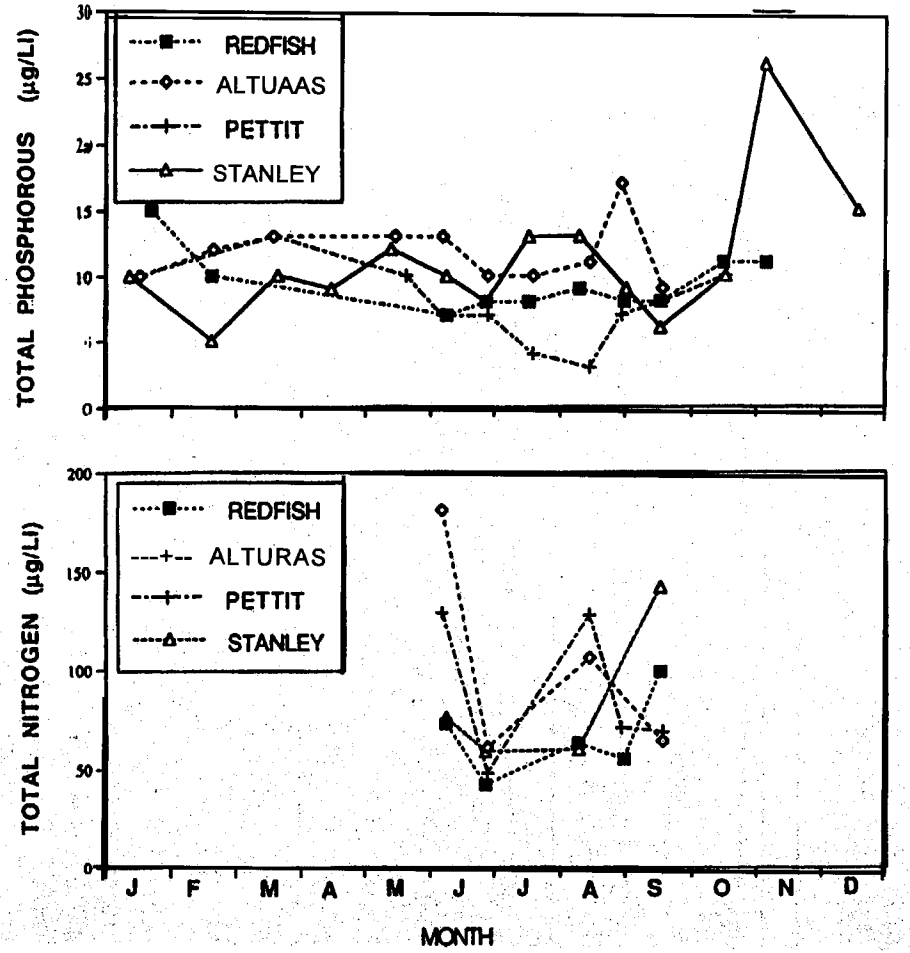


Figure 7. Epilimnetic (6-O m) total phosphorous (µg/l) [top] and total nitrogen (µg/l) [bottom] in all four lakes measured from 6-O m tube samples for 1994 data.

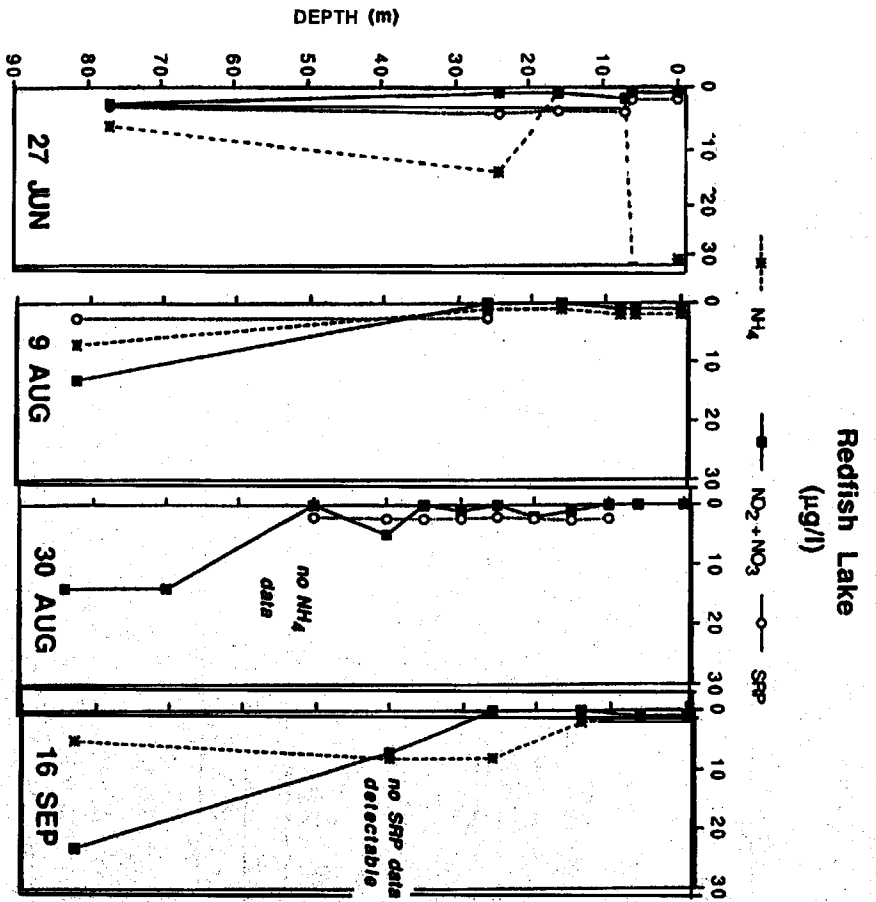


Figure 8. Total phosphorous (TP) and total nitrogen (TN) (µg/l) profiles for Redfish Lake, 1994 (TP and TN profiles for the other three lakes are provided in Appendix 1). On 17 Jul and 9 Aug only TKN data was available for calculating TN (TN = TKN + [NO, + NO,]).

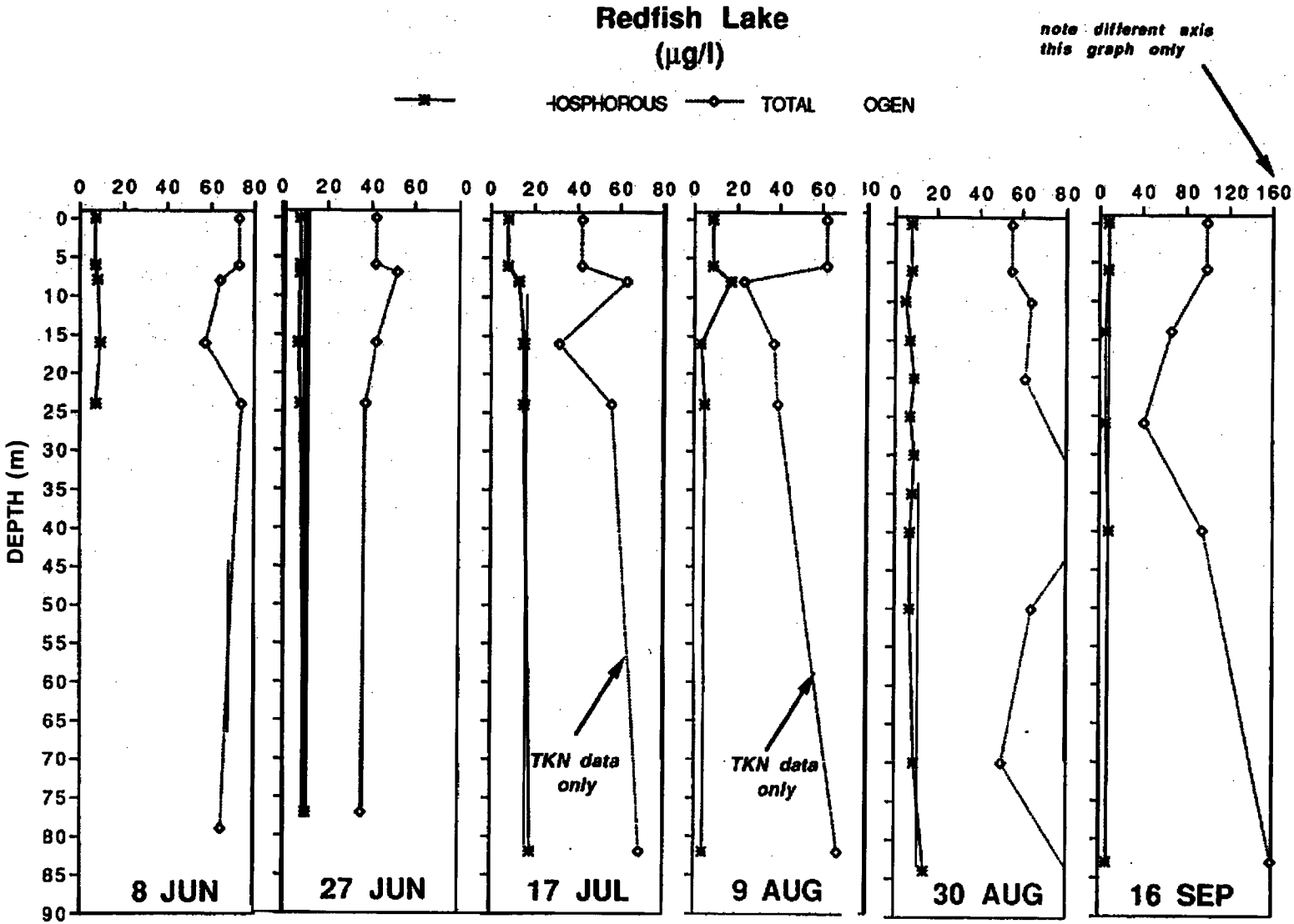


Figure 9. Soluble nutrient profiles for Redfish Lake, 1994 (soluble nutrient profiles are provided in Appendix 2) . Missing data points for SRP represent values less than the minimum detection limit for SRP data = 2 µg/l.

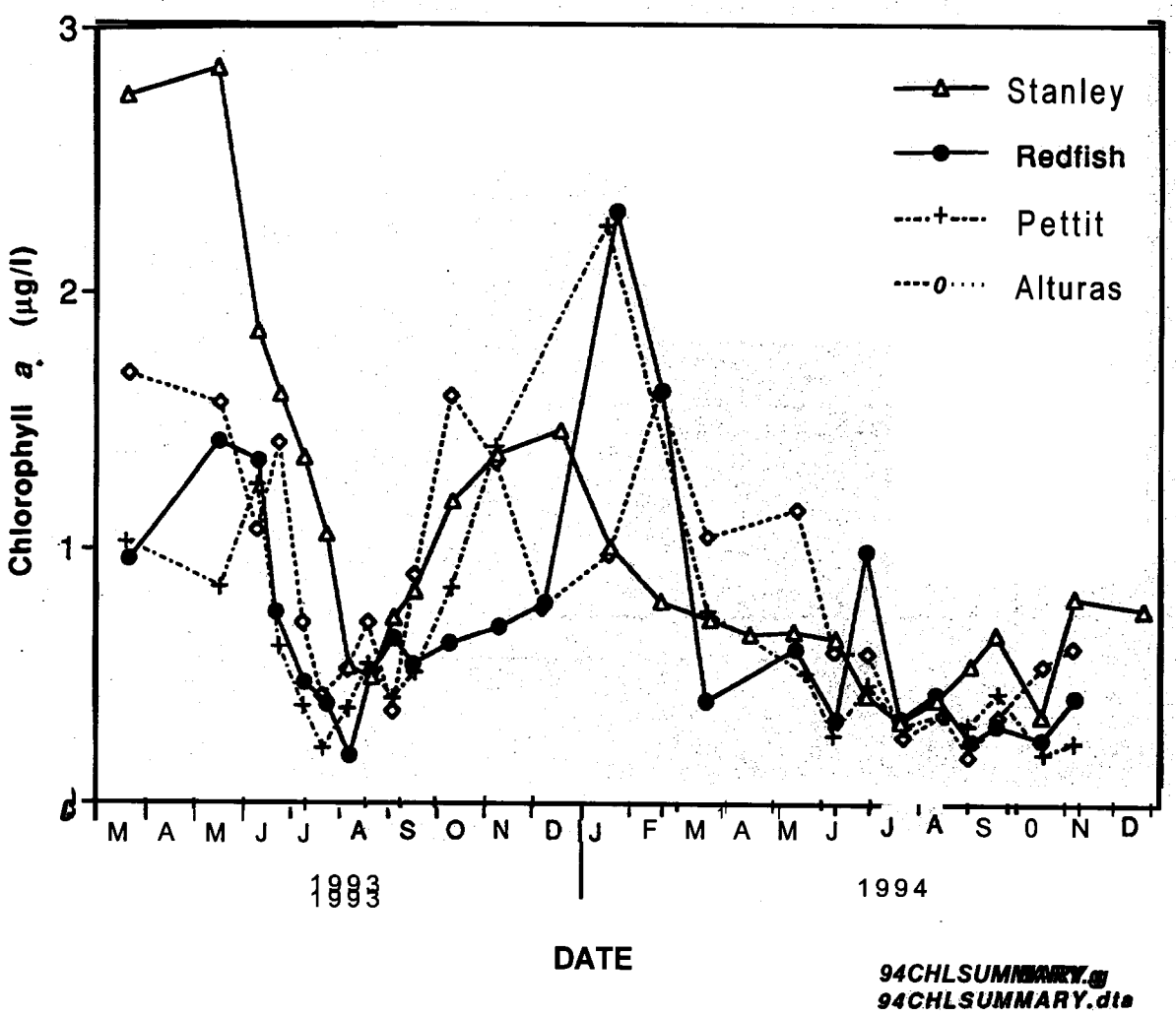


Figure 10. Epilimnetic lake chlorophyll *a* (µg/l) data from 6-O m tube samples, March of 1993 through December of 1994.

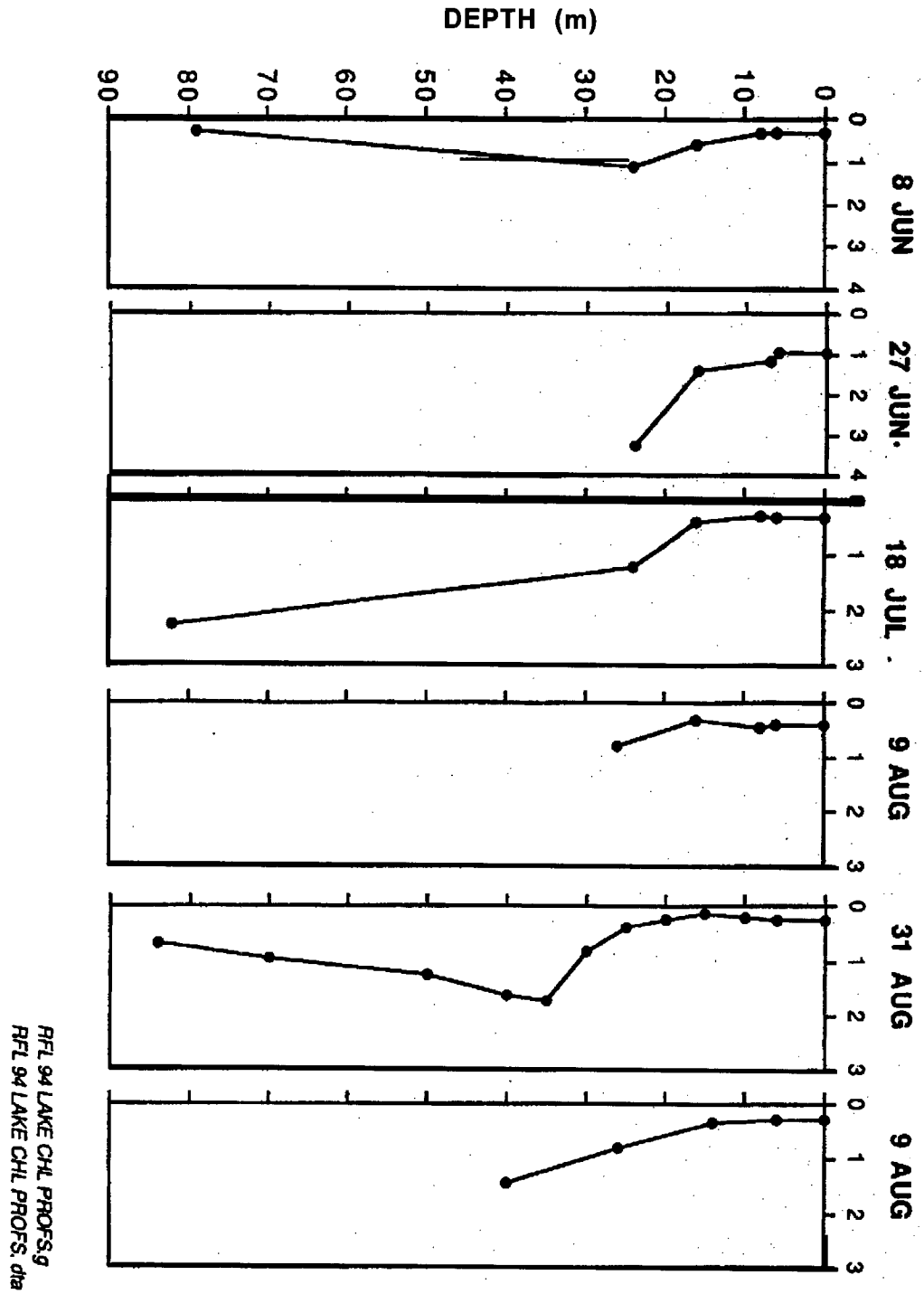


Figure 11. Chlorophyll *a* (*µg/l*) profiles for. Redfish Lake, 1994 (profiles for the other three lakes, Alturas, Pettit, and Stanley) are provided in Appendix 4.

Figure 12. Epilimnetic (6-O m) phytoplankton biovolumes by species for o-u m tube samples. Data are shown for 1992,1993, and 1994.

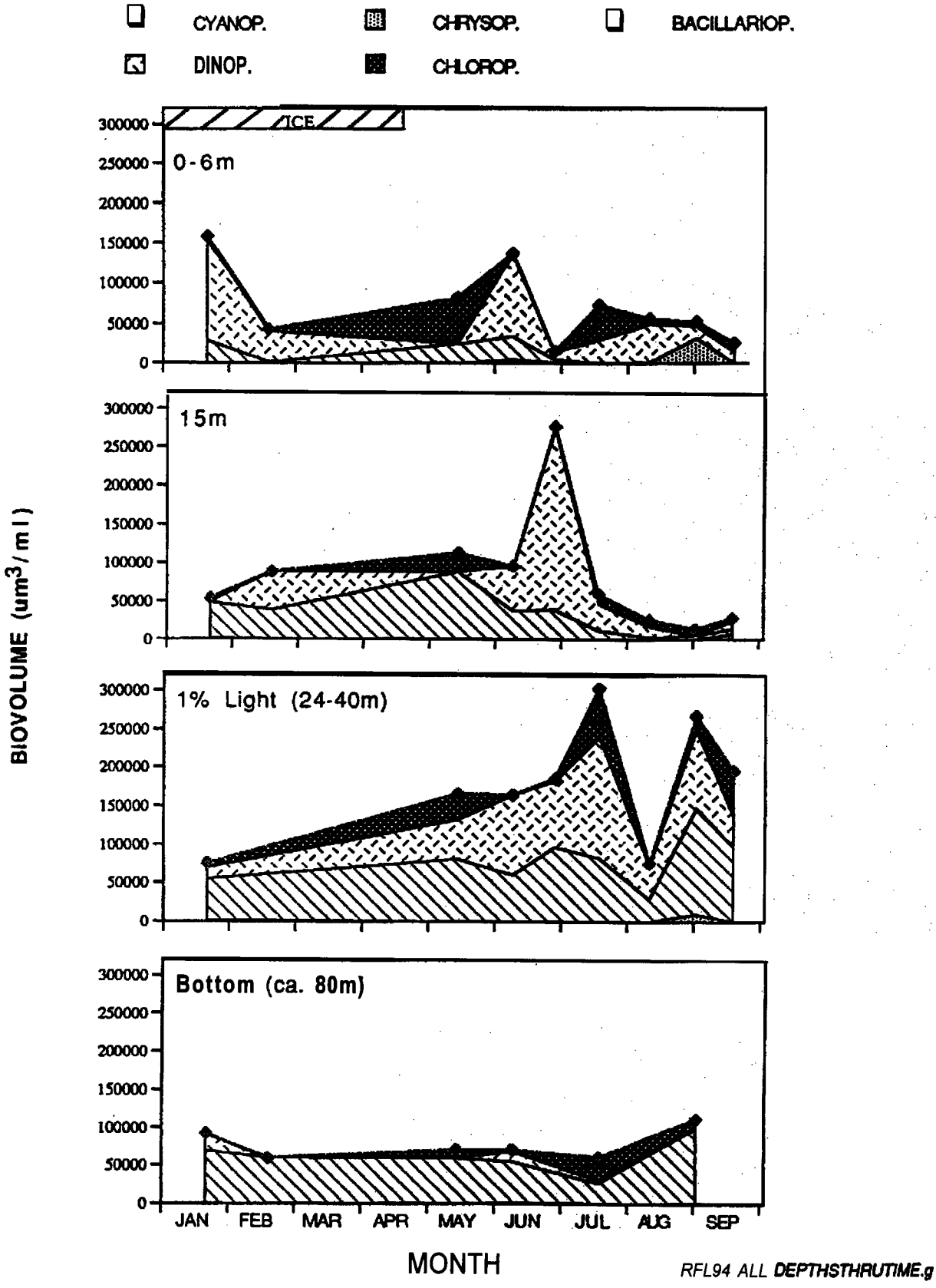


Figure 13. Phytoplankton biovolumes by species and by depth strata throughout the year for Redfish Lake, 1994.

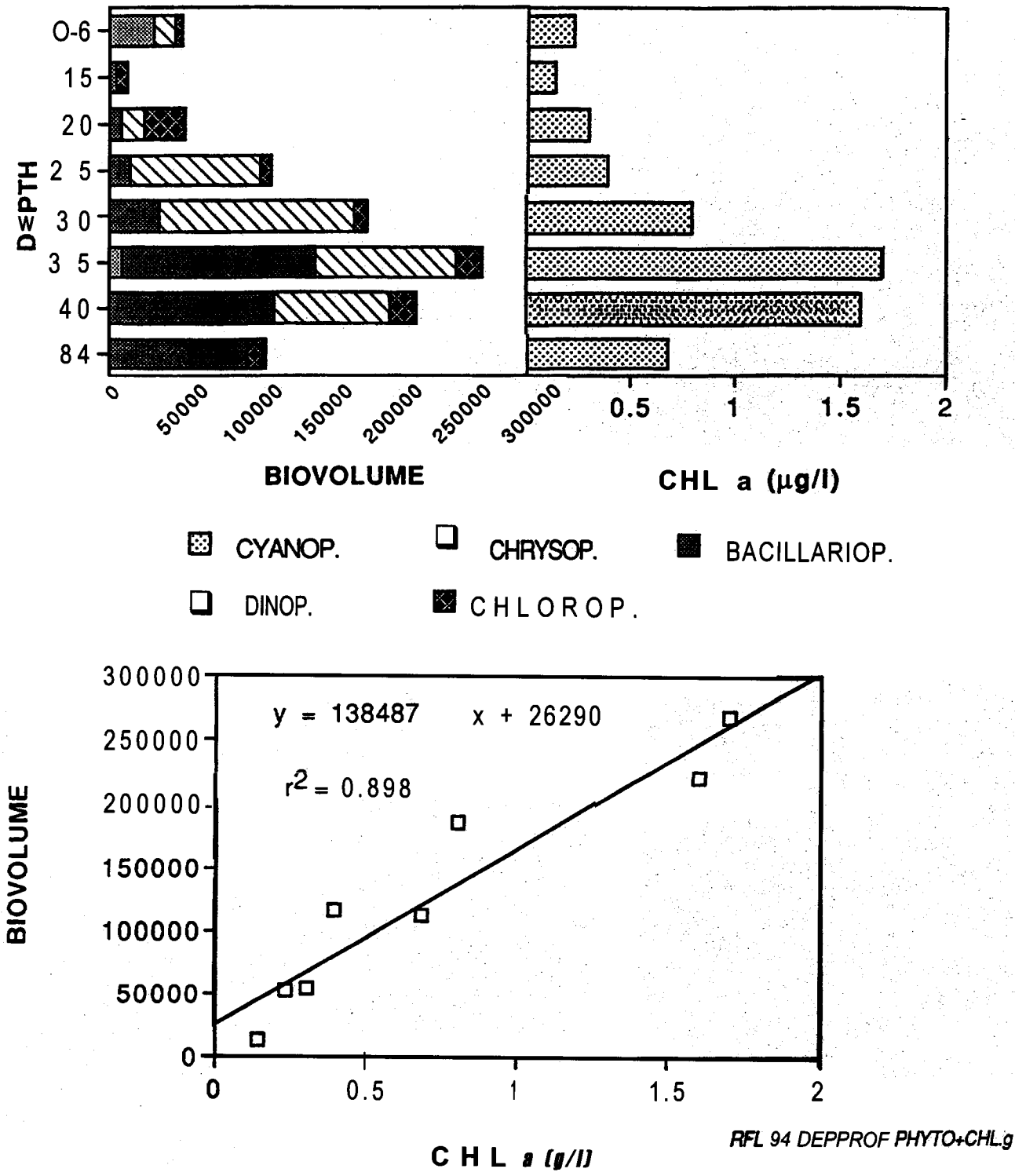


Figure 14. Depth profiles of chlorophyll *a* @g/l) and phytoplankton biovolumes data for Redfish Lake, 30 AUG 1994. The top graphs show the phytoplankton species composition by depth [left] and the chlorophyll concentration [right] by depth, and the bottom graph shows a regression of chlorophyll concentration and phytoplankton biovolume.

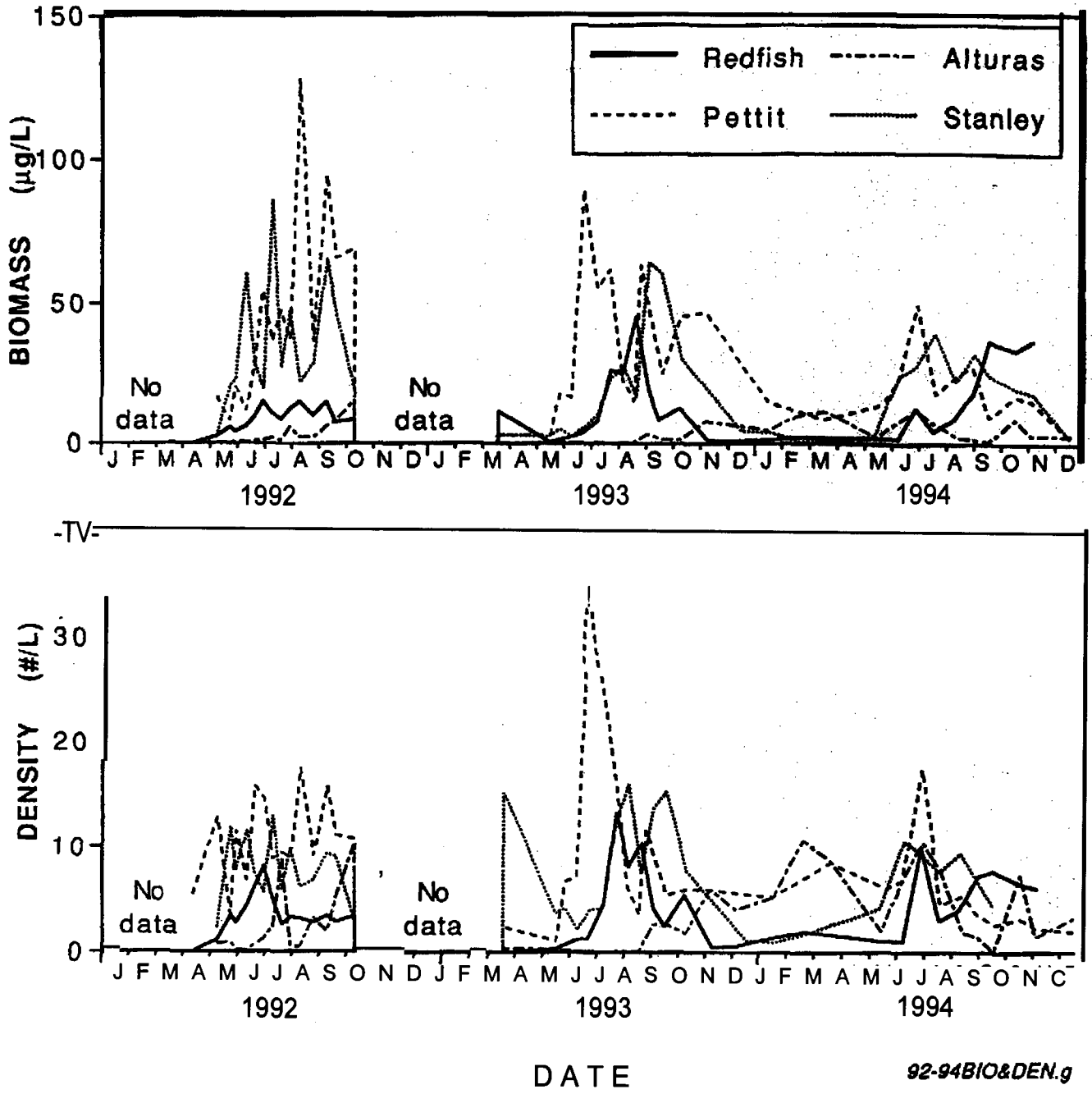


Figure 15. Crustacean rooplankton biomass [top] and density [bottom] for the epilimnion (0-10 m) of the Sawtooth Valley Lakes from 1992-1994.

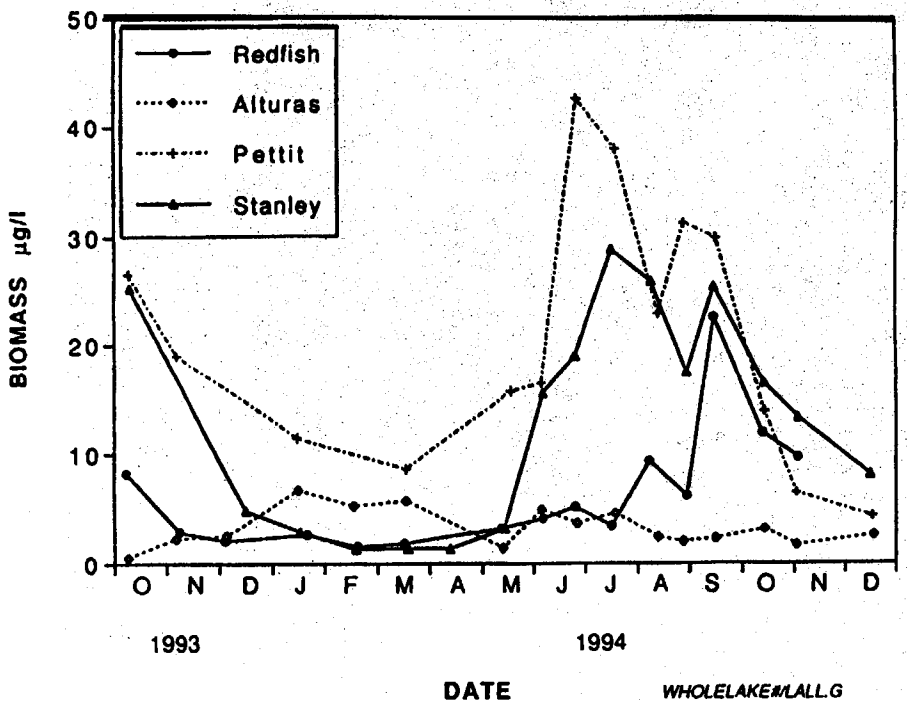


Figure 16. Whole-lake crustacean zooplankton biomass (µg/l) for the Sawtooth Valley Lakes in 1993-1994.

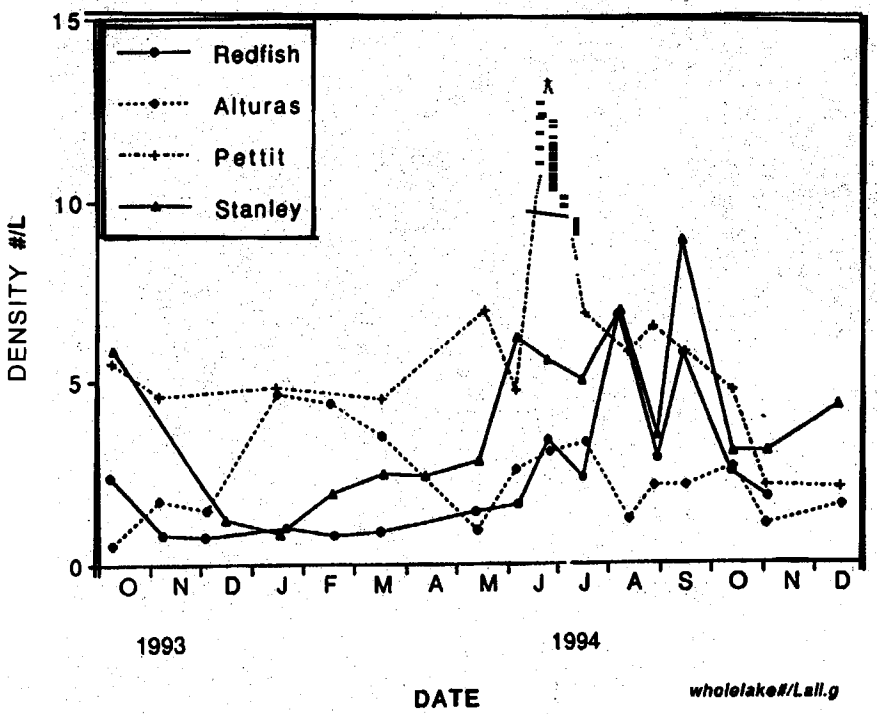


Figure 17. Whole-lake crustacean zooplankton density for the Sawtooth Valley Lakes in 1993-1994.

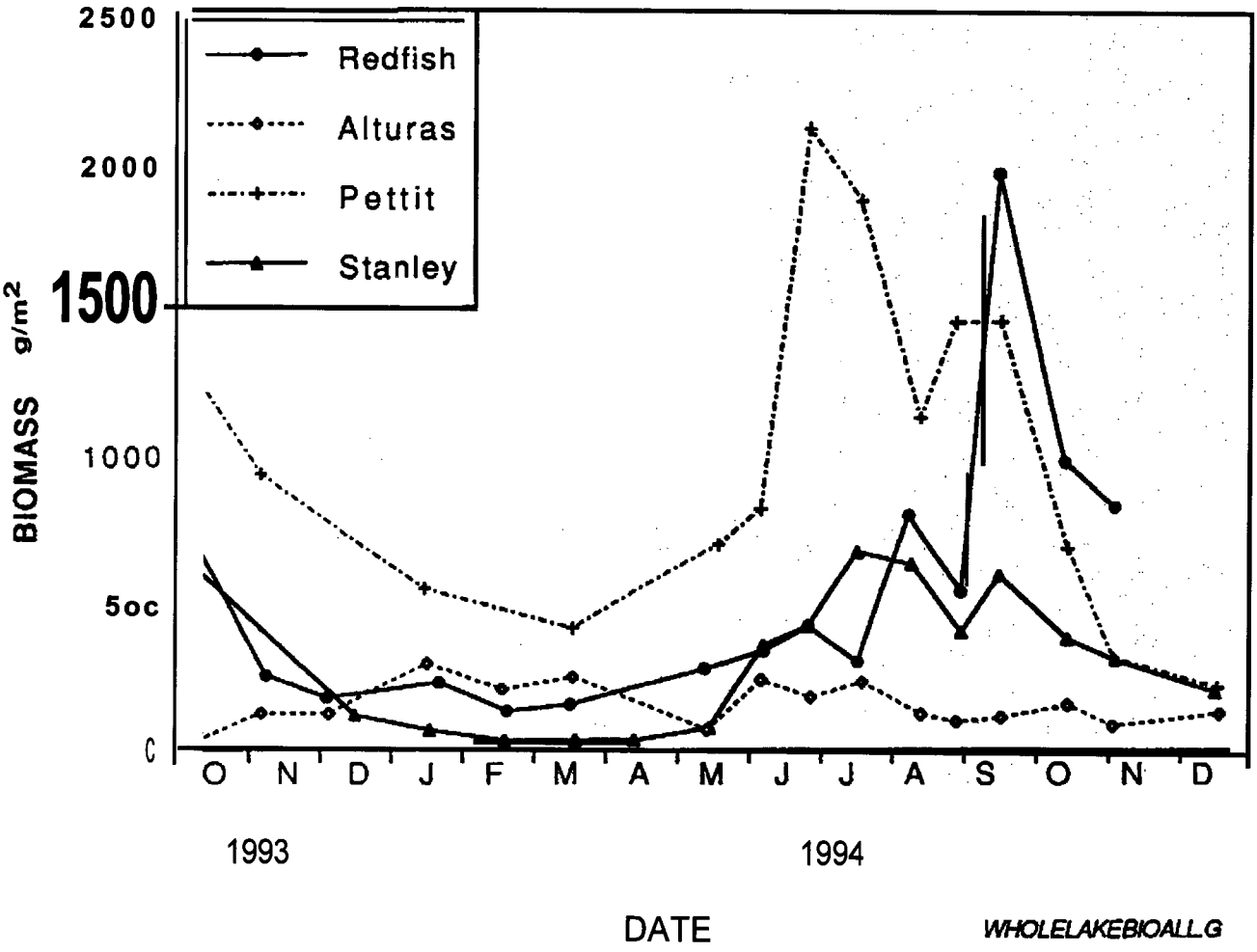


Figure 18. Areal whole-lake crustacean zooplankton biomass (g/m^2) for the Sawtooth Valley Lakes in 1993-1994.

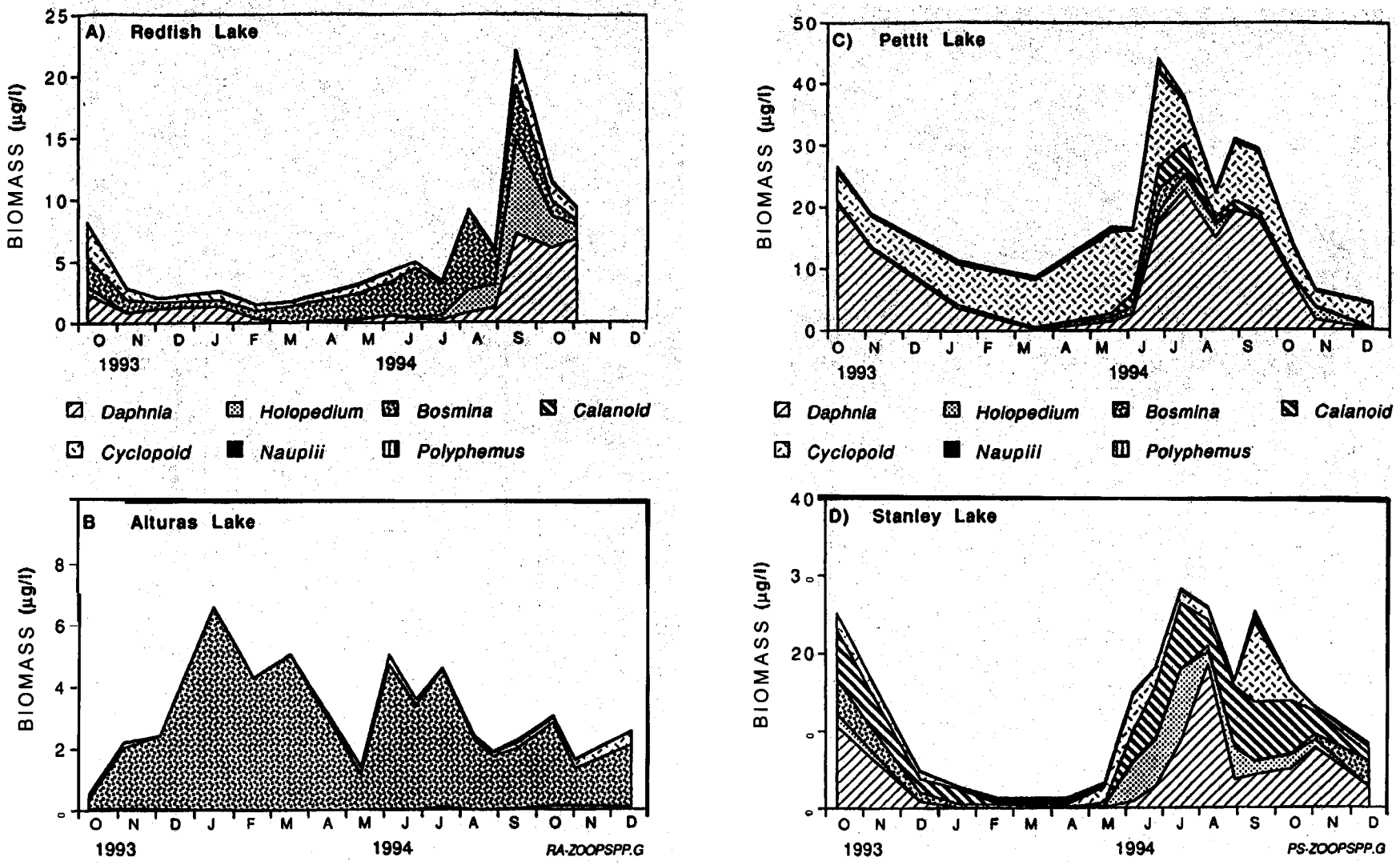


Figure 19. Seasonal changes in the whole-lake biomass of crustacean zooplankton taxa in the Sawtooth Valley Lakes in 1993-1994. A) Redfish B) Alturas C) Pettit D) Stanley. Note the difference in scale on the y-axes.

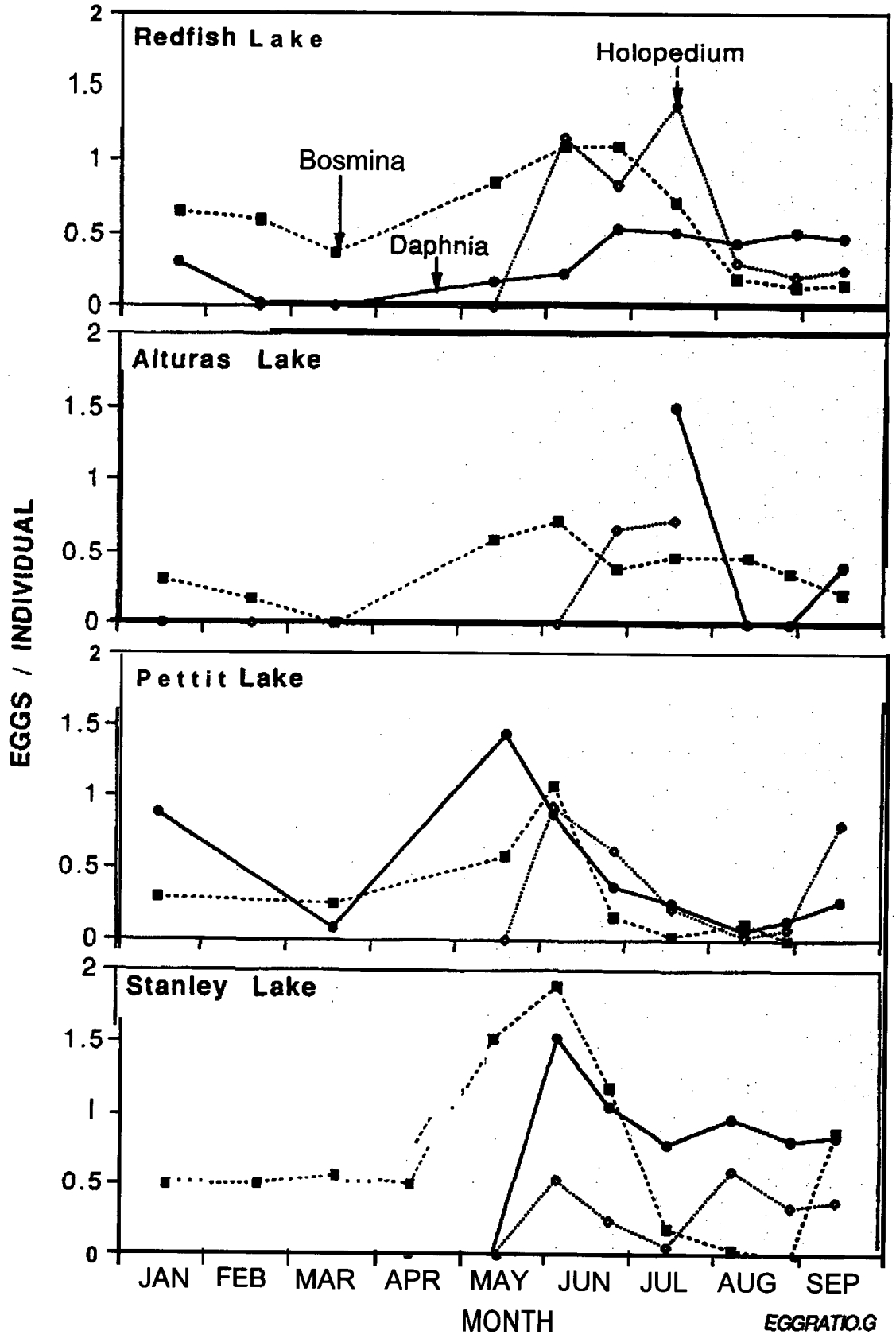
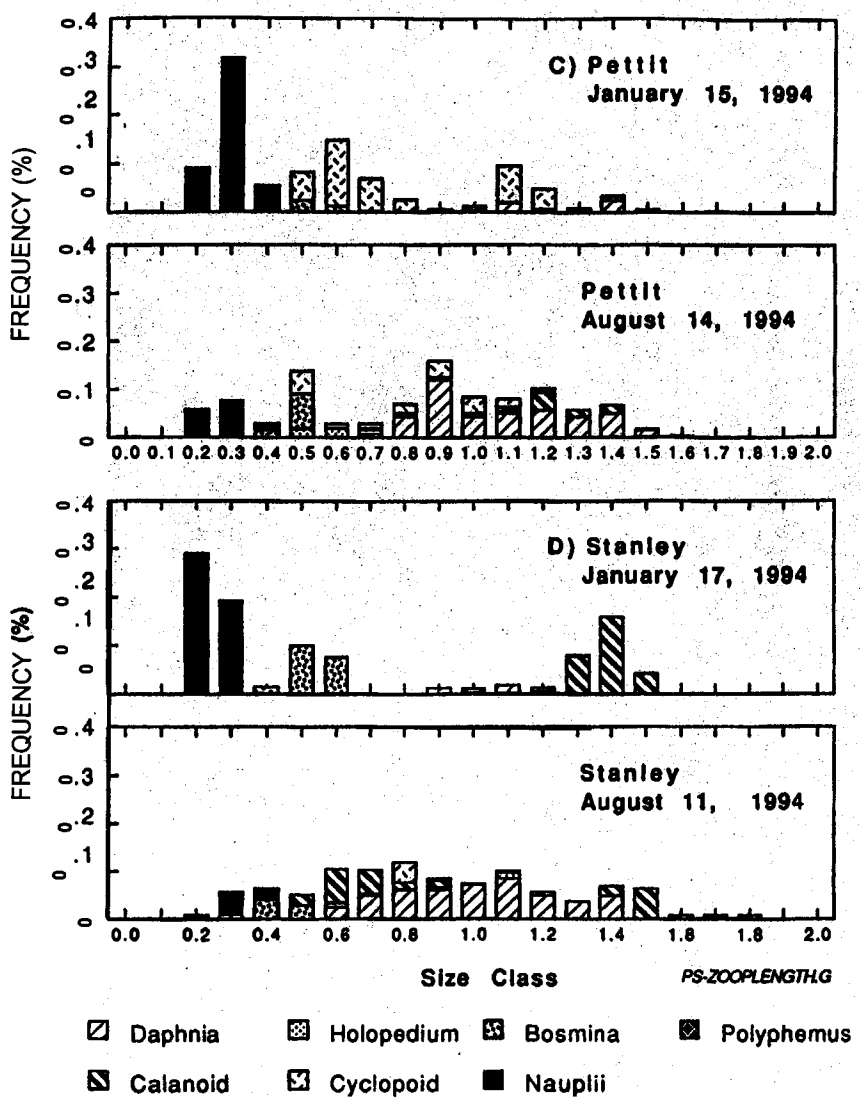


Figure 20. Seasonal changes in cladoceran zooplankton egg-ratios for the Sawtooth Valley Lakes in 1994.



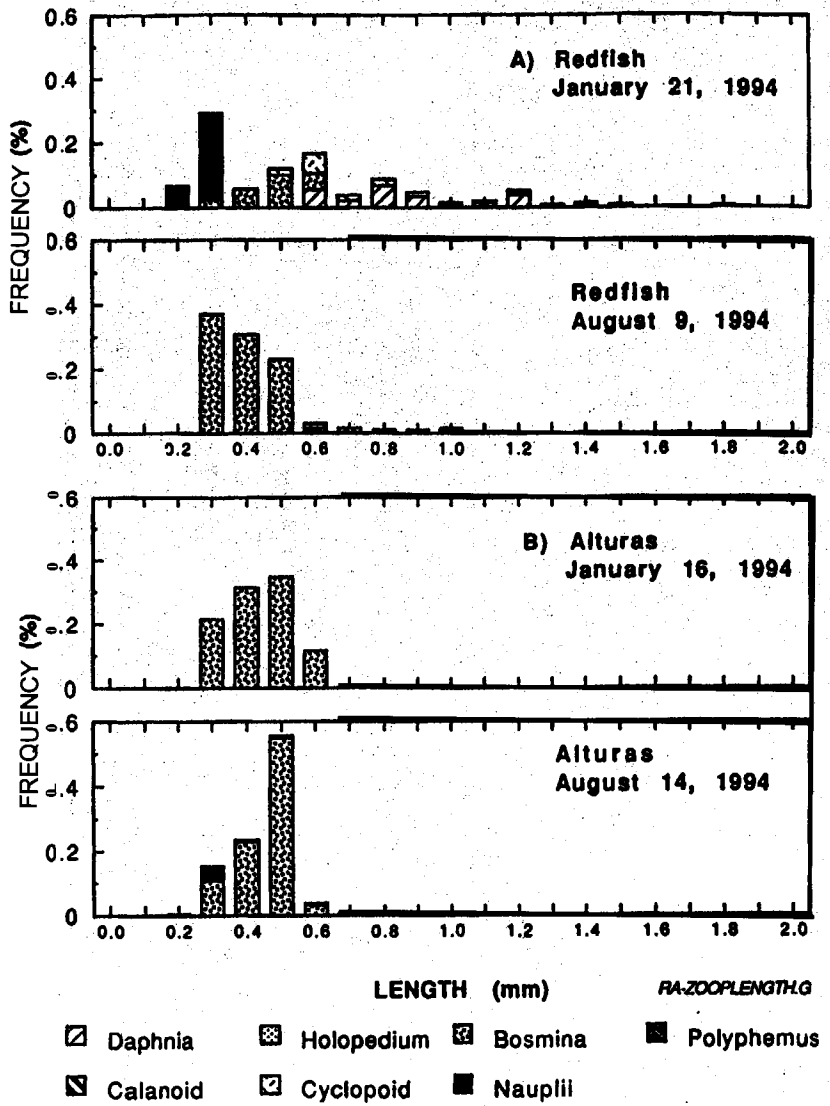


Figure 21. Length quency distributions of pla taxa un
January and during d-summer in August fo S: h Valle 1994.
A) Redfish B) Altu C) Pettit D) Stanely.

Table 1. Limnological characteristics of the Sawtooth Valley Lakes. Means are for the ice-free period, May-November, 1994.

Lake	Area (km2)	Depth (m)		Secchi Depth (m)		Light Extinction Coefficient	
		Maximum	Mean	Mean	Range	Mean	Range
Redfish	6.17	92	41	15.6	10.7-20.5	0.16	0.12-0.19
Alturas	3.38	53	33	14.2	9.8-16.8	0.19	0.17-0.22
Pettit	1.63	52	28	14.4	8.3-17	0.15	0.13-0.17
Stanley	0.73	26	16	7.6	5.3-9.5	0.32	0.23-0.41

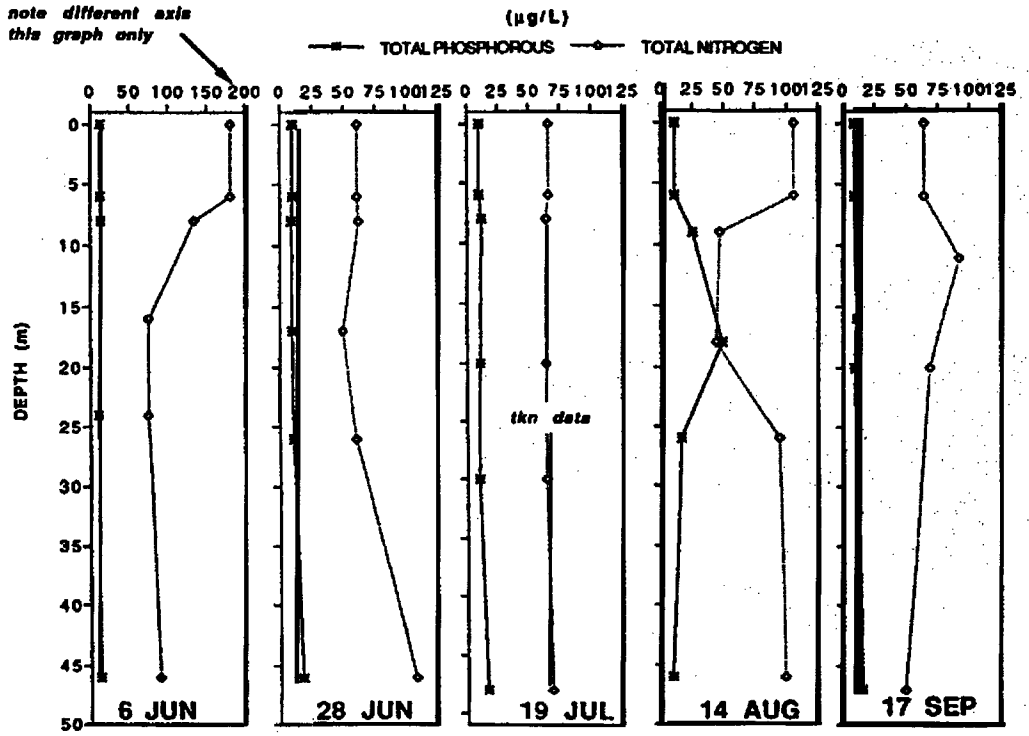
Table 2. Mean total phosphorus (TP) and total Kjeldahl nitrogen (TKN) in Fishhook and Redfish Creek for 1992, 1993, and 1994. The nitrogen to phosphorus ratio is also shown. For 1993 values in italics show the means when the extremely high concentrations measured on the rising limb of the spring hydrograph are included in the calculation. Mean ± (S.E.)

	TP	TN	N:P
1992	12.6 (1.0)	83 (7)	6.6
1993	11.0 12.2) 19.7 *(6.8)*	103 (11) 184 *(64)*	9.3 9.3
1994	10.3 (1.0)	81 (6)	7.8

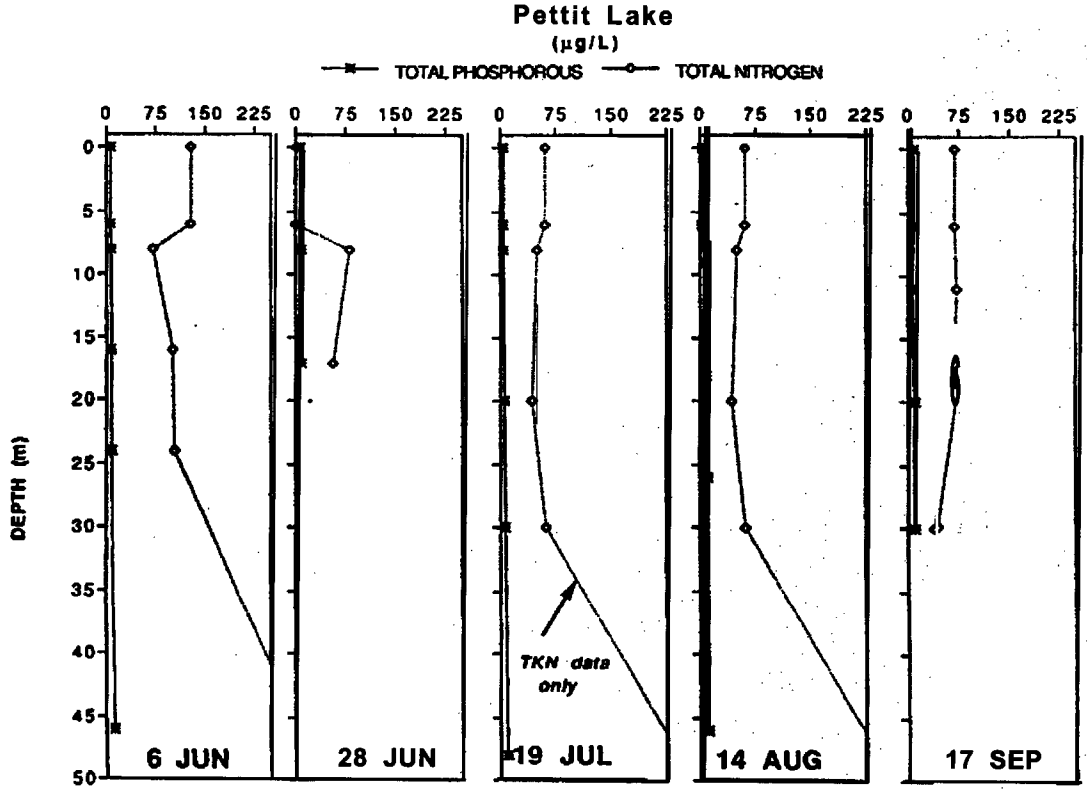
Appendix 1

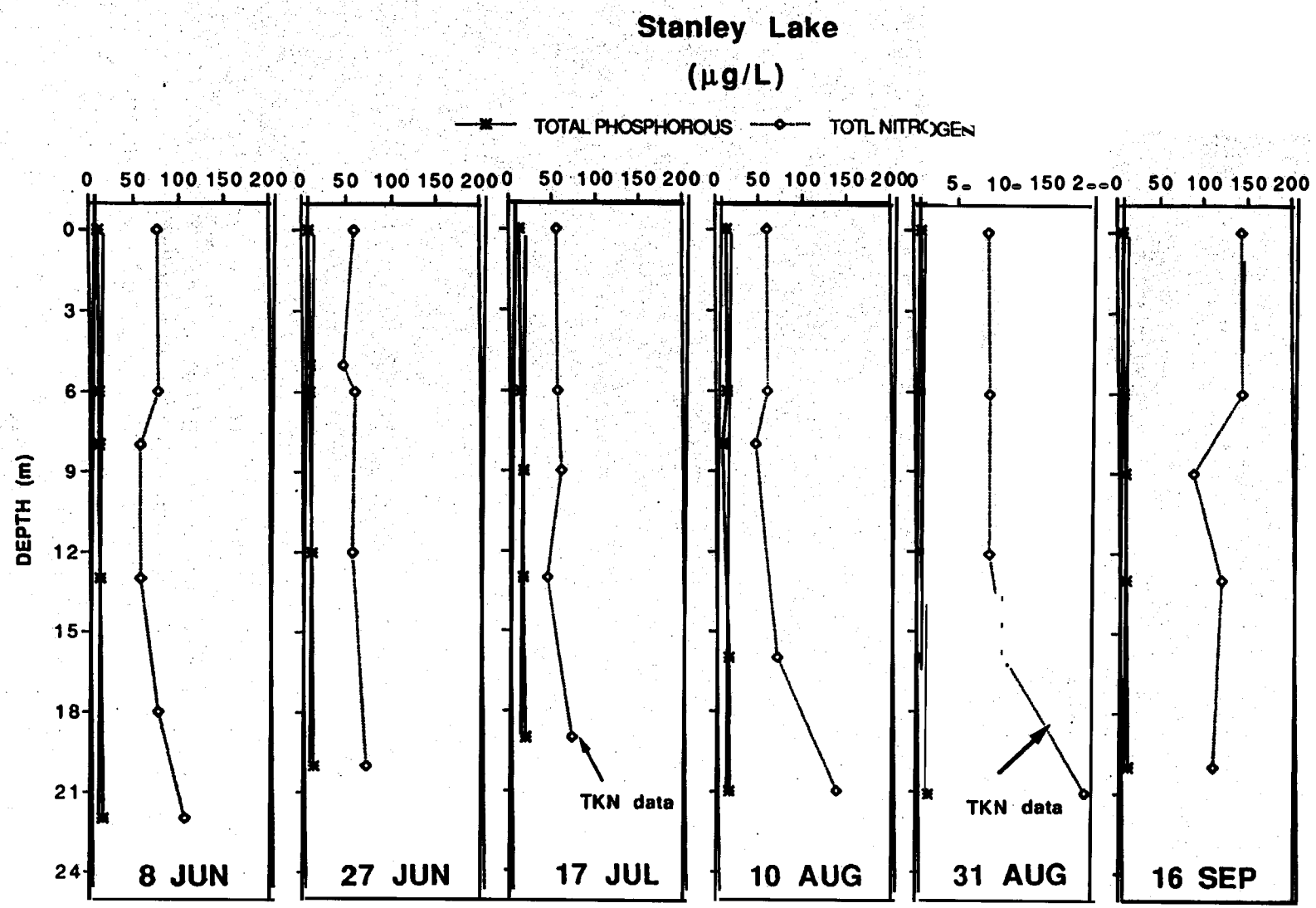
Appendix 1. Total phosphorous and nitrogen profiles for Alturas, **Pettit,** and Stanley Lakes.

Alturas Lake



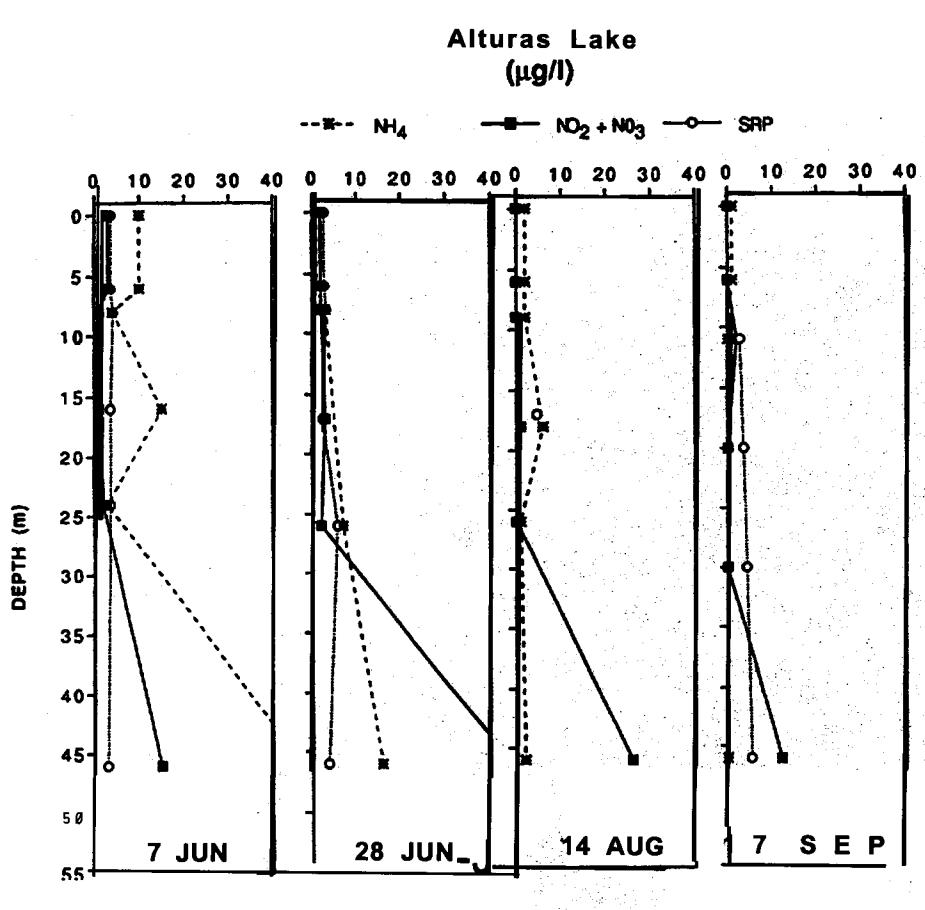
Pettit Lake

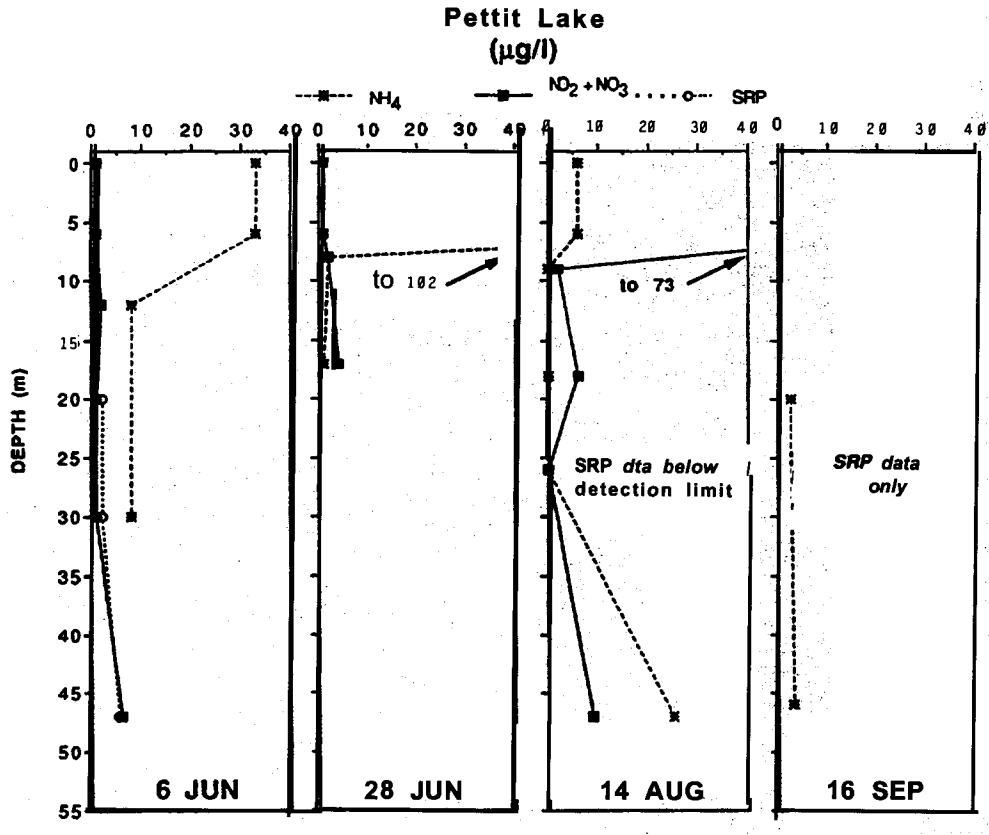


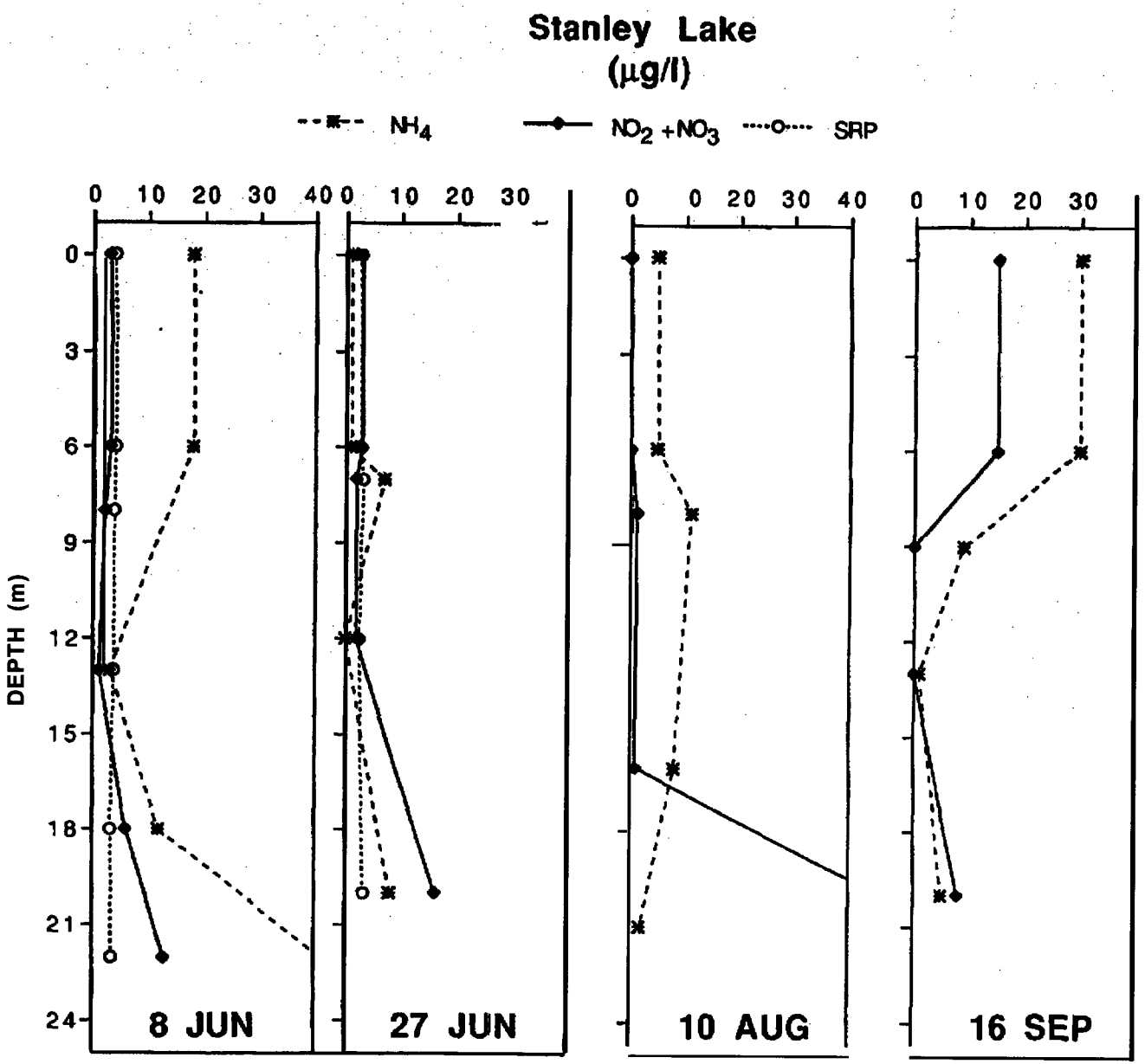

Stanley Lake
(µg/L)
TOTAL PHOSPHOROUS
TOTL NITROGEN
0 50 100 150 200
DEPTH (m)
0
3
6
9
12
15
18
21
24
TKN data
TKN data
8 JUN
27 JUN
17 JUL
10 AUG
31 AUG
16 SEP

Appendix 2

Appendix 2. a. Soluble nutrient profiles for Alturas, Pettit, and Stanley Lakes, 1994 (soluble nutrient profiles are provided in Appendix 2) . Missing data points for SRP represent values less than the minimum detection limit for SRP data = 2 µg/l.


Alturas Lake
(µg/l)
NH4
NO2 + NO3
SRP
DEPTH (m)
7 JUN
28 JUN
14 AUG
17 SEP


Pettit Lake
(µg/l)
NH4
NO2 + NO3
SRP
DEPTH (m)
to 102
to 73
SRP dta below detection limit
SRP data only
6 JUN
28 JUN
14 AUG
16 SEP


Stanley Lake
(µg/l)
NH4
NO2 +NO3
SRP
DEPTH (m)
0 10 20 30 40
0 10 20 30
0 0 20 30 40
0 10 20 30
0
3
6
9
12
15
18
21
24
8 JUN
27 JUN
10 AUG
16 SEP

Appendix 2. b. Spring overturn and mean seasonal epilimnetic nutrient concentrations for the Sawtooth Valley Lakes, 1992, 1993, and 1994. All values are in µg/l. Mean seasonal values for phosphorous include May-October, and mean seasonal values for nitrogen include June- September.

1994	Spring Overturn							
Lake	TP	TN	TP	SRP	NO3	TKN	TN	TN:TP
Redfish	6	64	9	<2	1	61	67	7.4
Alturas	13	81	12	<2	1	102	103	8.5
Pettit	5	101	7	<2	1	80	89	12.7
Stanley	12	91	12	<2	5	77	84	7

1993	Spring Overturn							
Lake	TP	TN	TP	SRP	NO3	TKN	TN	TN:TP
Redfish	10	63	7	1	2	63	65	9.7
Alturas	15	87	9	1	3	92	95	11
Pettit	9	109	7	2	2	73	76	11.7
Stanley	16	167	8	1	5	89	94	11.9

1992	Spring Overturn							
Lake	TP	TN	TP	SRP	NO3	TKN	TN	TN:TP
Redfish	6	66	8	2	7	44	51	6.1
Alturas	10	74	8	1	4	74	78	9.9
Pettit	9	87	6	2	5	83	87	14.7
Stanley	111	028		2	5	87	92	11.5

APPENDIX 3. Nutrient concentration and fluxes in Fishhook and Redfish Lake Creek inflows.

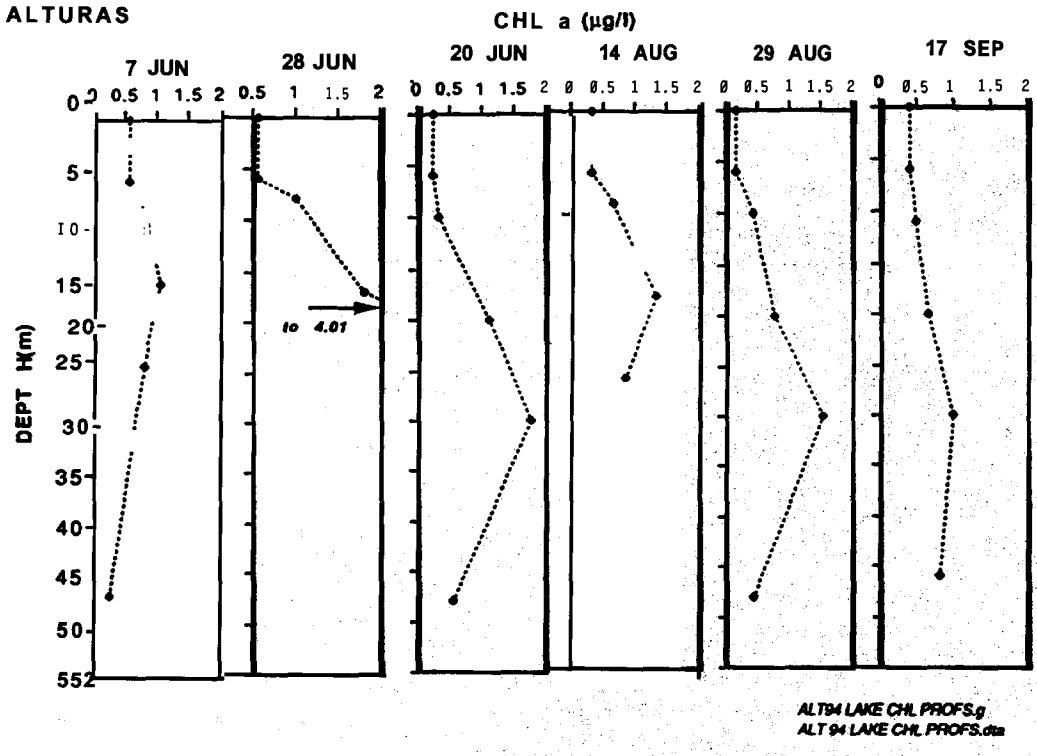
CREEK	Date	DISCHARGE (cms)	CONCENTRATION ug/L TP	TN	FLUX (KG/DAY) TP	TN
FISHHOOK	31-MAY-92	0.91	12	107	0.97	8.4
FISHHOOK	16-SEP-92	0.21	20	60	0.35	1.1
FISHHOOK	26-MAR-92	0.09	8	124	0.06	1.0
FISHHOOK	11-MAY-92	1.32	7	82	0.80	9.4
FISHHOOK	20-MAY-92	1.35	19	148	2.25	17.0
FISHHOOK	25-MAY-92	1.51	10	96	1.30	12.5
FISHHOOK	10-JUN-92	0.77	14	129	0.96	8.6
FISHHOOK	19-JUN-92	0.63	9	39	0.49	2.1
FISHHOOK	29-JUN-92	1.04	15	120	1.35	10.8
FISHHOOK	09-JUL-92	0.63	8	56	0.44	3.0
FISHHOOK	21-JUL-92	0.56	9	55	0.44	2.7
FISHHOOK	11-AUG-92	0.32	9	71	0.25	2.0
FISHHOOK	25-AUG-92	0.28	15	65	0.36	1.6
FISHHOOK	11-SEP-92	0.21	12	61	0.22	1.1
FISHHOOK	13-SEP-92	0.23	18	110	0.36	2.2
FISHHOOK	16-SEP-92	0.20	19	65	0.33	1.1
FISHHOOK	22-SEP-92	0.17	11	51	0.16	0.7
FISHHOOK	25-SEP-92	0.30	17	103	0.44	2.7
FISHHOOK	10-OCT-92	0.19	9	55	0.15	0.9
FISHHOOK	16-NOV-92	0.21	9	75	0.16	1.4
FISHHOOK	19-APR-93	0.16	11	90	0.16	1.4
FISHHOOK	03-MAY-93	0.38	19	104	0.61	3.4
FISHHOOK	12-MAY-93	1.31	42	439	4.74	49.7
FISHHOOK	16-MAY-93	2.88	31	263	7.64	70.4
FISHHOOK	17-MAY-93	3.03	158	1617	41.62	397.1
FISHHOOK	22-MAY-93	2.86	14	48	3.35	11.9
FISHHOOK	01-JUN-93	3.32	5	194	1.37	55.6
FISHHOOK	08-JUN-93	1.94	11	125	1.79	21.0
FISHHOOK	20-JUN-93	3.77	47	217	15.33	70.7
FISHHOOK	22-JUN-93	5.38	17	185	7.87	86.0
FISHHOOK	01-JUL-93	1.88	8	63	1.35	10.2
FISHHOOK	15-JUL-93	1.62	12	50	1.64	7.0
FISHHOOK	23-JUL-93	1.36	3	127	0.34	14.9
FISHHOOK	01-AUG-93	1.36	7	88	0.79	10.3
FISHHOOK	08-AUG-93	1.36	5	68	0.54	8.0
FISHHOOK	14-AUG-93	1.26	13	107	1.36	11.6
FISHHOOK	22-AUG-93	0.95	9	72	0.72	5.9
FISHHOOK	30-AUG-93	0.69	4	93	0.22	5.5
FISHHOOK	04-SEP-93	0.58	8	128	0.38	6.4
FISHHOOK	11-SEP-93	0.55	5	73	0.24	3.5
FISHHOOK	18-SEP-93	0.52	10	59	0.46	2.6
FISHHOOK	08-OCT-93	0.42	8	40	0.28	1.5
FISHHOOK	08-NOV-93	0.42	10	70	0.38	2.5
FISHHOOK	16-FEB-94	0.09	15	56	0.12	0.4
FISHHOOK	14-MAY-94	1.41	9	83	1.10	10.1
FISHHOOK	25-APR-94	0.59		98	0.00	5.0
FISHHOOK	27-MAY-94	1.95	7	122	1.15	20.6
FISHHOOK	02-JUN-94	0.49	7	52	0.30	2.2
FISHHOOK	12-JUN-94	0.62	8	65	0.45	3.5
FISHHOOK	26-JUN-94	0.62	11	77	0.56	4.1
FISHHOOK	20-JUL-94	0.36	10	80	0.31	2.5
FISHHOOK	09-AUG-94	0.26	12	90	0.26	2.0
FISHHOOK	30-AUG-94	0.21	14	84	0.25	1.5

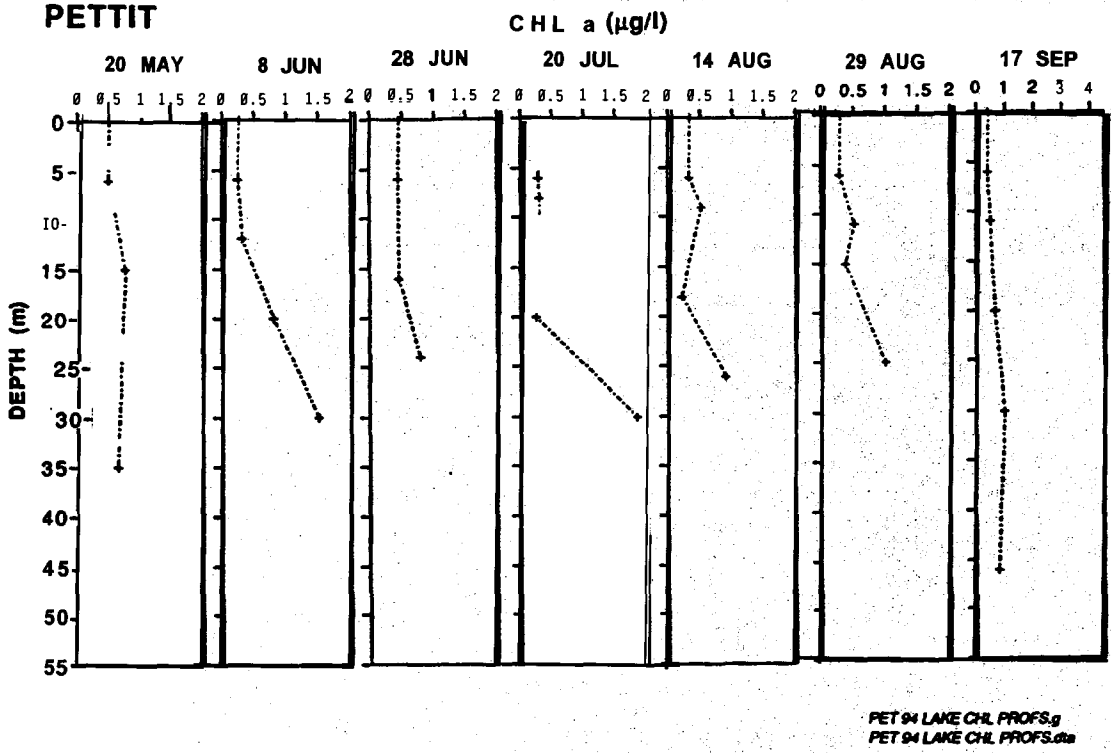
CREEK	Date	DISCHARG (cms)	CONCENTRATIO ug/L TP	TN	FLUX (KG/DAY) TP	TN
REDFISH	INLET 26-MAR-9	0.25	7	80	0.15	1.3
REDFISH	INLET 31 -MAY-9	3.30	8	80	1.71	22.7
REDFISH	INLET 13-MAY-9	2.38	8	53	1.22	10.8
REDFISH	INLET 25-MAY-9	4.41	9	84	3.43	24.4
REDFISH	INLET 12-JUN-9	2.39	5	111	1.12	22.9
REDFISH	INLET 19-JUN-9	1.40	9	57	1.09	8.9
REDFISH	INLET 29-JUN-9	2.34	39	185	7.88	33.4
REDFISH	INLET 09-JUL-92	1.30	7	83	0.79	7.1
REDFISH	INLET 21 JUL-92	1.04	8	58	0.72	5.2
REDFISH	INLET 01 -AUG-9	0.81	19	88	1.34	8.0
REDFISH	INLET 11 -AUG-9	0.89	7	58	0.42	3.3
REDFISH	INLET 25-AUG-9	0.89	9	52	0.54	3.1
REDFISH	INLET 13-SEP-9	0.49	7	38	0.30	*1.5*
REDFISH	INLET 18-SEP-9	0.53	7	42	0.32	1.9
REDFISH	INLET 1 o-OCT-9	0.57	8	58	0.30	2.8
REDFISH	INLET 15-NOV-9	0.83	7	88	0.38	3.7
REDFISH	INLET 21 -MAR-9	0.49	7	59	0.28	2.5
REDFISH	INLET 19-APR-9	0.51	4	82	0.17	2.7
REDFISH	INLET 16-may-9	7.24	13	131	8.07	81.9
REDFISH	INLET 22-may-9	7.82	5	129	3.48	87.2
REDFISH	INLET 01-JUN-9	9.28	7	108	5.81	88.8
REDFISH	INLET 07-jun-93	4.09	9	77	3.02	27.2
REDFISH	INLET 1 0-JUN-9	4.41	10	75	3.98	28.8
REDFISH	INLET 14-jun-93	3.99	7	79	2.35	27.2
REDFISH	INLET 23-jun-93	8.07	10	79	5.45	41.4
REDFISH	INLET IOJUL-93	4.89	3	99	1.22	41.8
REDFISH	INLET 17-JUL-93	2.78	3	70	0.74	18.7
REDFISH	INLET 23JUL-93	2.78	7	89	1.74	16.8
REDFISH	INLET 01 -AUG-9	2.77	9	102	2.05	24.4
REDFISH	INLET 08-AUG-9	2.77	4	88	0.84	21.1
REDFISH	INLET 16-aug-93	3.58	8	88	1.94	28.8
REDFISH	INLET 22-AUG-9	2.34	8	91	1.19	18.4
REDFISH	INLET 28-AUG-9	1.37	9	89	1.02	8.2
REDFISH	INLET 04-SEP-9	0.95	5	145	0.40	11.9
REDFISH	INLET 14-SEP-9	1.20	2	84	0.25	8.8
REDFISH	INLET 08-oct-93	0.84	4	88	0.30	4.9
REDFISH	INLET 08-nov-93	0.59	5	92	0.25	4.7
REDFISH	INLET 14-MAY-9	4.90	7	39	3.13	18.5
REDFISH	INLET 27-MAY-9	4.52	5	28	1.78	10.2
REDFISH	INLET 02 JUN-9	0.85	5	15	0.35	1.1
REDFISH	INLET 12 -JUN-9	3.48	5	40	1.81	12.0
REDFISH	INLET 20JUL-94	0.75	8	43	0.54	2.8
REDFISH	INLET 09-AUG-9	0.40	9	11	0.31	0.4
REDFISH	INLET 30AUG9	1.01	8	82	0.87	5.4

Bernoulli\budget\flux\rf92_94.wq1

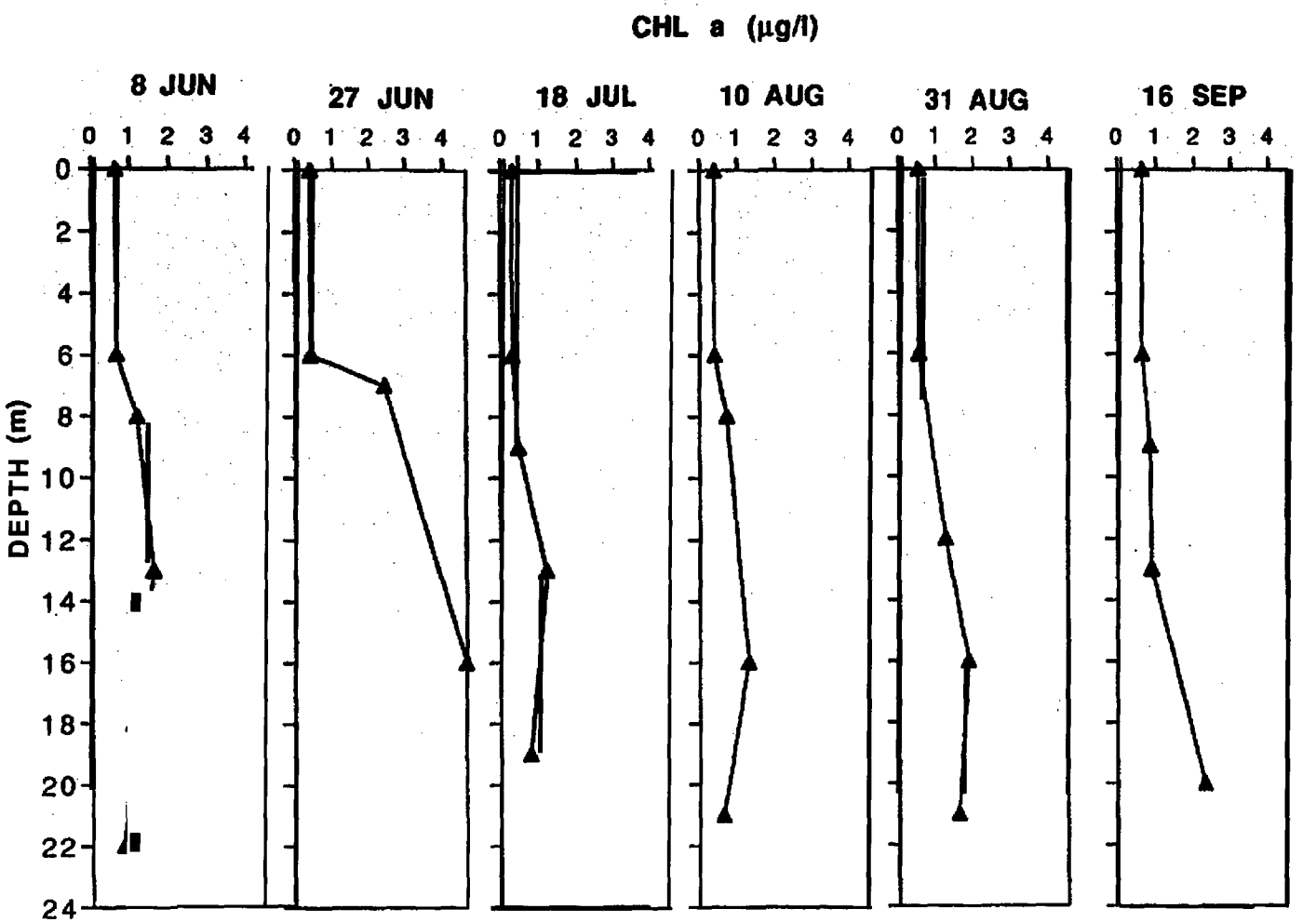
Appendix 4.

Chlorophyll a (µg/l) profiles for Alturas, Pettit, and Stanley Lakes, 1994


ALTURAS
CHL a (µg/l)
7 JUN
28 JUN
20 JUN
14 AUG
29 AUG
17 SEP
DEPTH (m)
to 4.01
ALT94 LAKE CHL PROFS.g
ALT 94 LAKE CHL PROFS.dta


PETTIT
CHL a (µg/l)
20 MAY
8 JUN
28 JUN
20 JUL
14 AUG
29 AUG
17 SEP
DEPTH (m)
PET 94 LAKE CHL PROFS.g
PET 94 LAKE CHL PROFS.dta

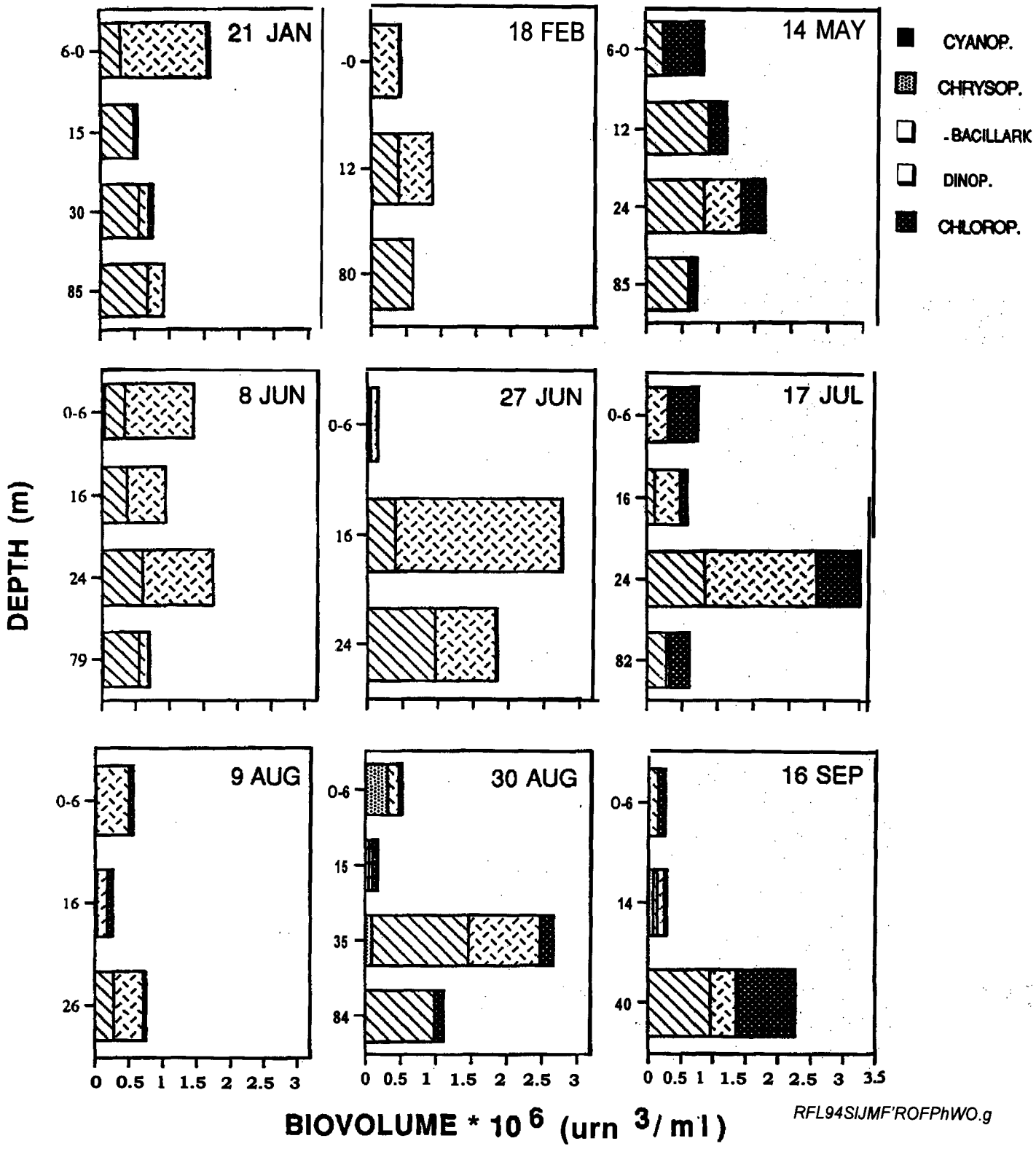
STANLEY



STA 94 LAKE CHL PROFS.g
STA 94 LAKE CHL PROFS.dta

Appendix 5.

Depth profiles of phytoplankton species composition by depth at each sample date Rehfish Lake, 1994.


21 JAN
18 FEB
14 MAY
8 JUN
27 JUN
17 JUL
9 AUG
30 AUG
16 SEP
CYANOP.
CHRYSOP.
-BACILLARK
DINOP.
CHLOROP.
DEPTH (m)
BIOVOLUME * 10 6 (urn 3/ml)
RFL94SIJMF'ROFPhWO.g

CHAPTER2

Adding nutrients to enhance production of endangered sockeye salmon: Trophic transfer in an oligotrophic lake

Phaedra Budy

Chris Luecke

Wayne A. Wurtsbaugh

INTRODUCTION

The effects of nutrient additions on ecosystem productivity are well known. These bottom-up effects have been predicted from theory (Oksanen et al. 1981; Liebold 1989; Power 1992), described from empirical relationships (McQueen et al. 1986; Ney 1990), and tested with whole ecosystem experiments (McNaughton 1977; Langeland 1982; Peterson et al. 1993). Adding nutrients to enhance ecosystem productivity has been practiced for thousands of years and forms the foundation of agricultural and aquacuhure practices. In fisheries sciences, lake fertilization has been shown to increase production of commercially exploited stocks for decades (Nelson and Edmondson 1955) and is now commonly used to increase growth and survival of juvenile sockeye salmon (*Oncorhynchus nerka*) (Stockner 1981, 1987; Stockner and Hyatt 1984; Kyle et al. 1994).

In spite of these well known effects of nutrient additions, we know of no instance where ecosystem fertilization has been used as a management tool to assist with the recovery of endangered populations. Although bottom-up effects obviously occur, our ability as ecosystem managers to predict such occurrences is weak. In fact, much of current ecological debate centers on determining under what conditions bottom-up effects prevail over top-down effects in defining ecosystem function (Carpenter and Kitchell 1988).

In many studies where nutrients have been added to aquatic systems, primary production has been enhanced (Drenner et al. 1986). In some cases, however, intense selective predation on primary consumers can weaken or obliterate the effects of nutrient additions resulting in a strong top-down control of ecosystem function (Carpenter et al. 1994). Omnivory or diet switching by consumers can also reduce trophic transfer resulting in "middleout" effects (Devries and Stein 1992). Finally, recruitment limitation of key species can uncouple the trophic link between primary producers and consumers (Koenings and Burkett 1987).

In this paper we examine the potential for using lake fertiliiion to enhance growth and survival of endangered sockeye salmon in Redfish Lake, Idaho. We conducted an experiment in large, impermeable enclosures to compare patterns of trophic transfer in control and nutrient addition treatments. We compare phytoplankton biomass, Primary production, zooplankton biomass, and fish growth between treatments. We used juvenile kokanee (a conspecific but non-endangered population of 0. *nerka*) as a surrogate for sockeye salmon in this experiment, and assessed the degree to which nutrient additions were manifested in increases in biomass of organisms at higher trophic levels.

Methods

Fertilization experiments were performed in nine large limnocorral enclosures, S-m in diameter and 17-m deep, made of impermeable, polyethylene plastic. Each limnocorral held approximately 350 m^3 of water. Bach enclosure was slowly dropped from the surface with the bottom open allowing it to fill with lake water for 24 h. Thus initial conditions within each limnocorral were similar to the lake. After filling, each limnocorral was tied off at the

bottom by SCUBA divers preventing further exchange of water.

Based on results from a preliminary limnocorral experiment, we chose to add the nutrients to the liiocorrals for a period of time before the fish were added. This allowed the phytoplankton and zooplankton communities to respond to fertilizer additions and adjusted for lag times between fertilizer addition and zooplankton production before planktivorous fish were added (Budy et al. 1994). Thus the experiment was conducted in two parts: in Part 1 (13 June, 1994- 12 August 1994), nutrient additions were added to three of nine limnocorrals, and in Part II (12 August, 1994- 11 September 1994), nutrient additions continued and fish were added to fertilized (3) and unfertilized (3) limnocorrals.

During the first 6 weeks of the experiment (Part 1; 13 June - 12 August 1994) six of nine enclosures were used as *controls* and three of the nine enclosures *were nutrient addition* treatments. The nutrient addition treatments received fertilizer additions of nitrogen and phosphorous applied regularly to the metahmnion of each enclosure. Metalimnetic fertilizations have been shown to effectively stimulate plankton production, while minimizing decreases in water transparency (Gross et al. 1994). Nutrients were added at 11 m during Part I of the experiment and at 12 m day Part II of the experiment as the thermocline moved deeper. The control treatments received no fertilizer. After 6 weeks, we began Part II of the experiment by adding fish to three of the control treatments (hereafter referred to as *fish-controls),* and to the three -nutrient addition treatments (hereafter referred to *as nutrient additions + fish).* Three treatments were left as *controls* and received neither fish nor nutrients for the entire experiment.

Nitrogen and phosphorous were added to the metalimnion (11-12 m) of nutrient addition treatments in the form of $(NH_4)_2HPO_4$ and NH_4NO_3 every 10 d after Limnological sampling. The nutrients were added at a 30:1 by mass TN:TP ratio (13.5: 1 molar ratios) with target loading of 5 µg/l of phosphorous and 150 µg/l of nitrogen to each nutrient addition treatment. This high TN:TP ratio was used to decrease the likelihood of stimulating nitrogen-fixing cyanobacteria (Schindler 1977).

Juvenile kokanee (land-locked *Oncorhynchus nerka*) were used as surrogates for the endangered sockeye salmon in these experiments. The kokanee were trapped shortly after emergence in May from Fishhook Creek, a Redfish Lake tributary, and retained in permeable mesh net pens in Redfish Lake until addition to limnocorral fish treatments on 12 August, 1994.

The limnocorrals were sampled every 10 d for 'Limnological parameters including nutrients, chlorophyll *a,* zooplankton, Secchi depth transparency, temperature, and oxygen. Water samples for dissolved and total nutrient depth profiles (TP, SRP, TKN, and NO,) were sampled four times throughout the experiment (13 June, 1994, 2 July, 2 August, and 3 1 August).

Secchi depth measurements were recorded in each limnocorral to estimate water clarity. Temperature and oxygen profiles were sampled using a Hydrolab® meter in

one limnocorral representing each of the three treatments every 10 d. Previous sampling indicated that little variation in temperature and oxygen occurred among limnocorrals in these types of experiments (Rudy et al. 1994). The three corrals chosen to represent each treatment were changed each sampling period.

Water for chlorophyll a analysis and nutrients were taken using weighted Tygon tubes for integrated depth samples and a 4 L Van Dom bottle sampler for discrete depth samples. Epilimnetic water samples were taken using a weighted 6-m Tygon tubing, and whole limnocorral water column samples were taken using a weighted 17 m Tygon tubing every 10 d. Discrete water samples were also collected from the metalimnion (11-12 m) every 10 d and from near the bottom of each limnocorral (17 m) on selected dates.

Samples collected for nutrient analysis were placed in polyethylene bottles first rinsed with 0.1 N HCL and then rinsed three times with aliquots of the actual sample. Nutrient samples were frozen until analysis. Nutrient samples were analyzed for total nitrogen (TN) and total phosphorous (TP) concentrations. TN was calculated from the sum of Total Kjeldahl Nitrogen (TKN) and nitrate (NO_3-N). Unfiltered water was used for TKN and TP analyses; samples analyzed for (NO_3-N) and SRP were filtered through 0.45-μm membrane filters. TP samples underwent a persulfate digestion and were then analyzed colorimetrically in our lab (Utah State University Limnology Laboratory) using the molybdate - ascorbic acid method (APHA 1992). Nitrogen analyses were conducted by the University of California, at Davis, Limnology Laboratory) using a Kjeldahl digestion for TKN and the hydrazine method for NO_3. Replicates, spikes, and standard solutions were used for quality control/ quality assurance.

Chlorophyll a measurements were used as a measure of phytoplankton standing crop. Two SO-ml aliquots per sample were filtered through a 0.45-μm cellulose acetate membrane filters. Filters were either temporarily frozen or placed directly into 6 ml of 100% methanol for chlorophyll pigment extraction in the dark for 24 h. Extracts were then analyzed before and after acidification (Holm-Hansen and Rieman 1978) using a Turner model 111 tluorometer with corrections made for phaopigments.

We preserved water for phytoplankton enumeration from the chlorophyll *a* and nutrient water samples described above. Phytoplankton samples were preserved with Lugol's iodine solution, and a 100-ml aliquot from each phytoplankton sample was filtered through a 0.45-μm cellulose filter. The filters were cleared and permanently mounted (Crumpton 1987). Cells were counted in a minimum of 10 fields per slide at 400x; the dimensions of a minimum of 10 individuals in each taxa were measured to calculate biovolume (Wetzel and Liens 1990). Phytoplankton were taxonomically classified as cyanobacteria (blue-green algae), chlorophyta (green algae), chtysophyta (*Dinobryon* sp.), bacillariophyta (diatoms), and Diiophyta (*Peridinium* sp.) . Phytoplankton data were analyzed as biovolume by taxon, depth, and date.

Periphyton accumulation on the sides of the limnocorrals was evaluated by

suspending weighted 17 m * IO-cm wide strips of the same polyethylene limnocorral material in the center of each corral. Every other sampling period, ~ 21 days, we removed two 12 mm squares from four depths on each strip (1, 6, 11-12, 17m) to measure periphyton accumulation on each strip. These squares were placed immediately in 6 ml of methanol, left to extract in the dark for 24 h, and then analyzed fluorometrically for chlorophyll *a* according to the methods described above.

Zooplankton were sampled from two depth strata, 10-O m and 17-10 m, every 10 d in each limnocorral. Two replicate samples were taken over each depth strata using a closing 80-μm mesh net, 35 cm in diameter and 150-cm long equipped with an anti-reverse General Oceanics^{R} flow meter. Samples were immediately preserved in a 10% formalin-sucrose solution and subsequently counted and measured. Biomass was calculated using length-mass regressions (McCauley 1984; Koenings et al. 1987), and densities were corrected for the volume sampled. Cladoceran egg-ratios (# eggs/ # adults) were calculated according to the methods of Paloheimo (1974).

Fiie juvenile kokanee were added each of the three nutrient addition treatment limnocorrals and three of the control limnocorrals on 12 August 1994. Fish were anesthetized uni MS222 measured (mm), weighed (g of wet weight), and given a unique fin clip for individual measures ofgrowth. Fish were allowed to recover for several hours before addition to each limnocorral. A subsample of fish from the lot stocked was sacrificed and frozen for subsequent dry weight analysis. At the end of the experiment, fish were removed from limnocorrals using dry ice asphyxiation, measured, weighed, and frozen and subsequently dried to constant weight at 60°C. At the end of the experiment we were able to recover 87% (26/30) of the fish initially stocked.

Repeated measure ANOVAs were used to analyze the nutrient, chlorophyll, Secchi, and zooplankton data for statistical significance accounting for time and initial differences among treatments (Wilkinson 1990). The experiments was statistically treated as two parts described above (Part I = nutrient additions, Part II= nutrient + fish additions). Thus there were two treatments for Part I with three replicates for the *nutrient additions* and six replicates for the *controls*. For Part II, there were three replicates of *nutrient plus fish additions*, three replicates. of *controls* without fish, and three replicates of *fish-controls*. Differences in fish growth (g/g/day) using both wet and dry weights were analyzed for statistical significance using a standard ANOVA and pooling fish by treatment.

RESULTS

Temperature and oxygen conditions were similar among treatments. and showed the same general pattern observed in the lake. Oxygen conditions remained above 5 mg/l throughout the duration of the experiment. Surface temperatures ranged from 1 1-15° C, metalimnetic (11-12 m) temperatures ranged from 6.5-12°C, and temperatures at the bottom of the bag remained below 7.5°C

Water transparency

Secchi depth transparency in the limnocorrals showed a general pattern of increasing water clarity through the

summer similar to that observed in the lake (Figure 1). The shallowest Secchi depth transparencies were observed on 22 June 1994, and the deepest transparencies were observed on 11 September 1994 in ail treatments and in the lake. Mean Secchi depth transparencies ranged from 9.5 - 13.4 m in nutrient addition treatments and from 9.5 - 17.9 m in control limnocorrals with measurements generally 2-4 m shallower in the nutrient addition limnocorrals compared to the control corrals. Before the fish were added, Secchi depth transparencies were *statistically* shallower in *nutrient* addition treatments than in the control treatments (df= 6, F= 2.99, P= .016). No statistical differences occurred between treatments after fish were added (df= 4, F= .22, P= .92). Control Secchi depth transparencies were similar to those observed outside of liiocorrals in the lake (Figure 1).

Nutrient Analyses

Means and ranges for IO-day interval nutrient samples are presented for the epilimnetic and whole water column concentrations in Table 1 (TP, SRP, TKN, NO, and the N:P ratio). The mean for all epilimnetic (6-O m) nutrients analyzed was higher in the nutrient addition treatments over the controls except for the SRP data which were higher in the controls. The mean N:P ratio was identical in both treatments. The same pattern, was observed in whole limnocorral samples (17-O m) in that all nutrient concentrations measured were higher in the nutrient additions over the controls except SEP. The N:P ratio was slightly higher in the controls over the nutrient additions in the whole limnocorral samples. In both the epilimnetic and the whole limnocorral samples, the difference between the two treatments was small for all nutrients measured except TKN. TKN in the whole limnocorral nutrient addition treatments was -20 % higher in nutrient additions,. but overall the nutrient differences *between* the *two treatments were minor.*

The total phosphorous' (TP) for the duration of the experiment in both the epilimnetic samples and the 17-O m samples was highly variable (Figure 2). The highest peaks in TP in both the epilimnetic samples and the 17-O m samples occurred in mid-July and late August. TP in the epilimntic fish- control treatments was slightly higher than the control treatments, but the fish-control TP in the 17-O m samples was similar to those controls. All treatments in both zones converged on values around 12 µg/l, the sameTP concentrations as those measured at the onset of the experiment.

Depth profiles of TP and TN showed little variation in TP with depth through time except an increase in hypoliietic TP on 2 Aug 1994. TN was markedly higher at all depths on 2 Aug 1994 especially in the metalimnion (the depth at which nutrients were added) (Figure 3). Total nitrogen values in the limnocorral experiment were highly variable among replicates, and thus SE bars are quite large for this parameter. Ammonia (NH_4) was especially difficult to measure and values were often much larger than biologically plausible suggesting contamination or analysis problems.

Phytoolankton

Mean epilimnetic chlorophyll *a* was greater in the nutrient addition limnocorrals than in the corrals which received no nutrients (Figure 4A).

Chlorophyll *a* in the nutrient addition limnocorrals was generally 0.5-1 mg/m³ higher than in control treatments with the exception of a peak in chlorophyll a on 22 June, 1994 when chlorophyll levels in nutrient addition treatments were approximately 2 mg/m³ greater. The range in mean epiliietic chlorophyll *a* concentrations for the duration of the experiment was 0.13 - 3 mg/m³ in the nutrients addition treatments and 0.2- 0.9 mg/m³ in the control treatments. Nutrient additions statistically increased epilimnetic chlorophyll *a* concentrations before fish were added to the experiment (df= 6. F= 11.17, P= 0.00), but there was no difference among treatments after fish were added (df= 4, F= 0.22, P= 0.92).

Mean chlorophyll *a* concentrations in the metahmnion (measured at 11 m before fish were added on 12 August, 1994 and 12 m after that point) ranged from 0.18- 5.95 mg/m³ in nutrient additions and from 0.14 mg/m³ to 1.95 mg/m³ in control corrals (Figure 4B). Metalimnetic chlorophyll a concentrations showed the same high peak on the 22 June, 1994 in nutrient addition and control treatments as observed in the epilimnion. The difference in chlorophyll *a* concentrations among treatments was statistically significant before fish were added (df= 6, F= 7.84, P= 0.00) but no significance was detected after fish were added (df= 4, F= 0.45, P= 0.77).

Chlorophyll *a* concentrations measured from whole water column integrated samples (17-O m) showed the same general pattern as the epiliietic and metalimnetic chlorophyll *a* measurements (Figure 4C). Nutrient addition + fish treatments ranged from 0. 1- 4.91 mg/m³ and control treatments ranged from 0.1 to 1.19 mg/m³. Overall, the highest values for chlorophyll *a* concentration and the greatest difference between controls and nutrient addition treatments were observed in the metalimnetic samples (11-12 m)--the depth of nutrient application. Additional chlorophyll data shown as depth profiles through time are provided in Appendix 6.

Nutrient additions enhanced phytoplanlcton biovolume in whole water column samples (17-O m) on seven out the eight dates analyzed (Figure 5). Diiophyta and Bacillariophyta dominated the phytoplankton biovolume in all treatments in the early part of the experiment. Early in the experiment nutrient additions stimulated Dinophyta and bacilariophyta 2-3 fold above control treatments. Nutrient addition appeared to stimulate Chlorophyta biovolume as the experiment progressed. After the fish were added, a small peak in cyanobacteria biovolume was observed in nutrient additions while Chrysophytes appeared in the controls and fish controls (no nutrient treatments). Phytoplankton biovolume was higher at all depths and on all dates in the nutrient addition treatments as compared to control treatments (Figure 6). General trends of increasing depth of maximum phytoplankton biovolume and an overall decline in biovolume at all depths through time was observed in all treatments. The greatest phytoplankton biovolume was observed at 17 m in control treatments and at 11 m in nutrient addition treatments on 2 July, 1994. Dinophyta and Bacillariophyta species dominated on 2 July, 1994 with Dinophyta species making up the bulk of the increase in total biovolume observed in nutrient additions. This pattern switched on 1 August, 1994 such that biovolume was considerably

reduced at all depths in the controls, and the greatest biovolume was observed at 17 m in nutrient addition treatments. Further, the species composition became dominated almost exclusively by Bacillariophyta in nutrient addition treatments. By 31 August 1994, all treatments showed a substantial reduction in phytoplankton biovolumes at all depths and in all treatments. Periphyton communities were not well established in any of the corrals until August (Figure 7). Subsequently, nutrient additions stimulated periphyton at all four depths sampled. Nutrients additions stimulated periphyton most at 12 m (the depth of nutrient application), and to a lesser degree at 17 m. The least amount of periphyton growth was observed in epilimnetic samples (1 and 6 m) in all treatments. Periphyton was negligible in control and fish-control treatments throughout the experiment.

Zoonlankton

Zooplankton total biomass showed both a clear response from nutrient additions and an overall increase during the experiment (Figure 8). Total zooplankton biomass was greater in nutrient addition treatments than in control treatments from 12 July 1994 to the end of the experiment. The increase in zooplankton biomass in nutrient additions was statistically significant before fish were added to the experiment (df= 6, F= 3.92, P= 0.00) but there was no statistical diierence among treatments after fish were added (df= 4, F= 1.72, P= 0.21). Mean bemuses ranged from 0.7 - 3 1.7 µg/L and 0.6 - 94.2 µg/L in control and nutrient addition treatments, respectively. After the fish were added (12 August, 1994), there was a slight increase over controls in the total zooplankton biomass in fish-addition treatments.

Total zooplankton species composition in the limnocorrals was dominated by cladoceran species such that cyciopoid zooplankton species usually composed < 5% of the total biomass. Cladoceran zooplankton species composition was dominated by *Bosmina* and *Holopedium in* all treatments early in the experiment (Figure 9). On 22 July 1994 the species composition shifted as *Daphnia* began to increase and both *Holopedium* and *Bosmina* declined. Control treatments showed a small peak in *Holopedium after* fish were added to the fish treatments (22 August 1994). This was not observed in either the fish-controls or the nutrient additions + fish. *Daphnia were* more abundant in the nutrient additions + fish than in either the controls or the fish-controls after the fish were added.

Cladoceran egg ratios were generally higher in limnocorrals receiving nutrients (Figure 10). Mean *Daphnia* and *Bosmina* egg ratios were greater in nutrient addition treatments than in controls although the differences were small. No increase in egg production in nutrient additions over controls *was* observed for *Holopedium.* Total cladoceran egg ratios were slightly higher in the nutrient addition treatments.

Fish growth

The mean growth rate (dry weights) of kokanee in the nutrient addition treatments was 57% higher than in the controls (11 vs 7%/day; Figure 11). Growth rates measured in wet weight were also higher in the nutrient addition treatments than in the controls.

Discussion

The results from these limnocorral fertilization experiments show that inorganic nutrients added to this food web resulted in an increase in phytoplankton, zooplankton, and fish growth. Addmg nutrients for a period of time prior to the addition of fish appeared to stimulate trophic transfer within the food web such that juvenile salmon could respond immediately upon stocking to increased zooplankton food availability. Furthermore, nutrient additions applied to the metalimnion of each limnocorral stimulated primary production and algal biomass primarily below the epilimnion, thus preserving water transparencies of aesthetically valuable surface waters while increasing fish growth.

In these experiments nutrient additions stimulated phytoplankton of species and size ranges grazeable by most cladoceran zooplankton (Sommer 1988; Lampert 1978; Porter 1973). The high N:P ratio appeared to suppress blooms on nitrogen-fixing cyanobacteria (Schindler 1971). A comparison of chlorophyll *a* in phytoplankton versus periphyton on the sides of the limnocorrals indicated that a substantial amount of nutrients *were* utilized by periphyton communities in the later half of the experiment. This accumulation of high levels of periphyton in the second part of the experiment likely contributed to the decline o f phytoplankton in the water column after mid-July. While the stimulation of periphyton is an alternative pathway of energy transfer that may influence whole-lake fertilization, the higher surface area: volume ratio in the limnocorrals as compared to the lake suggests that periphyton growth would be less pronounced during a whole-lake fertilization. Despite the loss of nutrients to periphyton in the liiocorrals, phytoplankton production was significantly stimulated by nutrient additions .

Although total zooplankton biomass increased considerably in nutrient addition treatments, zooplankton composition differed little among treatments. The small difference in zooplankton composition observed was due to an increase in domination by *Daphnia,* a zooplankton species preferred by juvenile salmonids (Lazzarro 1987; Luecke and Brandt 1993). Comparisons of zooplankton biomass in control treatments to that observed in the natural lake environment for similar time periods, showed that ambient densities of zooplankton in the lake were considerably lower than in control limnocorral. Thus the refuge from fish predation for the first half of the experiment may contribute to the greater zooplankton biomass observed in limnocorral enclosures over ambient lake densities. However, zooplankton biomass continued to increase in response to nutrient additions after the fish were added. Consequently, it is likely. that differences in fish growth between nutrient additions and controls would have continued to increase had the experiment run longer. The short time period over which the fish demonstrated a higher growth rate in response to nutrient additions makes small, but significant, difference in *this* growth rate more notable.

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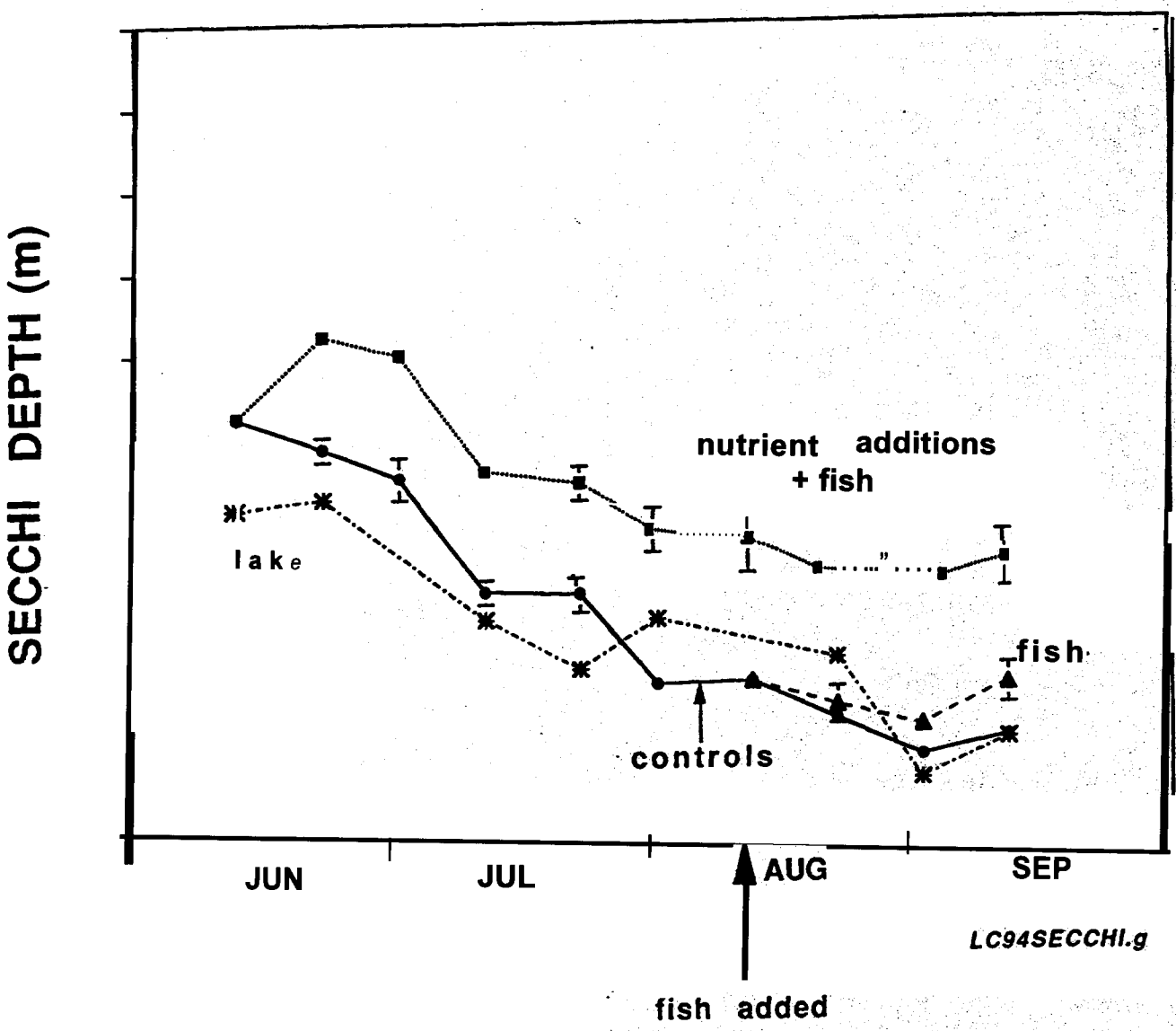


Figure I. Secchi depth distribution in the three treatments shown as the mean of three replicate limnocorrals with 1 standard error bars. The Secchi depth distribution for Redfish Lake is also shown as a reference.

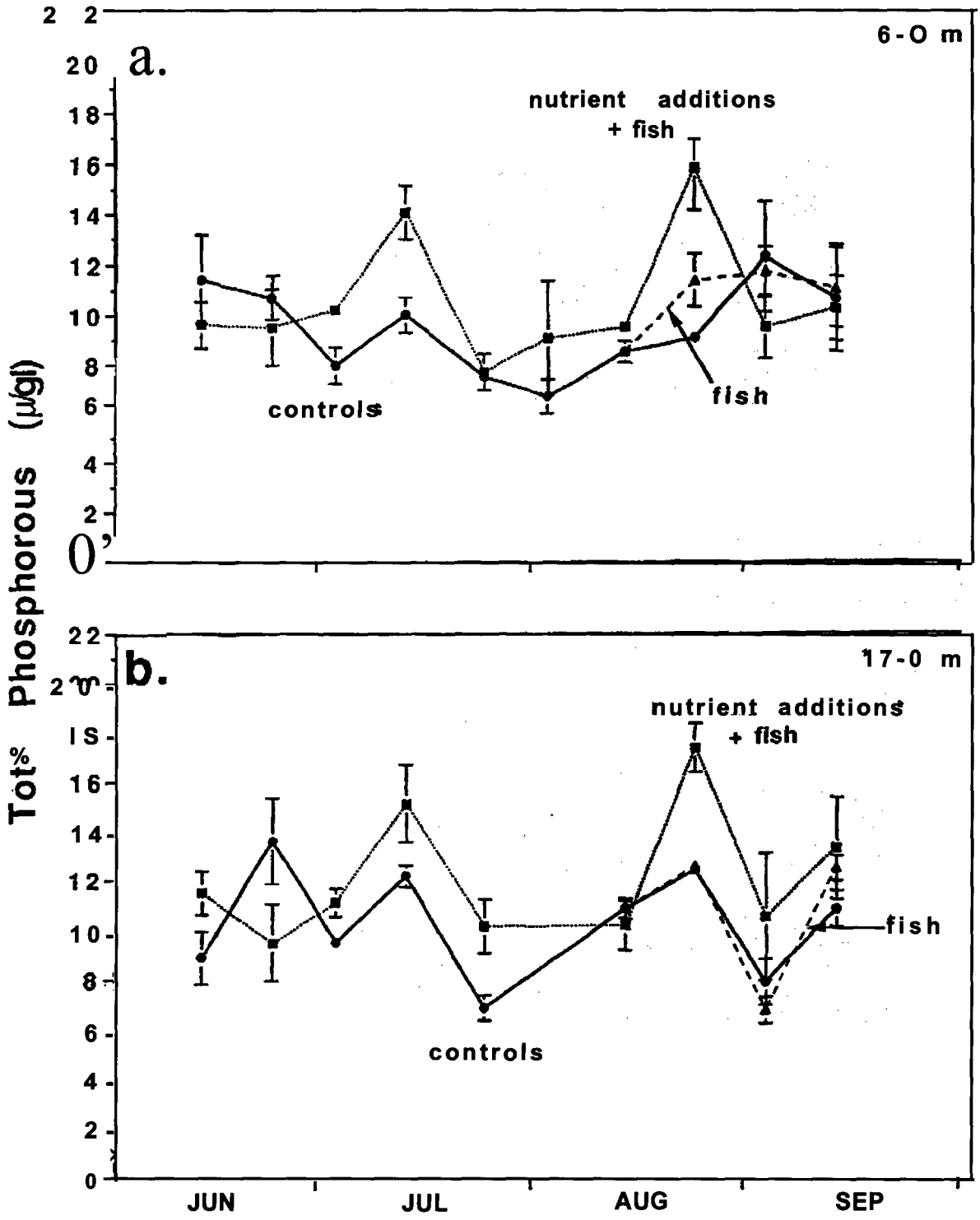


Figure 2. Total phosphorous (TP) for the three treatments from the beginning of the experiment to the end of the experiment for the limnocorral experiment, 1994. The top graph (a.) shows TP concentrations in the epilimnion (6-O m), and the bottom graph (b.)shows TP in the whole limnocorral samples (17-O m).

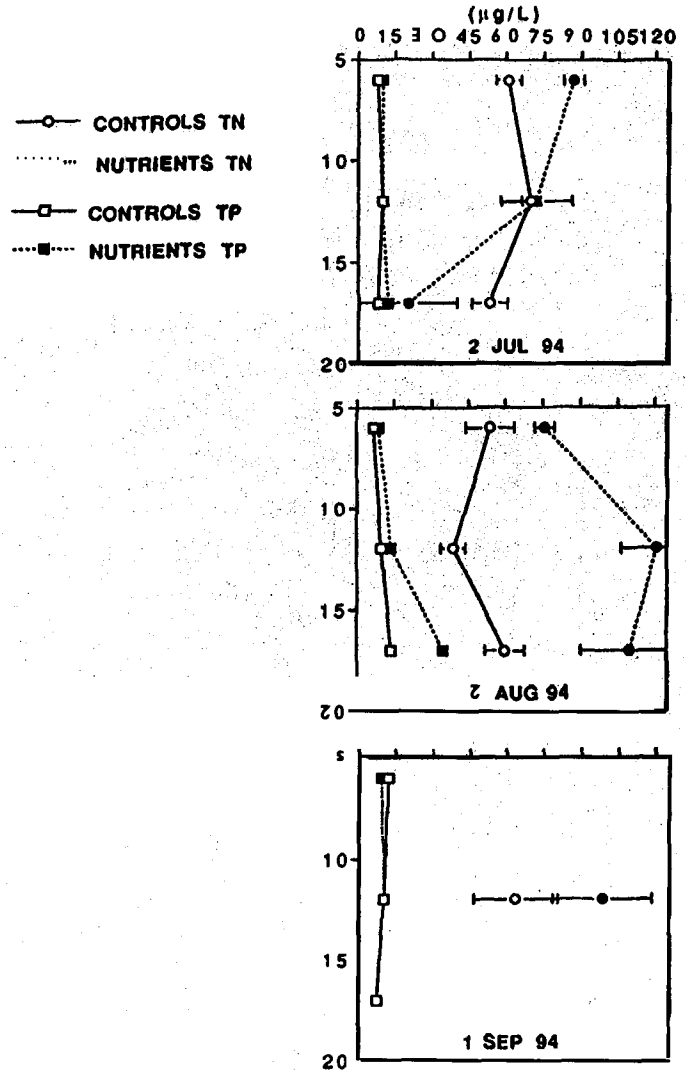


Figure 3. Total phosphorous (TP) and total nitrogen (TN) profiles for control and nutrient addition treatments by depth for three dates during the limnocorral experiment, 1994.

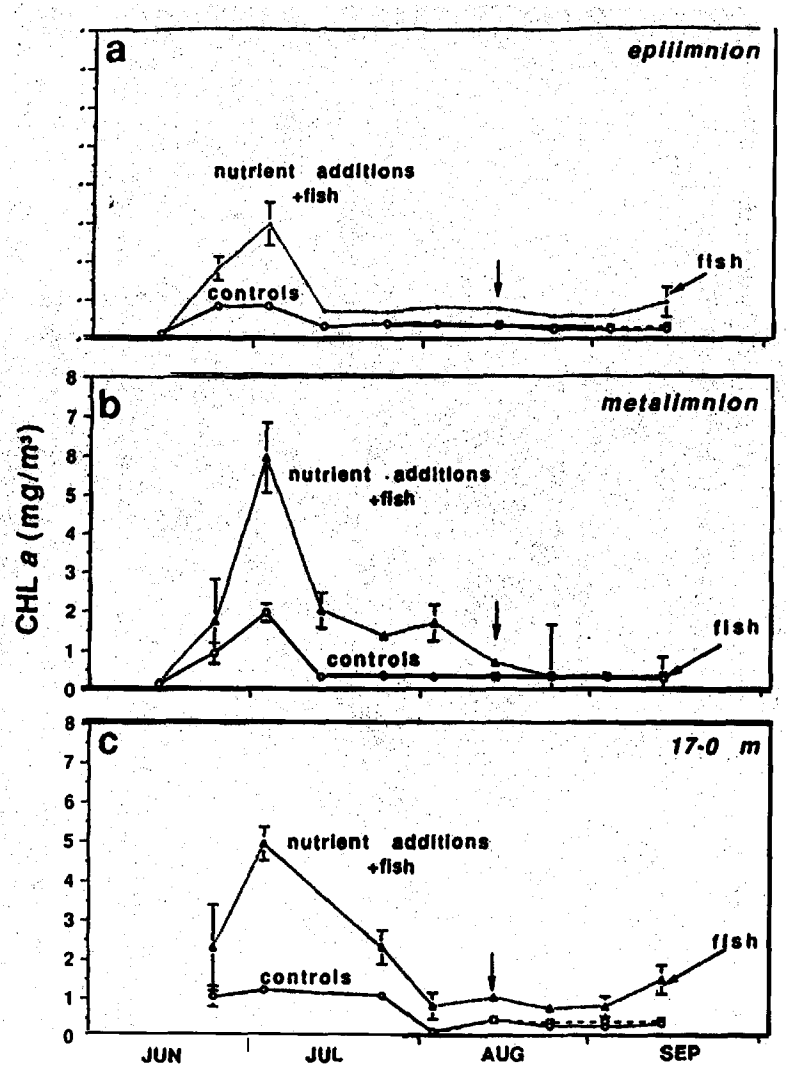


Figure 4. Chlorophyll *a* in the three limnocorral treatments shown as the mean of the three replicate limnocorrals with 1 standard error bars. The epilimnion samples (6-0 m) are shown at the top (a), the metalimnion (11 m before 13 Aug 94, 12 m after 13 Aug 94) is shown in the middle (b), and the whole limnocorral water column tows (17-0 m) on the bottom (c). The arrows point to the date the fish were added (13 Aug 94).

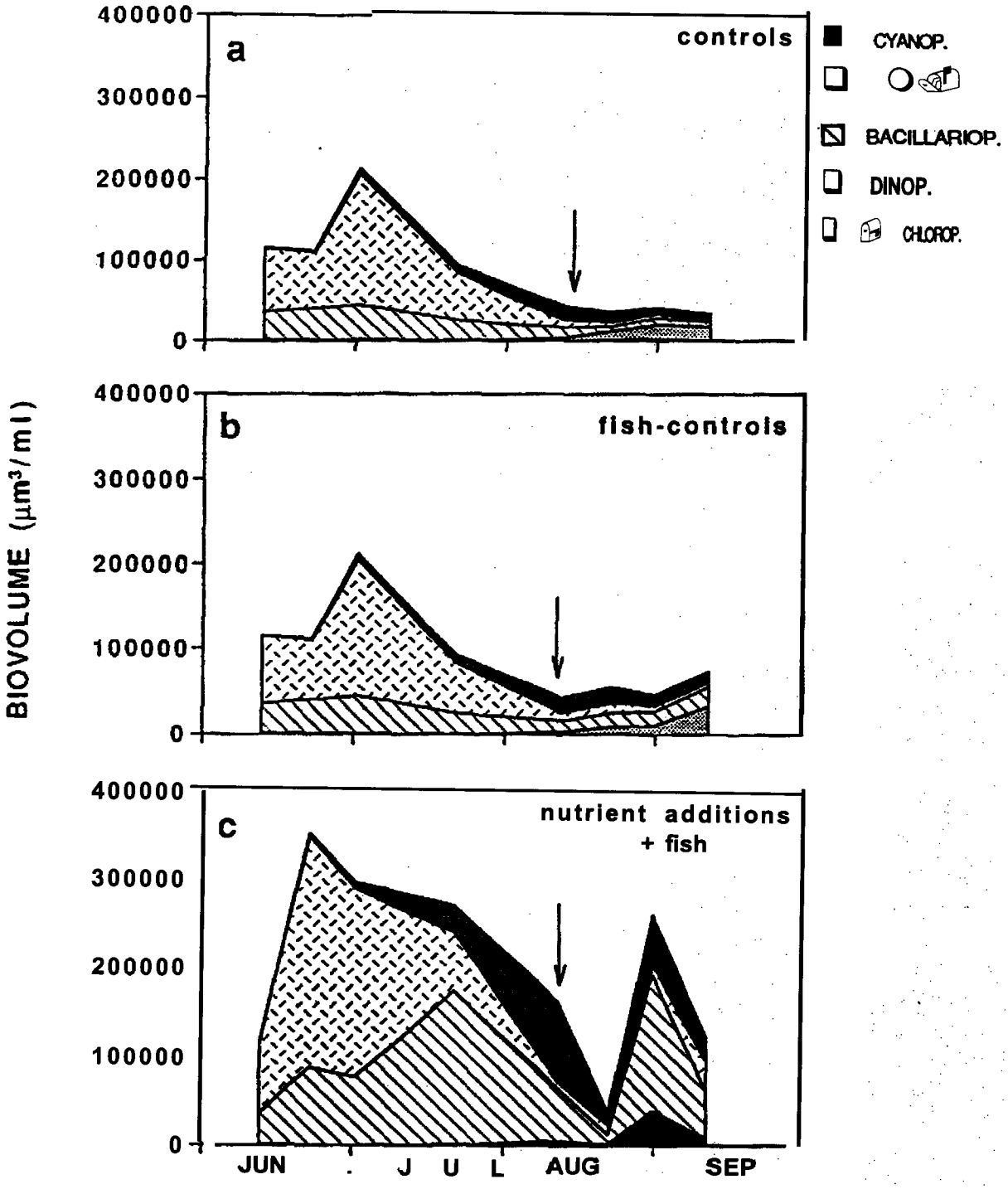


Figure 5. Seasonal changes in the biovolume of different phytoplankton taxa in whole the three treatments shown as the shown as the mean of three replicate limnocorrals. The control treatments are shown at the top (a), the fish-controls ara shown in the middle (b), and the nutrient additions + fish are shown on the bottom (c). The arrows point to the date the fish were added (13 Aug 94).

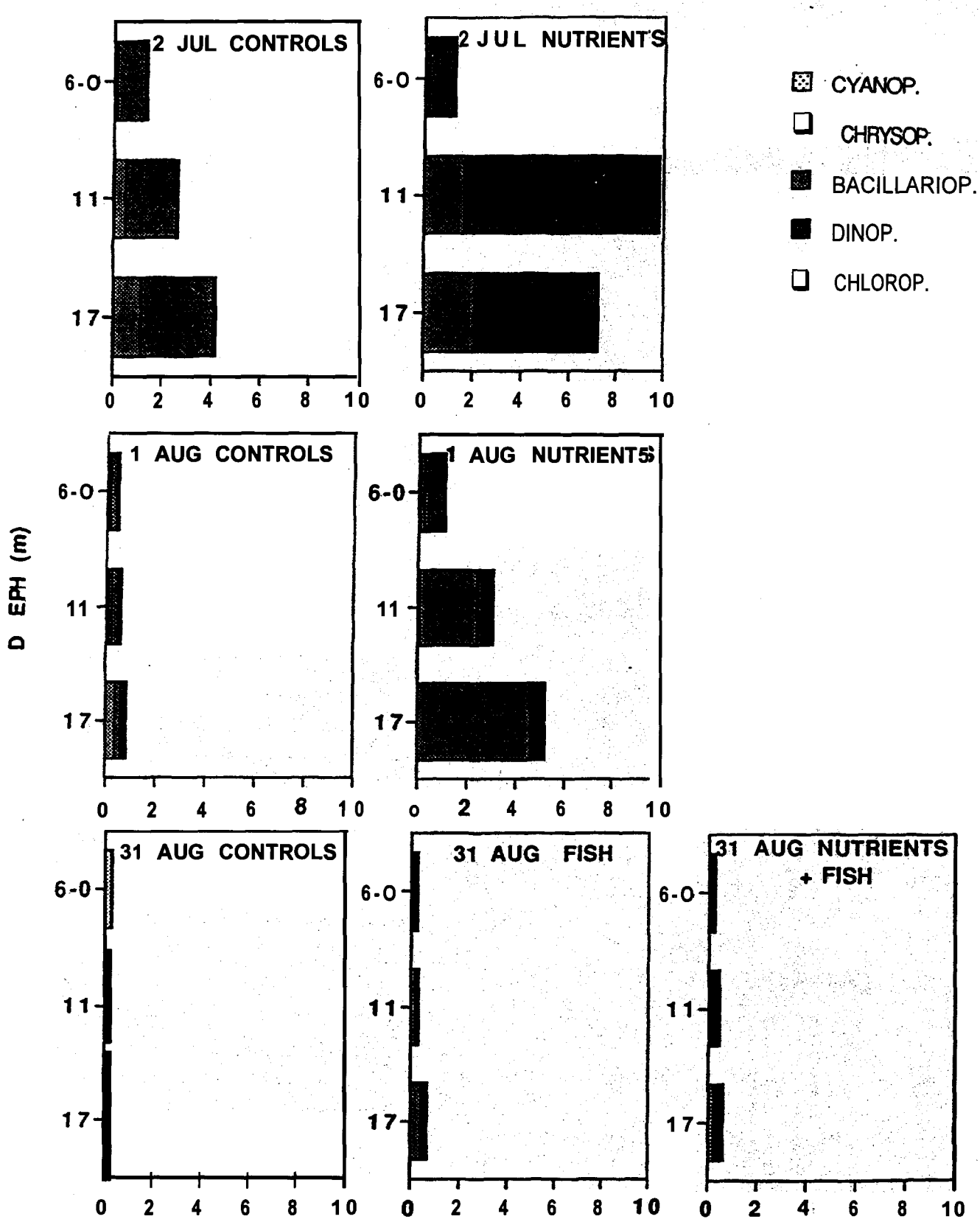


Figure 6. Mean algal biovolumes for Redfish Lake limnocorrals at epilimnetic, metalimnetic, and near-bottom depths, 1994.

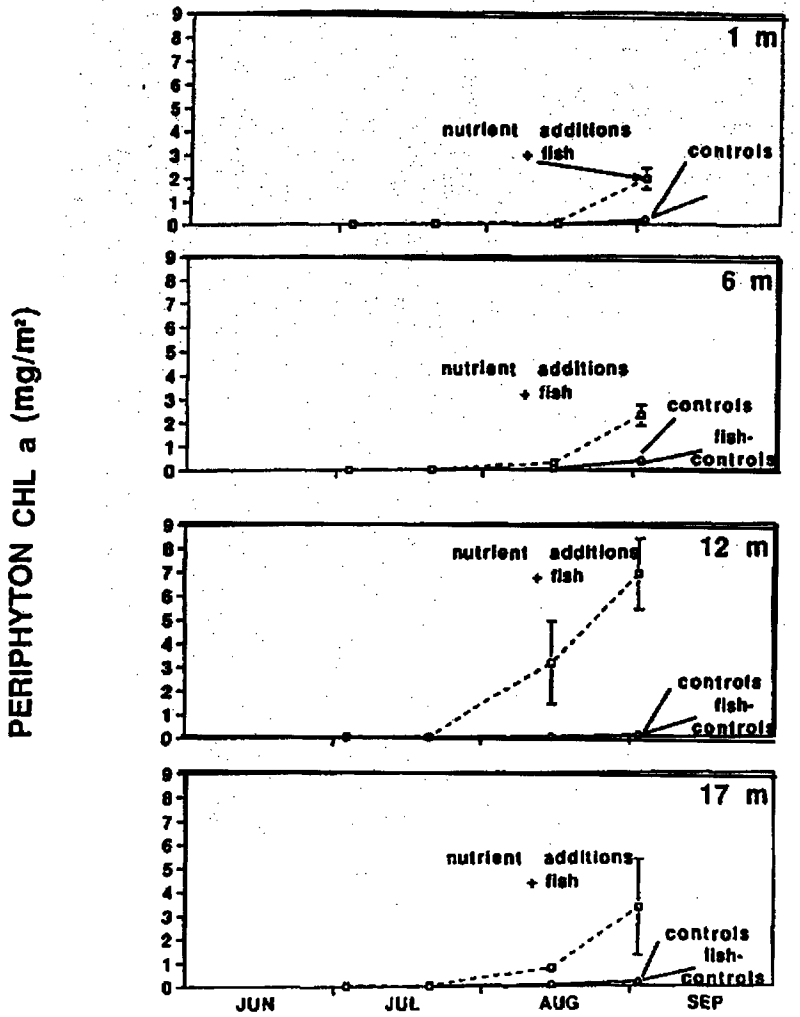


Figure 7. Periphyton chlorophyll a at four for the three treatments shown as the mean of three replicate limnocorrals with 1 standard error bars, 1994.

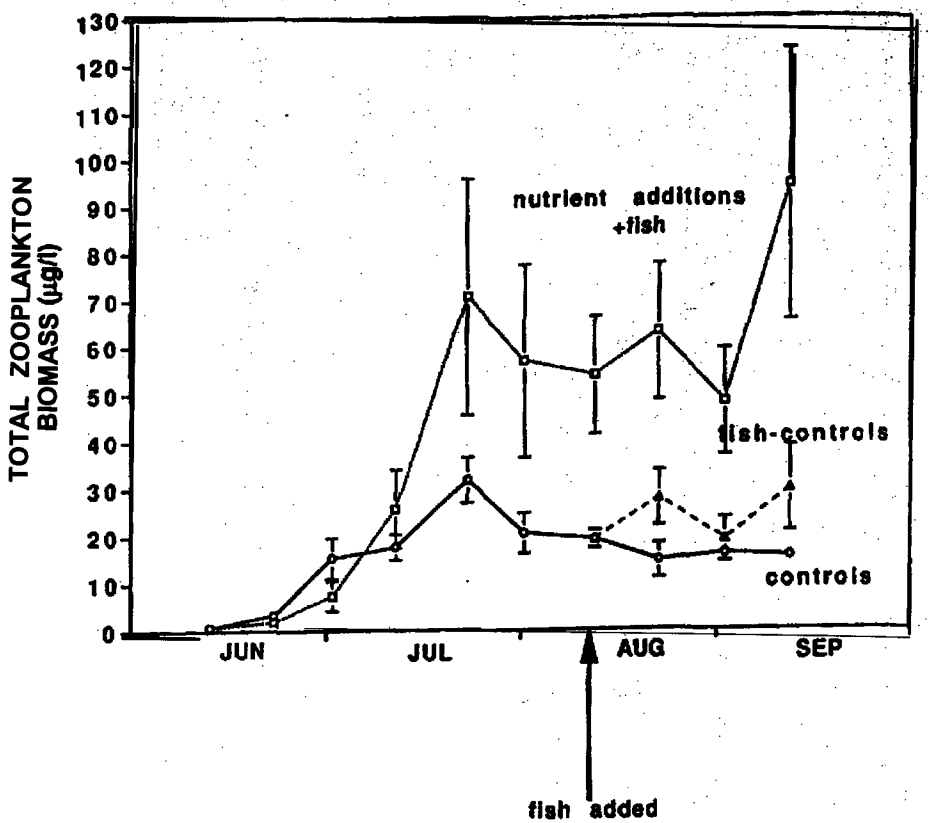


Figure 8. Total zooplankton biomass for the three treatments shown as the mean of three replicate limnocorrals with 1 standard error bars. The arrow points to the date the fish were added, 13 Aug 1994.

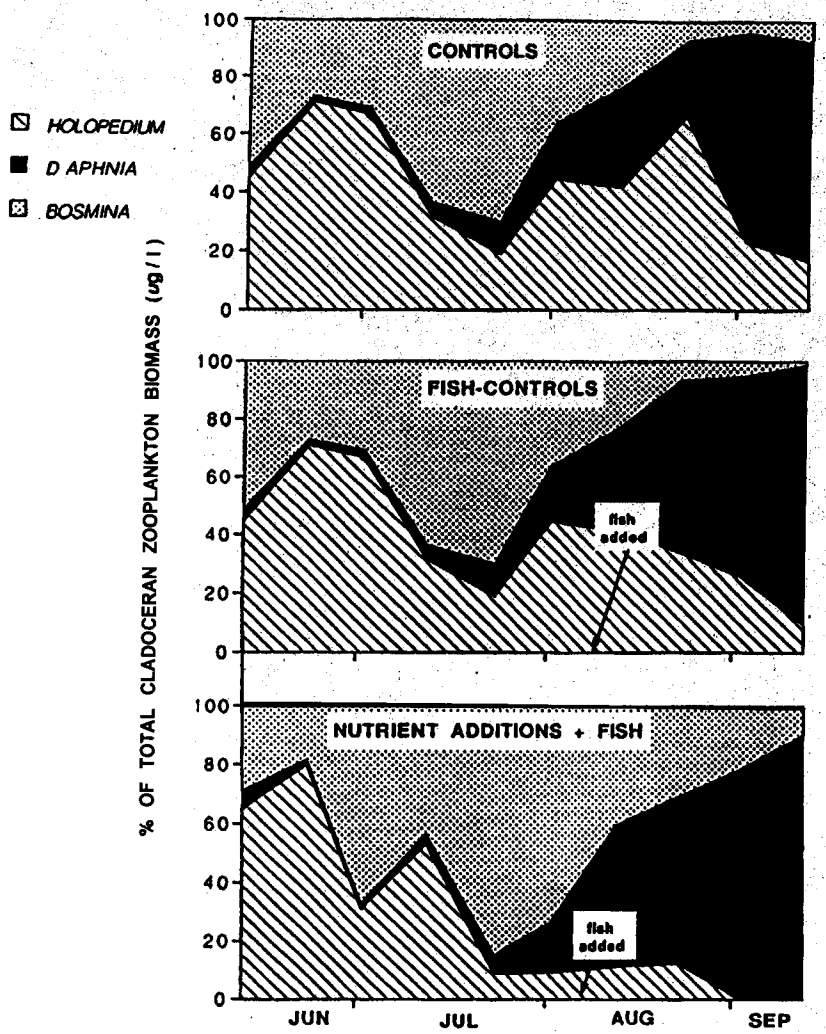


Figure 9. Cladoceran zooplankton species composition in the three treatments for the duration of the experiment. The three replicate limnocorrals are averaged for each treatment, and data are shown as percent of total biomass. Arrows point to the date fish were added, 13 Aug 1994.

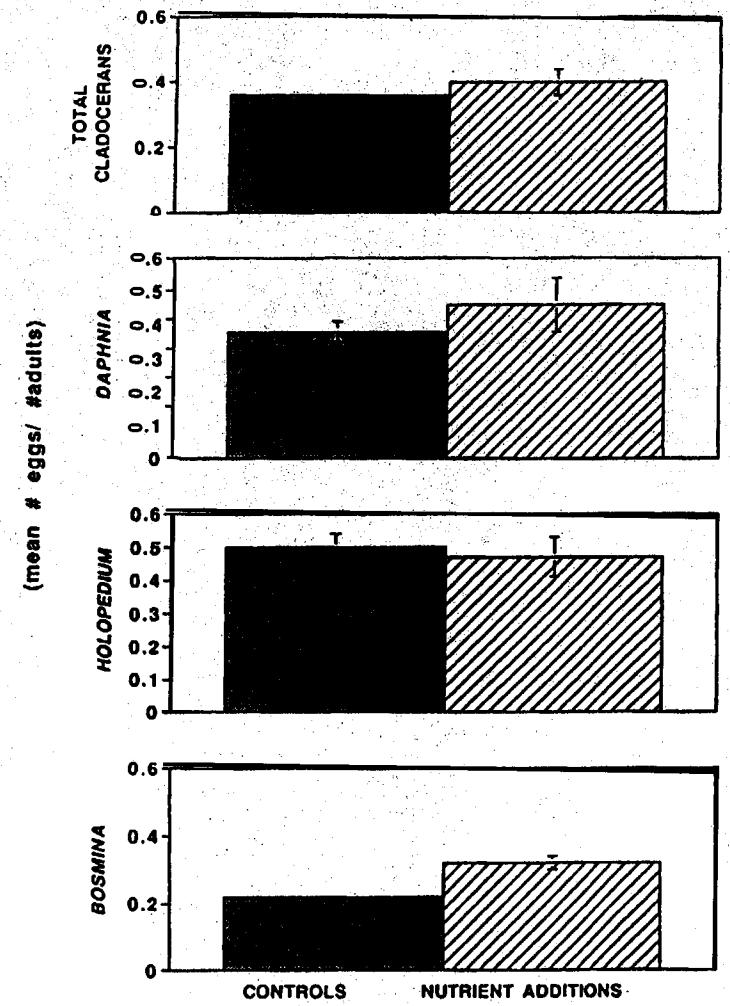


Figure 10. Cladoceran zooplankton egg ratios shown as the average number of eggs per female in the nutrient + fish additions (diagonal dashed bars) versus the controls (shaded bars). Total cladoceran egg ratios are shown at the top, and egg ratios of individual species are shown below.

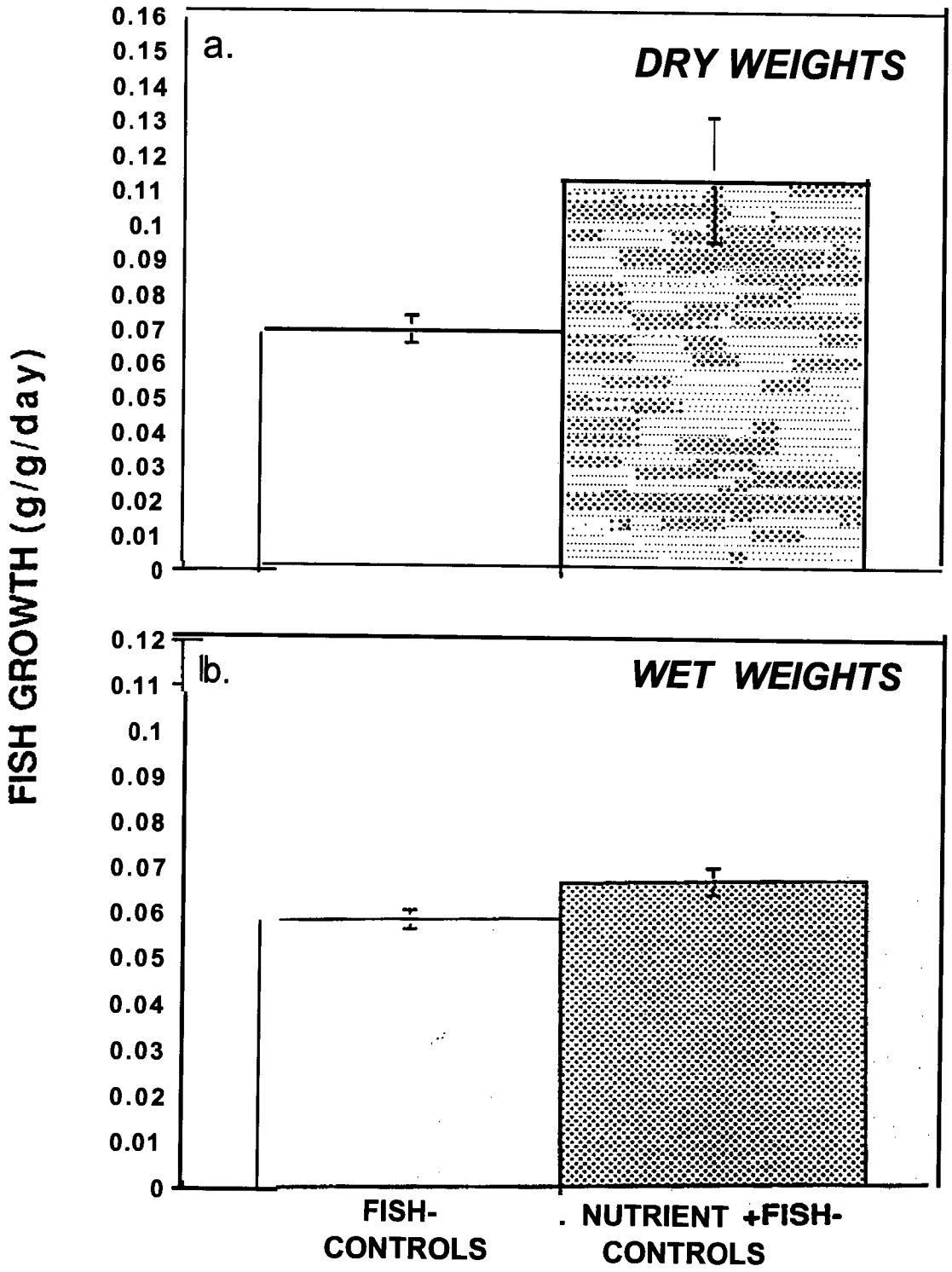
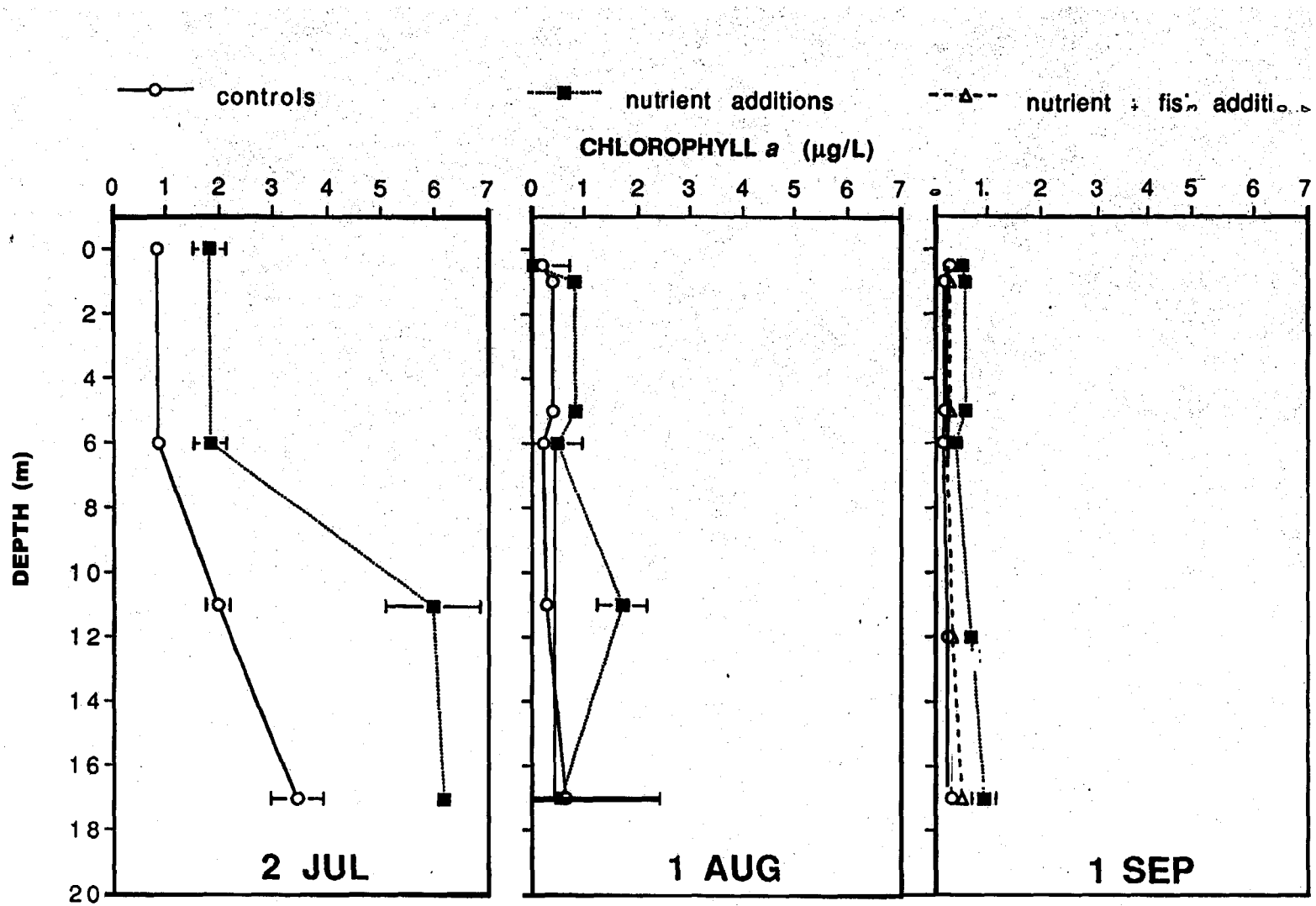


Figure 11. Final fish growth in the nutrient + fish addition treatments (stippled bars) versus the fish-controls (open bars) foi dry weights (g) at the top (a.) and wet weights (b.) at the bottom.

6-O m	CONTROLS		NUTRIENT ADDITIONS	
	MEAN	*1 SE*	*MEAN*	*7 SE*
total P	9.4	*0.2*	*10.9*	*0.3*
soluble reactive P (SRP)	*2.1*	*0.1*	*1.7*	*0.1*
TKN nitrogen	*61.9*	*2.4*	*70.2*	*4.0*
NO, + NO,	*1.8*	*0.3*	*2.8*	*0.3*
N:P	*6.8*		*6.8*	
17-O m				
total P	11.0	*0.5*	*13.3*	0.4
soluble reactive P (SRP)	*2.3*	*0.1*	*1.1*	0.1
TKN nitrogen	*52.5*	*3.3*	*66.0*	*10.3*
NO, + NO,	*1.1*	0.1	*5.4*	*0.6*
N:P	*5.8*		*5.4*	

Table 1. Limnocorral means for duration of the experiment in the nutrient and control (including fish-controls) treatments for all measured nutrients and for the N:P ratio. All values are in µg/l {except the N:P ratio}, and 1 standard error bars are provided.

Appendix 1. Depth profiles of chlorophyll a concentrations (μg/l) in the Redfish Lake limnocorrais at three dates during the experiment, 1994.


controls
nutrient additions
nutrient + fish additions
CHLOROPHYLL a (µg/L)
DEPTH (m)
0 1 2 3 4 5 6 7
0 2 4 6 8 10 12 14 16 18 20
2 JUL
1 AUG
1 SEP

CHAPTER 3

Winter Limnology and Ecology of Kokanee Salmon, *Oncorhynchus nerka,* in the Sawtooth Valley Lakes

Geoffrey B. Steinhart

and

Wayne A. Wuttsbaugh

INTRODUCTION

In order to reach the ocean, juvenile endangered Snake River Sockeye salmon (Oncorhynchus nerka) need to swim over 1400 Km and past seven dams. To complete this arduous journey, the young salmon must be healthy enough to swim great distances, to undergo smoltification, and to avoid predators. Rondorf et al. (1985) found that chinook smolts migrating down the Columbia River completely exhausted their lipid energy reserves in only 400 Km. In addition, fish beginning the journey with higher lipid contents had higher downstream survival The exact cause for the lower survival is not known but weak fish are more susceptible to disease and cannot avoid predators as well as healthy fish.

Lipid levels are not only important for downstream survival, but also influence the timing of seaward migration. In Atlantic salmon, high lipid levels lead to early maturation of males (Metcalfe and Thorpe 1992, Simpson 1992). Salmon with low fat levels, however, often delay their migration until the following year (Skilbrei 1991, Thorpe 1987).

The Snake River sockeye salmon generally spend one to two years rearing in the Sawtooth Valley Lakes before migrating to the ocean in April and May (Bjornn et al. 1968, Teuscher and Taki 1994). Since this migration occurs in early spring, the winter could be an important period for sockeye to build up or maintain ample lipid reserves prior to leaving the lake.

The winter is often thought of as a dormant period for many fish; however, sockeye salmon remain active at cold temperatures (Brett 1983, Egger 1981, Ruggerone 1992, 1993). Even at 4°C sockeye are active and require food, so the winter is unlikely a period of donnancy. In the oligotrophic Sawtooth Valley Lakes, low summer zooplankton densities become even more depressed during the long winter months (See Chapter 1, Figures). In addition diel migrations of zooplankton and very low light levels under-ice result in even less food being available to young salmon. Stocking of juvenile sockeye salmon into the Sawtooth Valley Lakes could further deplete food resources in the lakes. Low food levels could result in reduced growth or lipid levels, and thus decrease downstream survival of juvenile 0. nerka headed for the ocean.

The purpose of our winter work presented here is to answer the following questions: 1) How much do salmon grow over-winter? 2) Do lipid levels change over the winter? 3) How much zooplankton food is available for sockeye salmon during the winter months? and 4) Wbat food resources do salmon utilize during the winter?

Methods

Limnology

From October through May, sampling occurred at approximately monthly intervals on Redfish, Stanley and Alturas Lakes. Pettit Lake was sampled less often, with only two under-ice sampling dates. At each lake we sampled temperature, oxygen, light intensity, water chemistry, chlorophyll a, and zooplankton (see Chapter 1 for complete limnology methods).

In addition to routine sampling, diel zooplankton sampling was performed on Stanley Lake on December 17, 1993. Zooplankton were sampled at two different stations at four times during the day: dawn (0800), mid-day (1300), dusk (1730), and mid-night (0000). Zooplankton tows were performed at discrete depth intervals as follows: 0-2 m, 2-5 m, 5-10 m, 10-17 m, and 17-24 m. Samples were preserved in a buffered sugar formalin solution for later analysis. The zooplankton diel data were analyzed using a 2-factor repeated measures ANOVA with time and depth as the two factors.

Fish Sampling

Kokanee (0. nerka) were used as a surrogate species to study changes in diet, growth and lipid levels of the endangered sockeye salmon. Kokanee are the same species as sockeye, but they do not migrate to the ocean as smolts. Prior to migration, juvenile sockeye and kokanee are functionally similar. Salmon were only sampled in Alturas and Stanley Lakes. These lakes represent the extremes in productivity of the Sawtooth Valley Lakes.

Late in the fall, and early in the spring, a mid-water trawl was used to capture kokanee salmon for our diet and growth studies. During periods of ice-cover on the lakes, multi-panel horizontal gillnets, as well as small mesh vertical gillnets (12, 20, and 32 mm stretch mesh) were set to catch young salmon. The horizontal gillnets were set by stringing the nets through a series of holes cut through the ice. Horizontal gillnets were set on the bottom of each lake, while vertical gillnets were lowered from the surface to a depth of 25 m, through a single large hole cut in the ice. The gillnets were originally checked several times a day, but as catch rates proved negligible during daylight hours, the nets were subsequently checked only at dawn.

Fish were kept in a cooler until returning to the laboratory, where the fish were weighed (± 0.01 g) and measured (± 1 mm). Gut contents were taken either by dissecting the stomach and removing its contents, or by flushing out the digestive system using a small syringe., Gut contents were preserved in 95% ethanol for subsequent identification. Otoliths were removed from the fish and saved for later fish aging. The fish were then frozen for prior to lipid analysis.

In the laboratory, the 'dry weight of each fish was measured after drying the fish for 24 hours at 60°C. To measure lipid levels in the salmon, we used a Soxhlet lipid extraction (Bligh and Dyer 1959) using petroleum-ether as a solvent. Petroleum-ether does not dissolve the polar lipids that make up cell structure (Dobush et al. 1985), and thus extracts only lipids used as energy stores. The dried fish were extracted for 24 hours, and the weight of the extracted fat was measured. -Changes in mean weight and lipid content were compared using t-tests to see if there was a significant diierence between mean values from November and May.

To measure the vertical distribution of kokanee in the lakes we used a combination of hydroacoustic sampling and vertical gillnets. Hydroacoustic measurements were made using Lowrance X-1 6 chart recorders. By setting the

recorders on the lowest speed, a single roll of chart paper would last for approximately 24 hours. In February 1994, a single 45° downward pointing transducer was used to sample fish distributions in Redfish and Alturas Lakes. In March 1994, two transducers and two chart recorders were used on Stanley Lake. Smce transducers cannot detect targets within O-2 m of the transducer, a 45" transducer was suspended 10 m below the ice in order to effectively sample fish targets immediately below the surface. In addition, a downward-pointing 20" transducer was also used on Stanley Lake. Vertical gillnets were set to identify targets and to supplement the vertical distribution data.

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 (Iviev 196 1, Lechowicz 1982).

$$E = ri\text{-}pi \,/\, ri+pi$$

where ri is the proportion of prey found in the digestive tract and pi is the proportion that prey group in the lake. Electivities were calculated using frequencies of the diierent taxa in the whole water column.

As visual predators, salmon depend on light to see and capture their prey, but low tight penetration due to ice and snow could limit kokanee capture success under-ice. To determine what proportion of the water column was available for 0. nerka feeding, we calculated the capture success at each depth using data from light profiles to determine a profile of light intensity. Then, using data on sockeye capture success at different light intensities (Ah 1959), we were able to predict capture success at each depth. Depth stratified zooplankton samples were taken to determine what proportion of prey were available in different strata. Prey electivities were calculated to see which zooplankton the kokanee were preferentially feeding on.

Results

A total of 172 kokanee were caught in Stanley Lake from November ,&rough May, while in Alturas Lake, only 43 kokanee were caught during the same period. There were two distinct modes in the distribution of fish lengths in Stanley Lake (Figure 1). Although there appears to be little change in total length over the winter, this may be biased because gillnets are not as efficient at catching the smallest fish as the mid- water trawl. The lengths of fish in Alturas Lake are more evenly distributed (Figure 2), but the sample size was small because of the low densities of 0. nerka in Alturas Lake (Teuscher and Taki 1995, Figure 3). Kokanee in Alturas Lake did not grow in total length over the winter.

Otolith aging of fish from Stanley and Alturas Lakes has been diicult. In Stanley Lake, however, it appears that most kokanee under 150 mm total length (TL) are age- 0+ fish (D. Teuscher, pers. comm.). For the purposes of this study, we treated all fish in Stanley Lake under 150 mm TL as age-O+. In Alturas Lake, age data is even more confusing, so the data presented in this report represents fish under 150 mm TL, but not necessarily age-0+ kokanee.

Kokanee in Stanley Lake grew little over the course of the winter (Figure 3). In November, age-O+ kokanee averaged 11.9 g, while in May they averaged 12.3 g. This difference was not significant (t=0. 15, df=8, p>>0.25) due to high variability in weights of age-0+ kokanee on all sampling dates. Dry weights in November averaged 3.15 g and decreased to 2.2 g in May, but the change was also not significant (t=1.19, df=8, p>0.1).

Lipid levels in kokanee caught in Stanley and Alturas Lakes did decrease from November to May (Figure 4). This decrease in percent lipid was significant for both lakes (Stanley: t-2.45, df=8, p=0.02; Alturas: t=2.1, df=8, p<0.05). Stanley Lake kokanee had a higher mean percent lipid than did Alturas in November (7.1% and 6.2 %, respectively); however, by May, Stanley kokanee had a lower mean percent lipid than did fish from Alturas (4.3 % and 4.8 %, respectively).

When age-O+ kokanee were split into diierent size classes, it becomes apparent that the larger kokanee had higher percentages of lipid in their tissue than did small fish (Figure 5). Both the 100-125 mm and 125-150 mm size classes of kokanee showed a drop in lipid levels. There was little change in lipid content of the smallest kokanee (<100 mm), but this was likely an artifact of the small sample size.

There is a high correlation between percent water and percent lipid in kokanee caught in both Alturas and Stanley Lakes (Figure 6). The equations of the two lines differed slightly, possibly due to diierent sample sizes or the inclusion of fish older than age-O+ in the Alturas Lake data.

Kokanee salmon in Stanley, Alturas, and Redfish Lakes did not appear to undergo. a diel vertical migration under the ice (Figure 7). In Alturas and Redfish Lakes,. hydroacoustic targets ranged from 3 m to 40 m. Most targets, however, were between 5 m and 20 m, with a mean depth of 12 m. In Stanley Lake, targets were detected from 2 m to 23 m deep, with most of the targets being around 17 m deep. After correcting for sampling volume, however, a large proportion of the kokanee population was found to be in the 2-5 m depth strata (Figure 8). There appeared to be little or no difference in daytime and nighttime depth distributions in Stanely. Kokanee capture depths from the vertical gillnets did not support the hydroacoustic results (Figure 9). In March, no kokanee were caught above 12 m, even though the hydroacoustics showed that over 60% of the population was above 12 m. This disparity in results is likely an artifact of a small sample size, but for the acoustics and the gillnet data. In April, most fish were captured deep in Stanley Lake, but by May, a large proportion of fish were captured near the surface.

In mid-December, kokanee salmon in Stanley Lake ate mostly calanoid copepods (Teuscher and Taki 1994, Appendix 2A), even though Bosmina sp. were twice as abundant as calanoids in the lake. In fact, it kokanee salmon had a negative electivity for both Bosmina and Daphnia, but a positive electivity for calanoids (Figure 10). Cyclopoid copepods had a nearly neutral electivity on this date.

Both Bosmina and Daphnia underwent a significant diel vertical migration on December 17, 1993 in Stanley Lake, while calanoid and cyclopoid copepods did not

migrate as strongly (Table 2). During the day, calanoid copepods had the highest mean depth of all the zooplankton taxa (Figure 11), and nearly 80% of the calanoid copepods *were* at light intensities where kokanee have at least a 50% capture success (Figure 12). For Bosmina, on the other hand, approximately 50% of the population was at depths where the light intensity reduced kokanee capture success below 50% during the daylight hours (Figure 13). At night, however, Bosmina moved much higher in the water column.

Discussion

The length and wet weights of kokanee in both Stanley and Alturas Lakes did not change significantly during the winter months. In Stanley Lake, however, the kokanee show a significant decrease in percent lipid. This suggests that despite cold temperatures, there is not an abundant supply of food available for salmon during the winter in Stanley Lake. In Alturas Lake, the kokanee also experienced a significant decrease in lipid content; however, the decrease was not as large as in Stanley Lake.

Due to the large variation in the sizes of age-0+ fish caught in Stanley Lake on any particular date, it is not reasonable to conclude that fish do, or do not grow during the winter from length and weight measurements alone. Mean wet weights hardly changed over the winter, while dry weights decreased only slightly. Lipid content, however, did decrease significantly, demonstrating that the fish are losing energy reserves even though their is little change in growth. By measuring the lipid content of fish we can get an accurate measure of the amount of energy reserves a fish has at its disposal. There is a strong correlation between the percent water and percent lipid content of kokanee. This relationship can be used to make estimates of lipid content without performing expensive and time-consuming fat extractions.

Extremely low winter zooplankton populations probably resulted in the loss of lipid in kokanee salmon. During the tinter, Alturas Lake has a higher density and biomass of zooplankton than does Stanley Lake (Table 1, see also Chapter 1). This more plentiful food may have resulted in the smaller loss of lipid reserves from Alturas Lake. This again suggests that the salmon in Stanley and Alturas may be food limited during the winter months.

Zooplankton density and biomass were lower during winter than summer for all the Sawtooth Valley Lakes, except Alturas Lake (Table 1, see also chapter 1). In general, large cladocerans disappeared first, followed by copepods. Nauplii and Bosmina were present throughout the winter the latter makes up most of the crustacean zooplankton biomass in Alturas Lake (Chapter 1).

Not only were zooplankton densities much lower during the winter than in the summer, but decreased light intensities under-ice appear to provide a refuge for some zooplankton where kokanee cannot effectively feed *on* them. In Stanley Lake, Bosmina undergo diel vertical migrations, presumably to avoid predators (Clark and Levy 1988, Stich and Lampert 1981). Snow and ice block so much of the light that kokanee likely have diiculties locating and capturing zooplankton at

deeper depths. This is supported by the fact that even though Bosmina were quite abundant in Stanley Lake in December, they did not make up a large fraction of the kokanee diet. In contrast, calanoids, which did not retreat to the darkened depths of Stanley Lake, were a significant component of kokanee diets and were selected for on that date. Stanley Lake is the shallowest of the four main lakes in the Sawtooth Valley. In the deeper lakes, even more of the water column will be too dark for effive kokanee feeding and will provide an even larger refuge for zooplankton.

In conclusion, the winter is likely a crucial period for juvenile 0. nerka growth. In preperation for their long seaward migration. Low lipid levels can result in lower survival of migrating smolts, a problem that is compounded by the addition of several dams along the Snake and Columbia Rivers. This decline in lipid levels is likely due to the extremely deficient zooplankton populations in the Sawtooth Valley Lakes during the winter. Ice and snow cover compound this problem by reducing the amount of light available for the visually feeding kokanee. As a result, kokanee in Alturas and Stanley Lakes lose valuable lipid resources during the winter. By stocking more fish into these lakes, we could potentially reduce an already limited food resource, resulting in even greater lipid losses.

It should be noted, however, that Pettit Lake has a higher biomass and density of crustacean zooplankton during the winter than all the other study lakes. Therefore, Pettit Lake may provide a system where sockeye salmon can preserve, or even increase lipid levels prior to migration. In addition, lake fertilization has been shown to increase sockeye smolt production (Hyatt and Stockner 1985, LeBrasseur et al. 1978). The fertilizing of Redfish Lake in 1995 could greatly improve its potential as a rearing lake for young sockeye. Stanley and Alturas Lakes, however, have relatively low zooplankton numbers and are probably not as suitable for producing increased numbers of juvenile 0. nerka.

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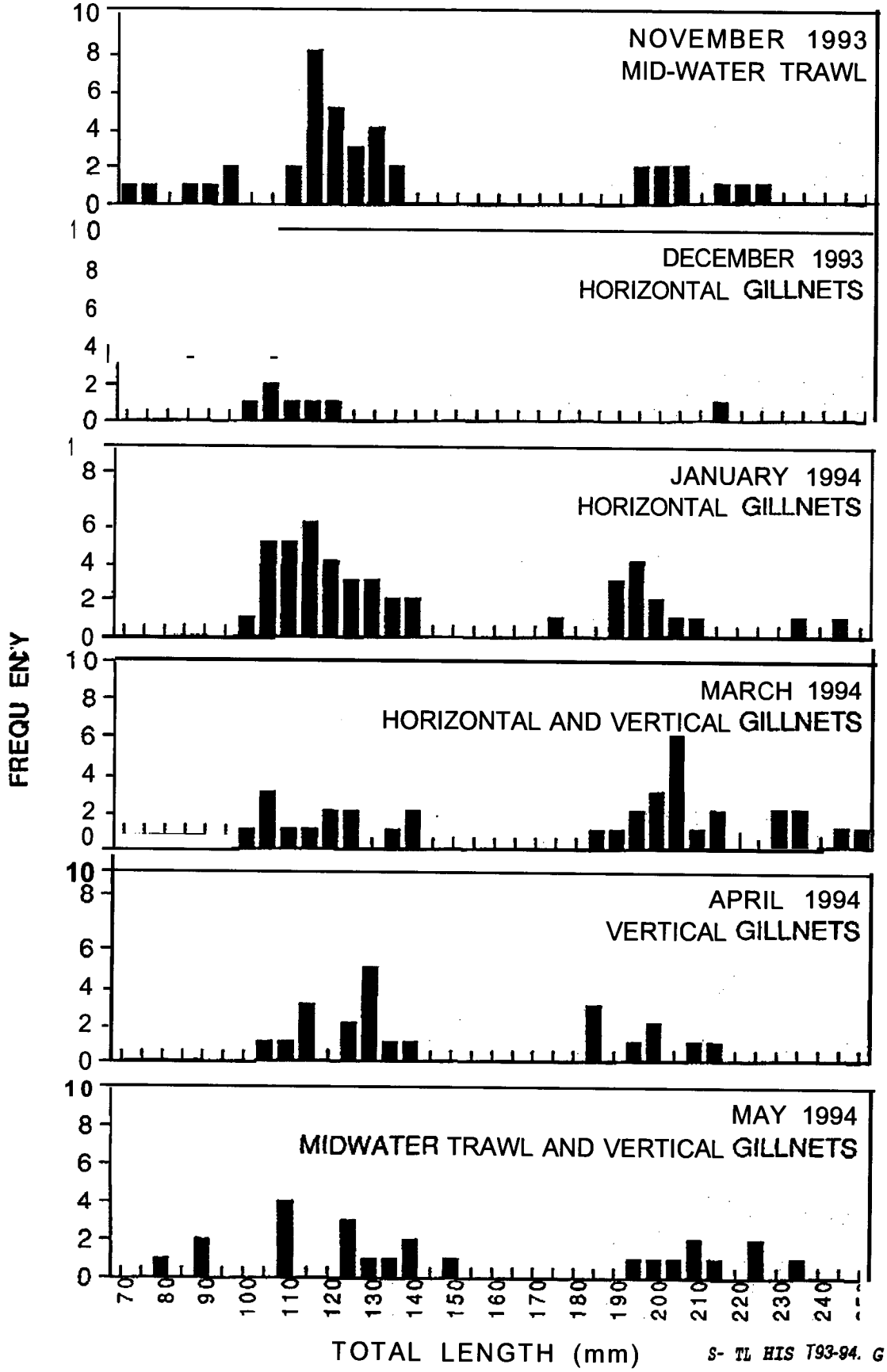

Figure 1. Total length-frequency histograms for kokanee caught in Stanley Lake in 1993-94.

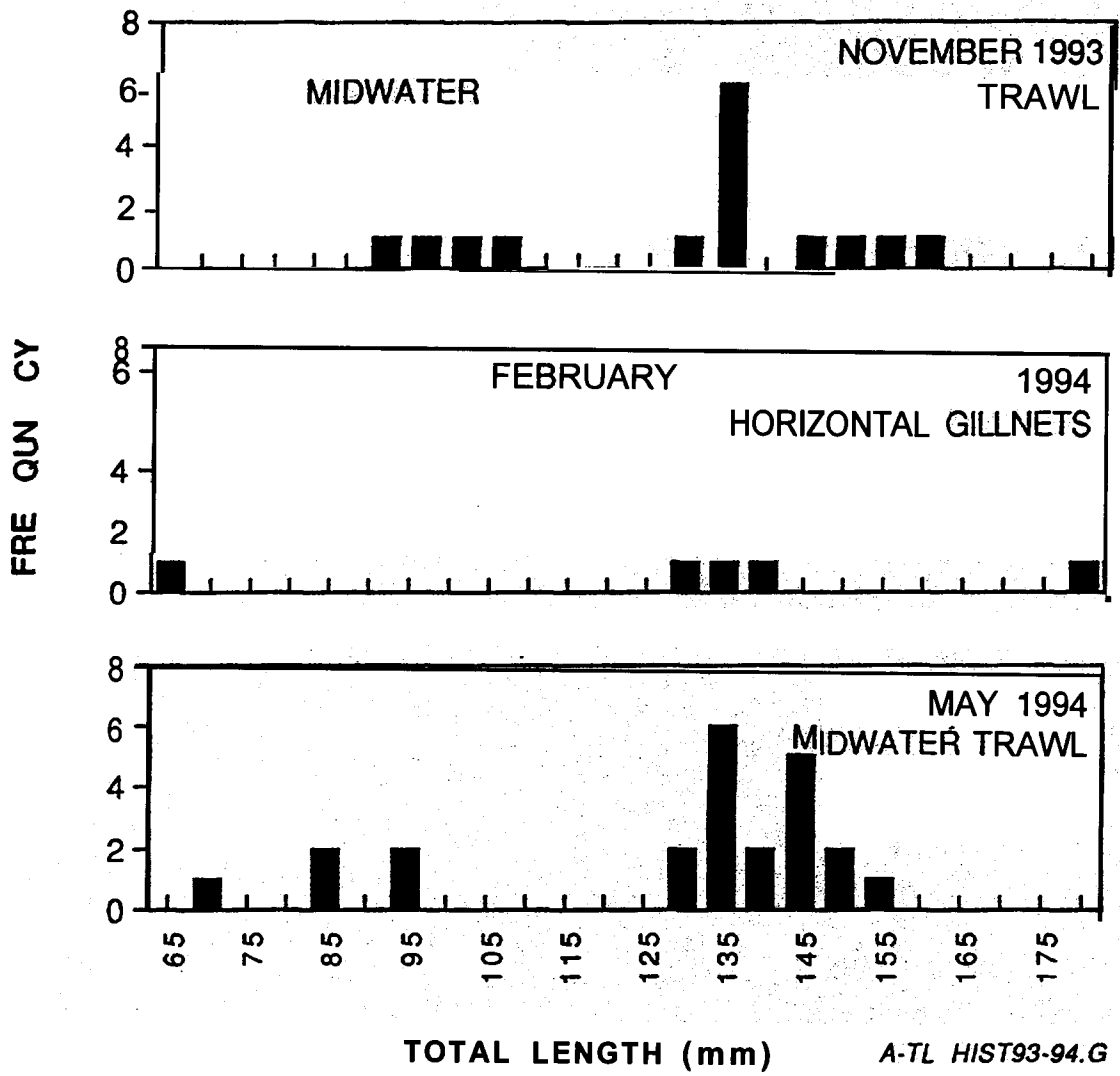

Figure 2. Total length-frequency histograms for kokanee caught in Alturas Lake in 1993-94.

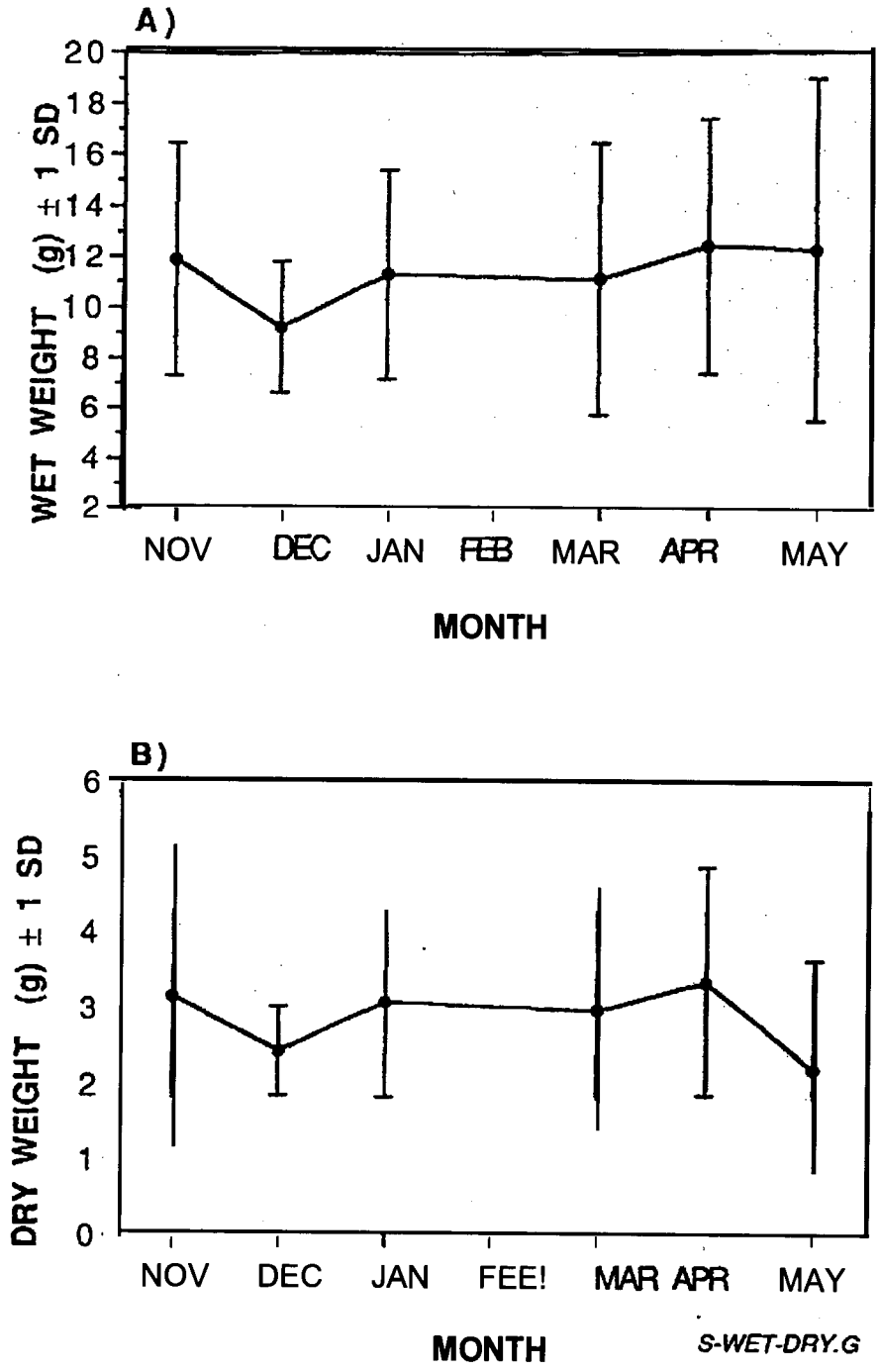

Figure 3. A) Mean wet weights and B) mean dry weights for kokanee caught in Stanely Lake in 1993-94.

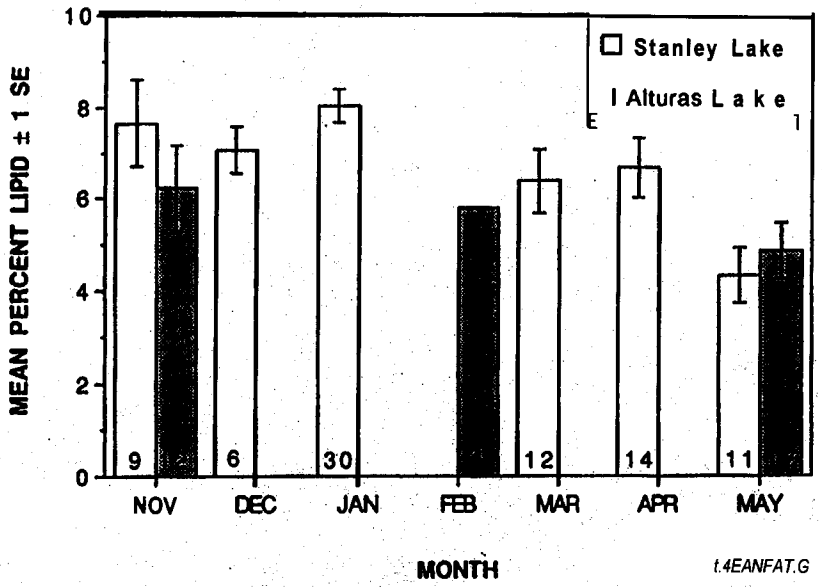

Figure 4. Mean lipid as a percent of wet weight for kokanee caught in Stanley and Alturas Lakes in 1993-94. The sample size is indicated at the base of each bar.

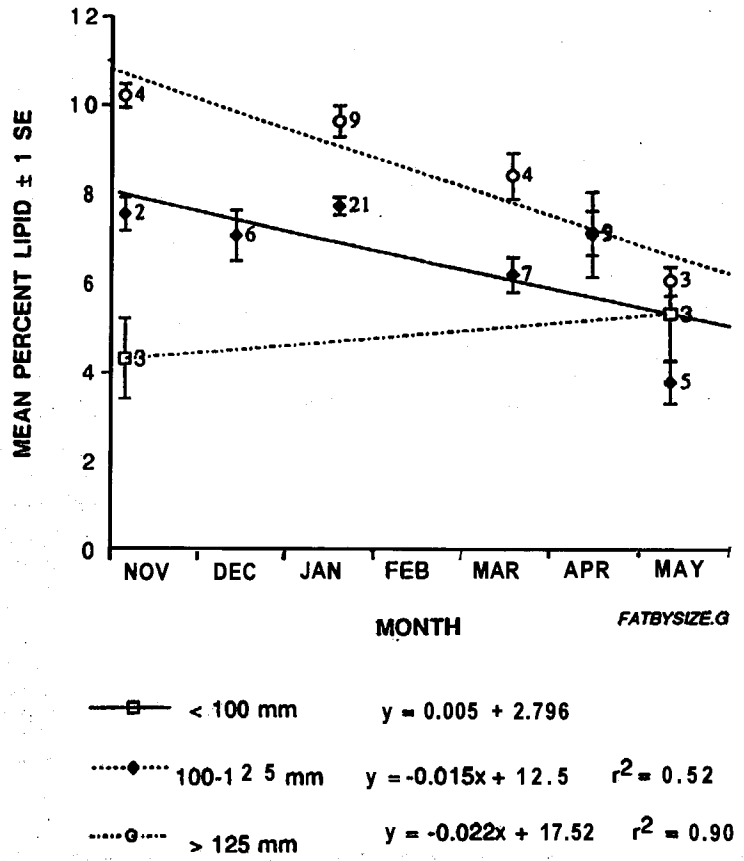

Figure 5. Mean lipid as a percent of wet weight for three size classes of kokanee caught in Stanley Lake in 1993-94. The sample size is indicated by each point.

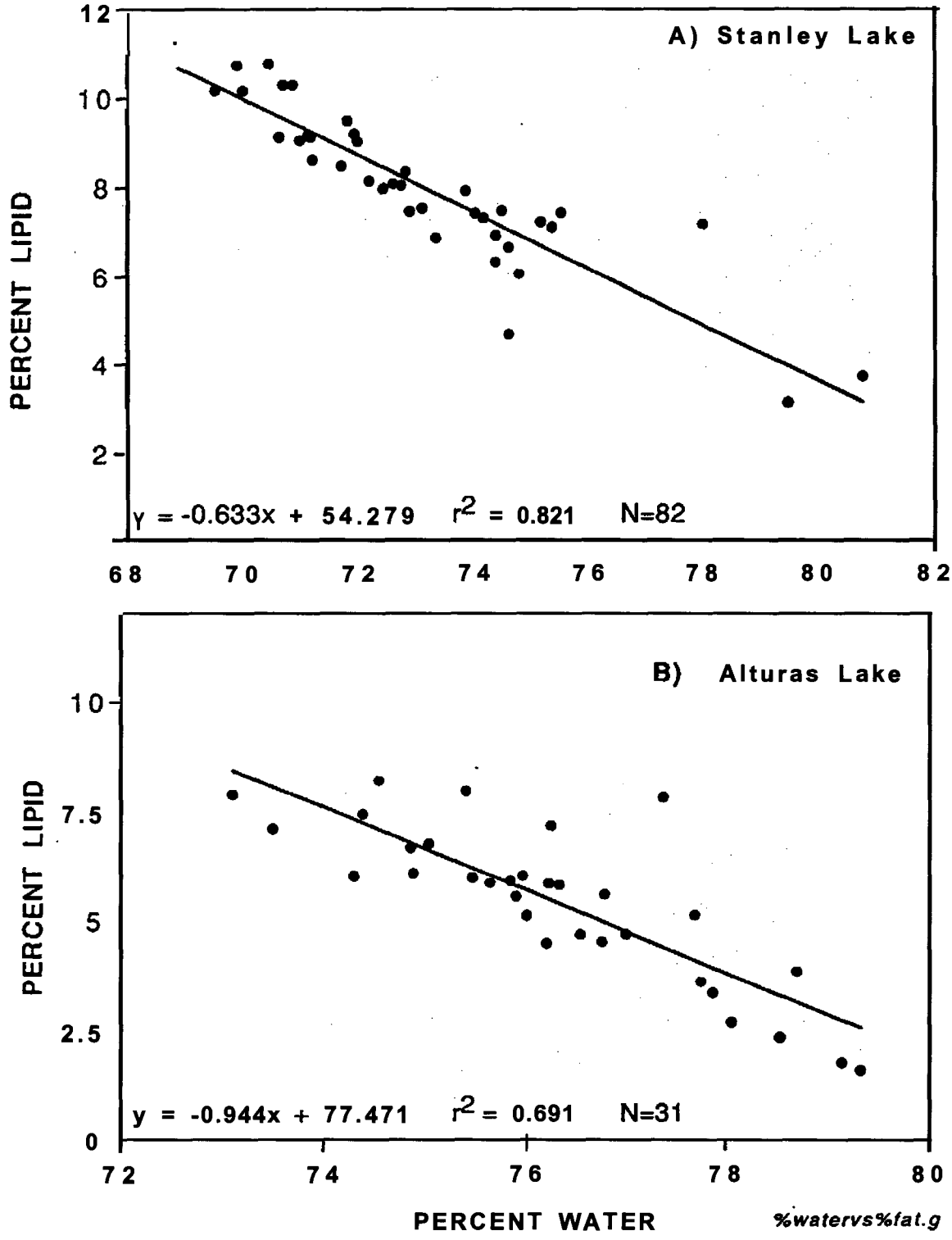

Figure 6. Relationship between percent water and percent lipid for kokanee from: A) Stanley Lake and B) Alturas Lake.

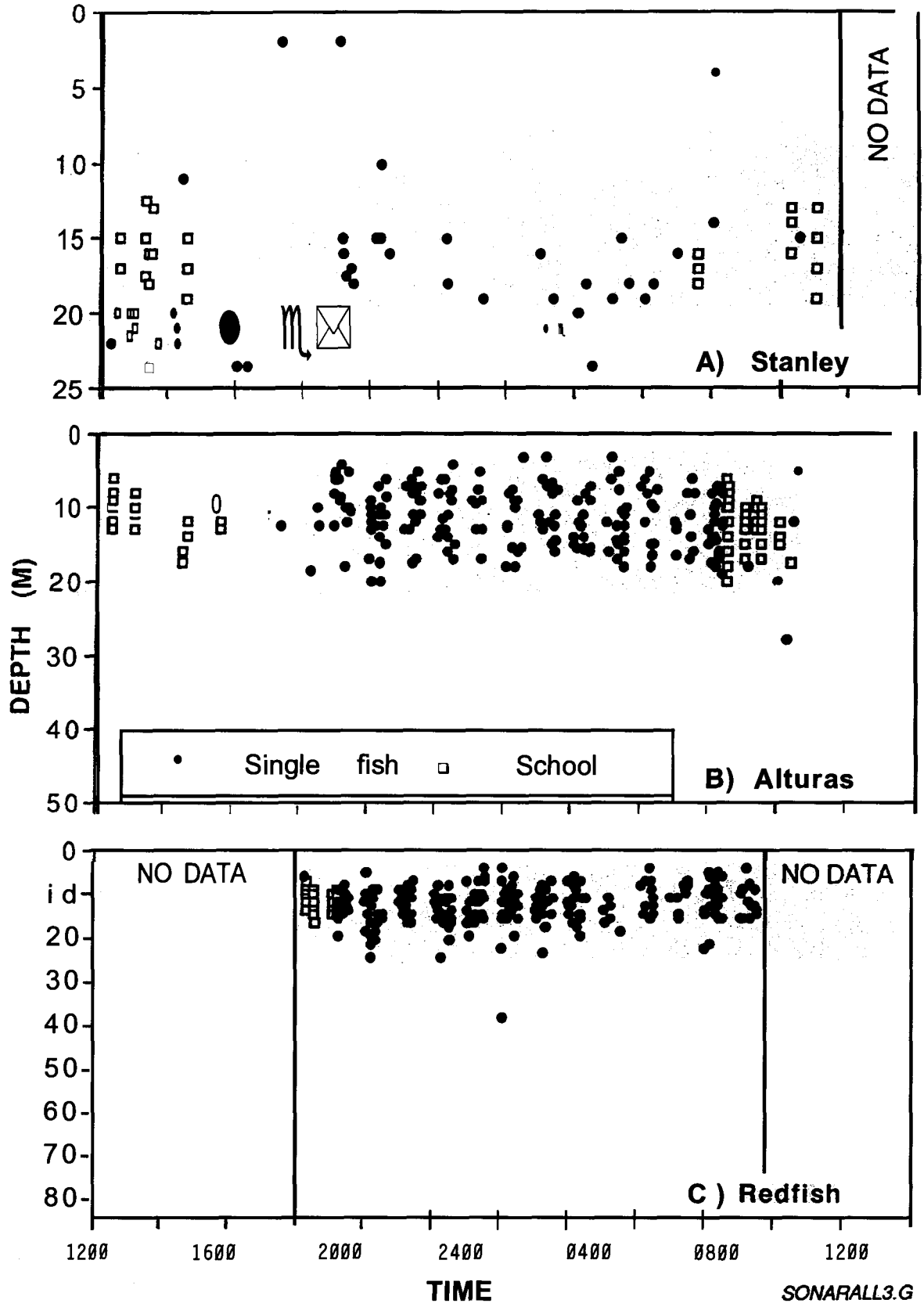

Figure 7. Kokanee vertical distribution from 1994 hydroacoostic data for: A) Stanley (27-22 March), B) Alturas (17-18 February), and C) Redfish (19-20 February). These data represent all targets and are not corrected for sampling volume. Note the different target types and depth scales.

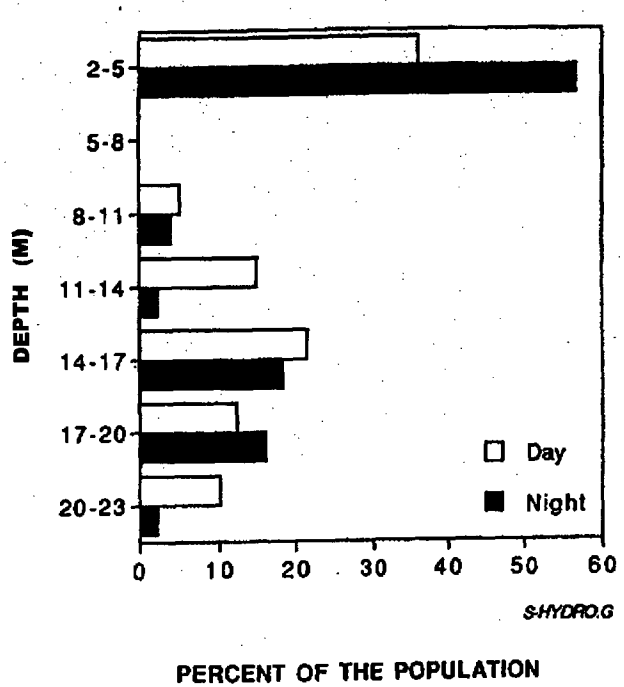

Figure 8. Kokanee diel vertical distribution in Stanley Lake on March21-22, 1994. These data give the percent of the kokanee population in different depth strata and are corrected for sampling volume.

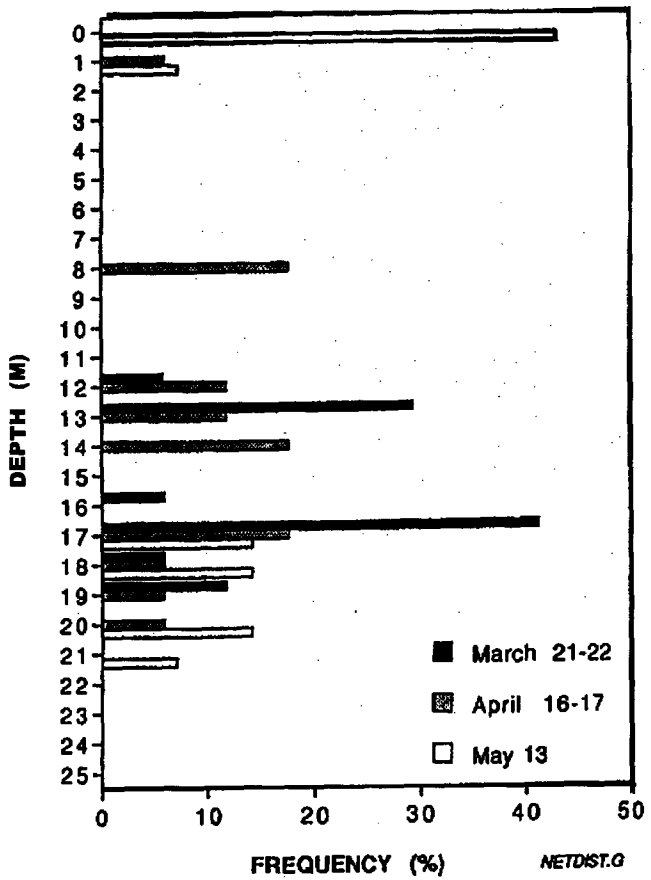

Figure 9. Kokanee vertical distribution from 1994 vertical gillnet data for Stanley Lake.

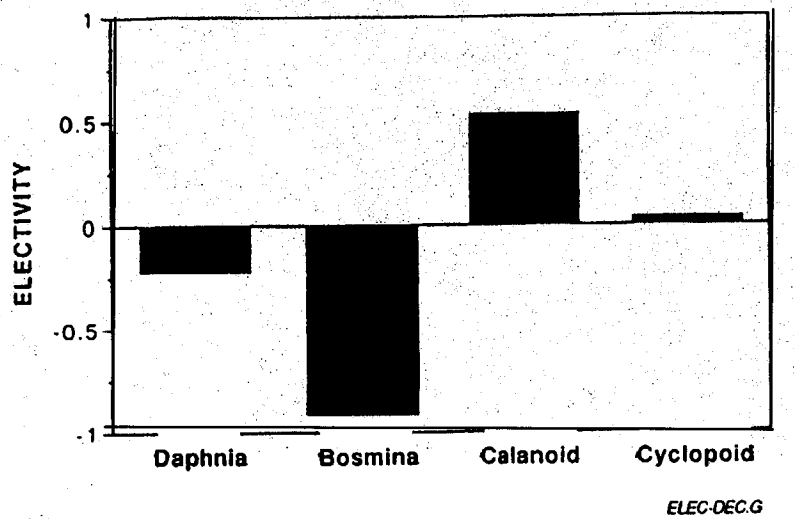

Figure 10. Kokanee electivities (o r different zooplankton prey in Stanley lake o r December 17, 1994. A value of 1 means that prey item is selected for, while a value of -1 means that prey item is n o t available or is avoided.

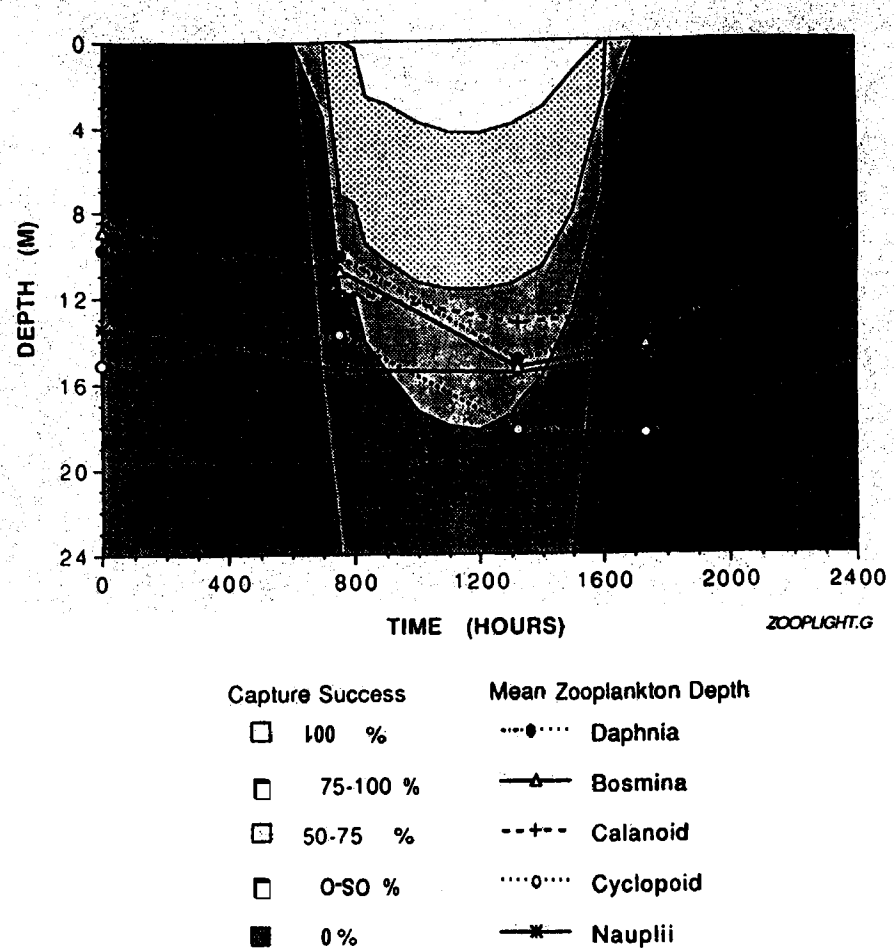

Figure 11 Mean d e p t h of different zooplankton taxa plotted over a c t i v e capture success isopleths for Stanley Lake diel sampling on December 17, 1993. Kokanee capture success is based on light intensity and data from Ali (1959).

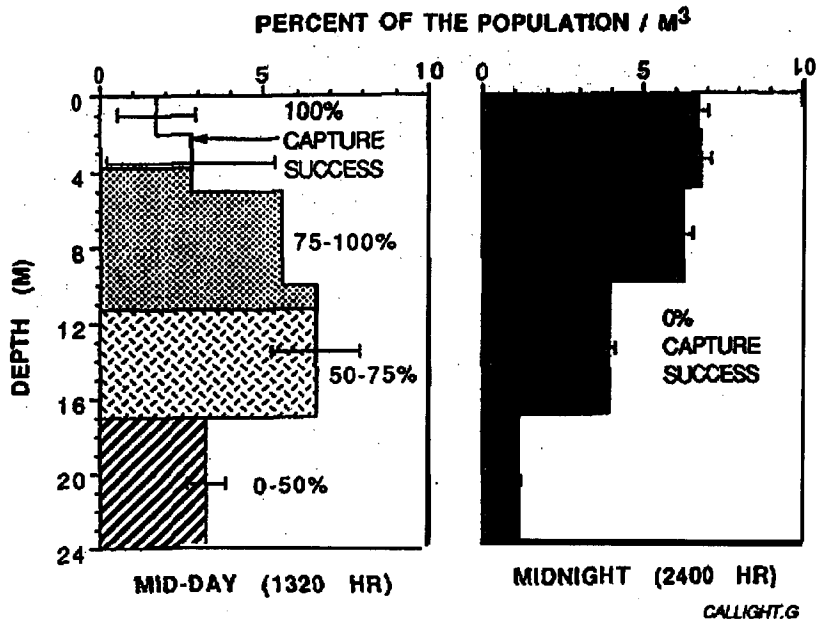

Figure 12. Diel vertical migration of calanoid copepods plotted with kokanee capture success rates based light intensity. Kokanee capture success is from Ali (1959).

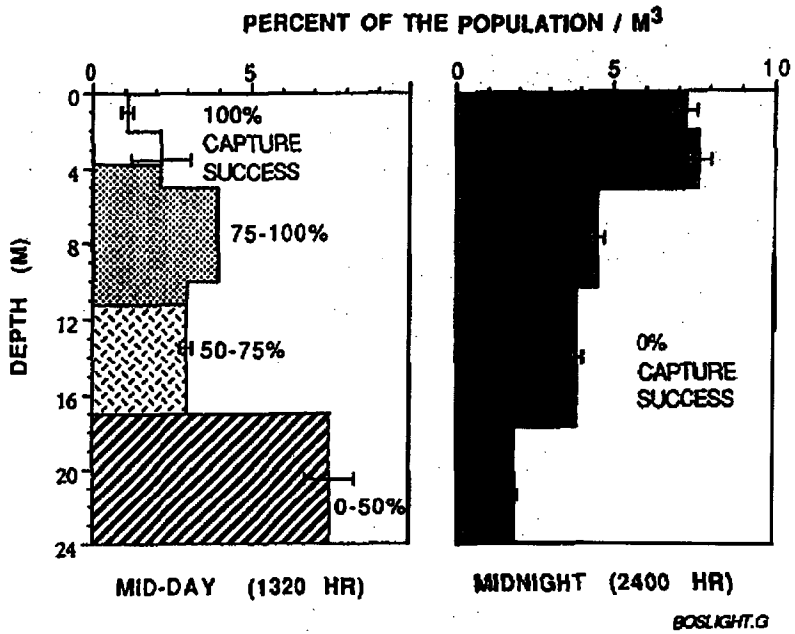

Figure 13. Diel vertical migration of bosmina plotted with kokanee capture success rates based light intensity. Kokanee capture success is from Ali (1959).

Table 1. Mean zooplankton biomass and density for the Sawtooth Valley Lakes during the summer and winter periods of 1994.

	May - November		December - April	
	mg/m²	µg/L	mg/m²	µg/L
Redfish	710	8.4	170	2.0
Pettit	140	2.9	197	4.6
Stanley	1465	24.1	390	8.1
		18.3	77	3.3

Table 2. Zooplankton diel migration results from Stanely Lake on December 17, 1993 using a repeated measures ANOVA.

Zooplankton Taxa	df	F-ratio	P-value
Daphnia	12		
Bosmina	12	10.14	0.0002
Calanoid	12	4.55	0.007
Cyclopoid		1.93	
Nauplii	12	2.59	0.06
	12	0.40	0.94

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