SNAKE RIVER SOCKEYE SALMON HABITAT AND LIMNOLOGICAL RESEARCH

ANNUAL REPORT 1993

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PREFACE

In 1990 the Shoshone-Bannock Tribes (SBT) petitioned the National Marine Fisheries Service (NMFS) to list Snake River Sockeye salmon as endangered. As a result, Snake River Sockeye were listed and the Bonneville Power Administration (BPA) began funding efforts to enhance sockeye stocks. Recovery efforts include development of a brood stock program, genetics work, describing fish community dynamics in rearing lakes, and completing limnology studies. The SBT, in cooperation with Idaho Department of Fish and Game (IDFG), are directing fish community and limnology studies. IDFG is managing the brood stock program. The University of Idaho and NMFS are completing genetics work.

Part I of this document is the SBT 1993' annual report that describes findings related to fish community research. Part II is a document completed by Utah State University (USU). The SBT subcontracted USU to complete a limnology investigation on the Sawtooth Valley Lakes. Management suggestions in Part II are those of USU and are not endorsed by the SBT and may not reflect the opinions of SBT biologists.

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EXECUTIVE SUMMARY

In 1993 we completed research directed at characterizing the 0. nerka populations and their interactions with other fish species in five Sawtooth Valley Lakes. Historically, Redfish, Alturas, Pettit, Stanley, and Yellow Belly Lakes provided Snake River sockeye *(Oncorhynchus nerka)* spawning and rearing habitat (Evermann 1896; Bjornn 1968). All of these lakes, with exception to Yellow Belly, still support *0. nerka* populations. In chapter 1 of this report we describe 0. nerka spawning locations and densities, tributary fry recruitment, and results from a habitat survey completed in Redfish Lake. In chapter 2 we review foraging habits of fish that may compete with, or prey on 0. *nerka* populations.

Kokanee fry emergence from Fishhook Creek in 1993 was 160,000. Fry emergence increased nearly five fold over that reported in 1992. Interestingly, spawning densities in 1991 and 1992 were somewhat similar (7,200 and 9,600, respectively). Discharge from Fishhook Creek was markedly higher in 1992 and may have caused the better egg to fry survival. *0. nerka* spawning on sockeye beach appeared limited (< 100 fish). Additionally, sockeye beach was the only area that wild or residual sockeye were located. Of 24 adult sockeye released into Redfish Lake, from the brood stock program, two were found spawning in the south end of the lake. Results from the habitat survey indicated that substrate

composition on sockeye beach is poor.

0. nerka diet patterns shifted from chironomid prey in June to zooplankton prey in September. Rainbow trout consumed a broad range of prey, with few instances of significant diet overlap with *0. nerka.* Northern squawfish, bull char, and lake trout preyed on *0. nerka.* Utilization of 0. *nerka* by predators was greatest in September.

PART ONE:

FISH COMMUNITY AND HABITAT RESEARCH FOR SNAKE RIVER SOCKEYE SALMON IN THE SAWTOOTH VALLEY LAKES

INTRODUCTION

Historically, thousands of Snake River Sockeye salmon returned to the Sawtooth Valley to spawn. Evermann (1896) reported that the Sawtooth Valley Lakes were teeming with red fish. Bjornn (1968) estimated that 4,360 sockeye returned to Redfish Lake in 1955. These numbers no longer exist. In the 1980's, less than 50 Snake River sockeye salmon survived to spawn (Bowler 1990). Since 1990, only 13 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. These lakes (Redfish, Alturas, Pettit, Stanley, and Yellow Belly) provided critical spawning and nursery habitat for Snake River Sockeye salmon. Species introductions, low productivity, changes in spawning conditions, and abundant kokanee populations may have changed the ability of these systems to produce sockeye smolts. Specific research objectives of the SBT for 1993 were to (1) continue monitoring resident 0. *nerka* population characteristics (i.e., monitor spawning densities and locations, complete a spawning habitat survey, and estimate fry recruitment), (2) describe the foraging habits of fish species that may compete or prey on 0. nerka, and (3) complete limnology studies in nursery lakes. Objective (3) was completed by Utah State University, under a subcontract agreement, and is reported in Part II of this document. Our 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.

CHAPTER 1:

0. nerka POPULATION *CHARACTERISTICS*

In 1991, we began researching 0. *nerka* population characteristics in the Stanley Basin. Currently, four of the five large Stanley Basin lakes contain 0. *nerka* populations. Some of the 0. *nerka* stocks are distinguished as evolutionarily significant units (ESU). Others are not. A better understanding of the dynamics of all the 0. *nerka* populations in the basin should help us restore the anadromous form.

Spaulding (1993) reported initial efforts to describe 0. *nerka* population characteristics in Redfish and Alturas Lakes. These studies focused on fry recruitment from tributary streams and spawning escapement estimates. Complementary studies were completed in 1993 and are reported here. Also, a pilot study quantifying spawning habitat in Redfish Lake is included.

Methods

Tributary Fry Recruitment

In 1992, fry emergence estimates were made from Fishhook Creek, one of Redfish Lake's tributaries, and Alturas Lake Creek. In Fishhook Creek, fry were collected using three small fyke nets (30 cm wide and 60 cm deep). Nets were placed under Fishhook Creek bridge and anchored to the substrate using rebar. Corrugated pipe was used to connect the cod end of each fyke net with a live box. Fyke nets were operated daily -with few exceptions- between 20 April and 1 July 1993. Nets were generally fished between 1800 and 0800 hours. Fish captured in the nets were counted and immediately released back into the stream. During peak 'emergence, live boxes were checked every 20 min. Because of high water conditions and equipment constraints, we were unable to quantitatively assess fry emergence from Alturas Lake Creek.

Daily fry recruitment into Redfish Lake was estimated by summing the number of fish caught in all three traps, and applying a correction factor for trap efficiencies. Trap efficiencies were estimated using mark-recapture techniques and radishes. For the mark-recapture method, we marked fish with bismarck brown dye. Because of difficulties distinguishing dyed fish, radishes were also used to estimate trap efficiencies. Radishes are neutrally buoyant and should simulate passively migrating fish. On 27 May, similar results were reported when comparing both techniques (Table 1). Trap efficiencies were completed when obvious changes in stream discharge occurred.

Smolt Monitorinq

Recently, The Idaho Department of Fish and Game (IDFG) has monitored the migration of 0. *nerka* smolts at the Sawtooth Fish Hatchery weir. In 1991, an estimated 11,000 smolts passed the weir (Sawtooth Fish Hatchery; unpublished data). It is not known, however, whether these fish originated from Alturas, Pettit, or Yellow Belly Lakes. In order to describe which lakes are producing these smolts, we set a rotary screw trap upstream of the confluence of Pettit and Alturas Lake Creeks. The placement of the screw trap allowed us to estimate sockeye smolt production from Alturas Lake as well as contributions from Pettit and Yellow Belly Lakes.

Table 1. Trap efficiency estimates using dyed fish and radishes

Trap	Creek	Date	Method	Efficiency
fyke	Fish H.	12 May 1993	dye	14%
fyke	Fish H.	17 May 1993	dye	17%
fyke	Fish H.	27 May 1993	dye	6%*
fyke	Fish H.	27 May 1993	Radish	5%*
fyke	Fish H.	9 June 1993	Radish	6%
screw	Alt L.	9 June 1993	Radish	30%

* indicates comparison between dye and radish technique.

On 8 June 1993 we began operating the rotary screw trap. 0. *nerka* smolts generally begin migrating several weeks earlier (mid-May; unpublished data from the Sawtooth Fish Hatchery). Permit delays prevented an earlier start. Trap operation generally occurred between 1800 and 0700 hours. To protect juvenile salmonids from being consumed by squawfish or large salmonids, special care was taken to maintain cover (sage brush bundles) in the screw trap's live box. To monitor downstream migration timing and survival, we intended to PIT tag 0. *nerka* smolts. However, no 0. *nerka* smolts were collected. All other fish were identified to species and released.

We also placed a smolt monitoring station on Stanley Lake Creek. Stanley Lake, an historic rearing lake for sockeye salmon (Bjornn 1968), maintains a relatively high 0. *nerka* population (Luecke and Wurtsbaugh 1993). To find out if Stanley Lake is producing smolts, we deployed a fyke on the outlet stream. The fyke net (1.5 m X 1.5 m) was placed approximately 4.8 km below the lake. We operated the trap from 4 May to 19 June 1993. The trap was checked twice daily.

0. nerka Spawning Surveys

Stream Spawning: Stream surveys were conducted on all Stanley Basin lake tributaries that contain 0. *nerka* populations. Surveys began in August and were 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 and dividing by average stream life (stream life ≈ 12 days; Spaulding 1993). To distinguish areas of spawning selectivity, each stream was divided into a series of transects (see Figure 1A, B, and C). Other data collected during the surveys included otolith samples for age analysis and fork lengths.

Beach Spawning: In 1992, beach spawning was observed on sockeye beach, Redfish Lake (Spaulding 1992; unpublished data). With exception to genetic research, information describing the beach spawning population (residual sockeye) is limited. In 1993, we qualitatively estimated spawning densities on sockeye beach and surrounding areas.

Snorkel surveys were used to estimate spawning densities on sockeye beach. Each survey consisted of 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 to 5 m. During each survey, we counted spawning 0. *nerka* densities within the boundaries of sockeye beach as delineated by U.S.D.A. Forest Service signs. Surveys were completed at two week intervals. In addition to snorkel and dive surveys, a boat was used to search for spawning in other areas of the lake. Similar techniques were used to search for beach spawning in Alturas and Pettit Lakes.

Habitat Survev

Sockeye beach, appropriately named, is the area where most of the historic observation of sockeye spawning have

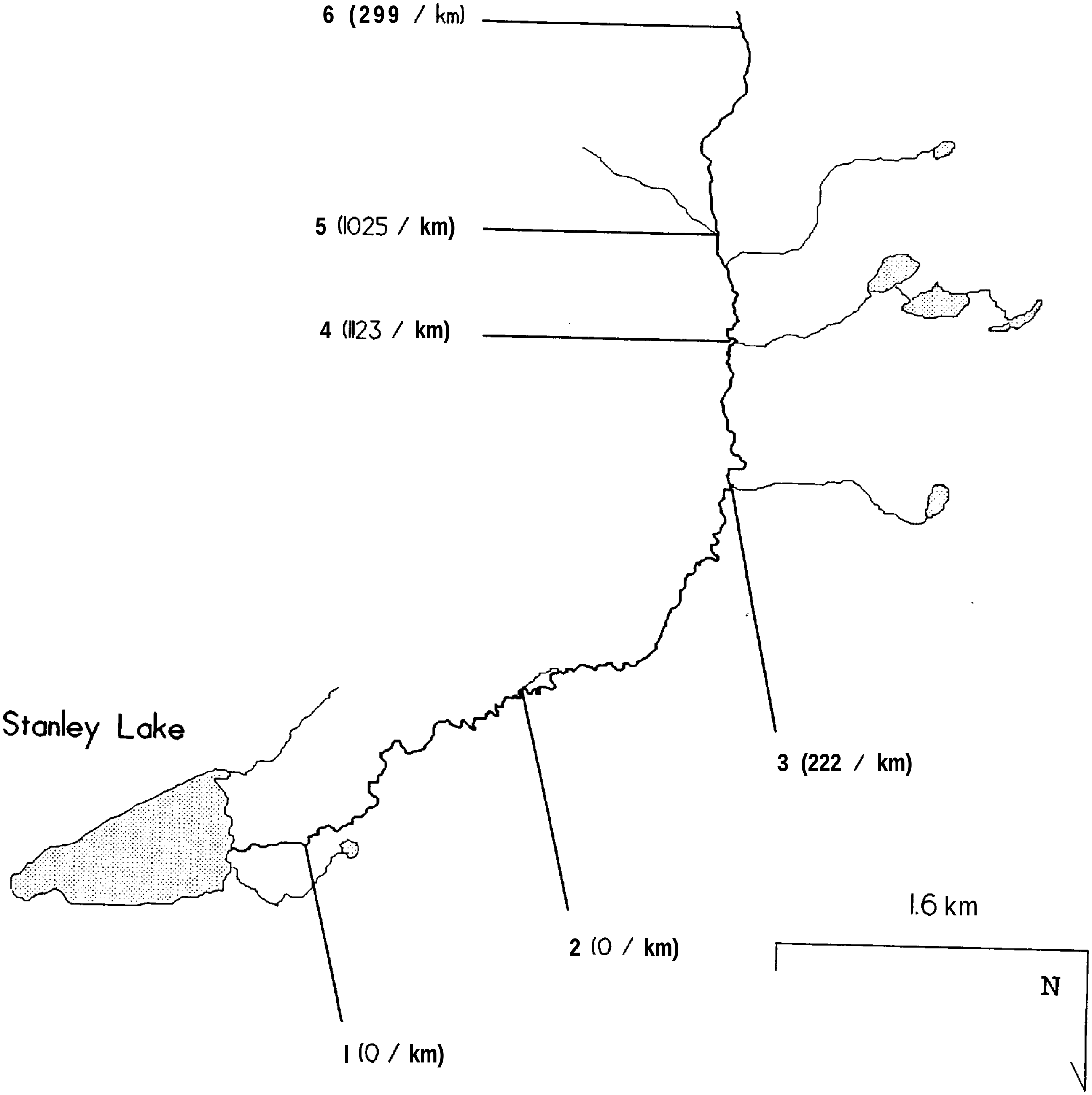

Figure 1A. Stanley Lake Creek spawning transects, Spawning densities (# / km) are shown in parenthesis,

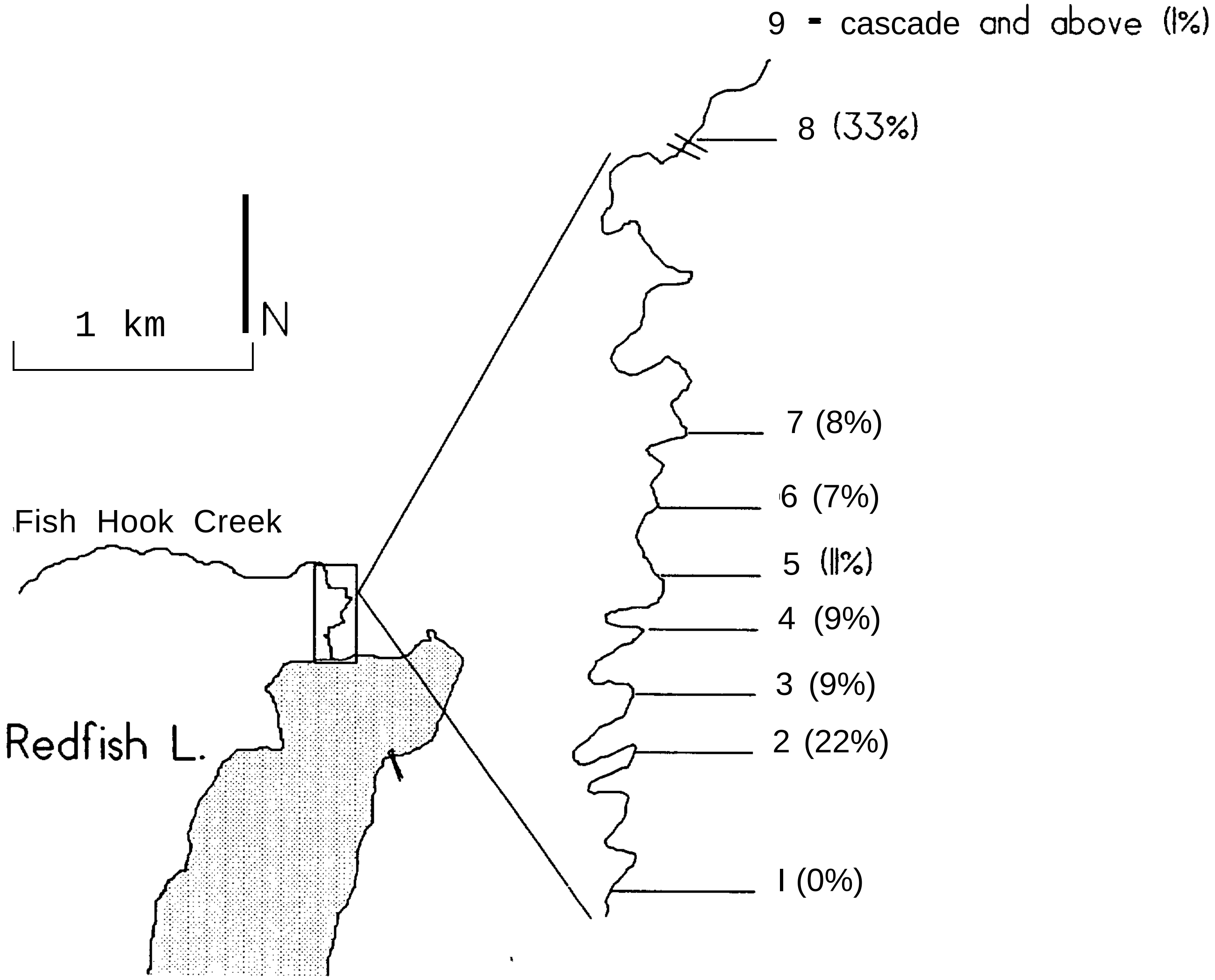

Figure 1B. Survey transects for O. nerka spawning grounds. Proportions of total spawning densities are shown in parentheses.

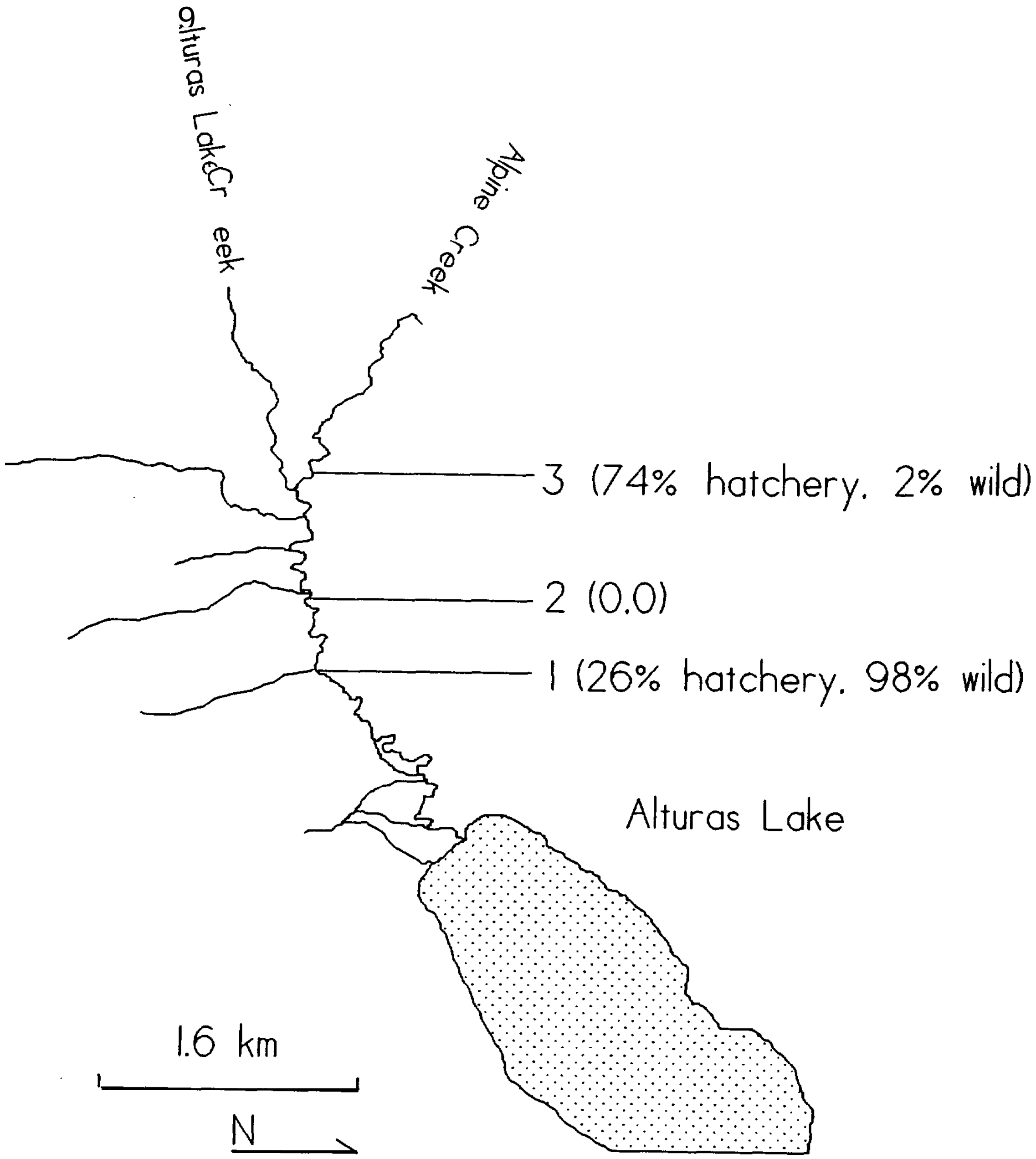

Figure 1C. **Alturas** Lake Creek spawning transects. Spawning densities (proportions) of hatchery and wild fish are indicated in parentheses (hatchery, wild).

occurred. Our objective was to quantify some of the habitat variables that make sockeye beach a successful spawning ground. In addition to quantifying habitat characteristics on sockeye beach, we sampled numerous other locations in the lake. This information should be useful for assessing available spawning habitat in Redfish Lake as well as for determining the suitability of other Stanley Basin Lakes for supporting introduced Redfish Lake stocks.

Substrate composition is an important variable that affects salmonid egg to fry survival (Platts et. al 1983). In this pilot study, we quantified substrate composition of known spawning sites and compared them to other shoal areas in Redfish Lake.

Habitat surveys were completed on Redfish, Alturas, and Pettit Lakes. Each lake was divided into four regions. Within a region, at least three replicate sites were sampled. Sample sites are shown in Figure 2. At each site two transects were completed (one parallel to shore and one perpendicular). Snorkel and SCUBA techniques were used to complete the parallel and perpendicular transects, respectively.

Parallel transects were 100 m long and completed in water $\approx$ 1.0 m depth. Residual sockeye were observed at similar depths in 1992 (Spaulding 1992; unpublished data). Every 2 m along the transect, a 0.1 m^2 plexiglass plate was held against the substrate and estimates of percent fines (< 6 mm), gravel

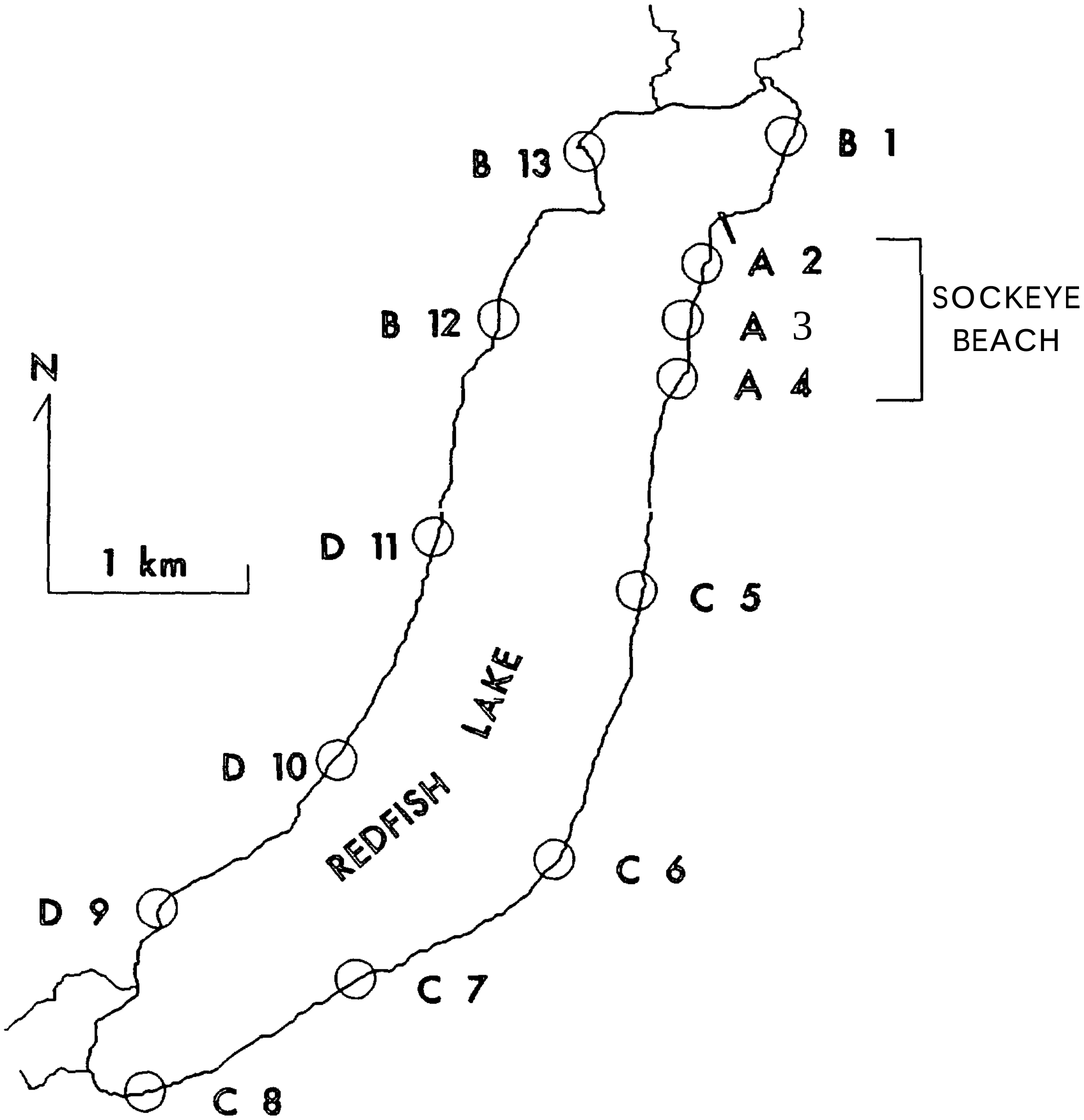

Figure 2, Sample sites for Redfish Lake Habitat Survey, Coordinates for each site are shown in Appendix 1.

(6 to 50 mm), cobble (51 to 100 mm), large cobble and boulder (> 100 mm), and embeddedness were recorded.

Perpendicular transects were completed to quantify habitat changes with depth. Perpendicular sites began near shore and continued to a maximum depth of 12.2 m. *O. nerka* have been reported to spawn in deeper water (20 m), but in most lakes and reservoirs shallower water is preferred (Gipson 1992). Three depth strata were chosen for data analysis. Strata criteria include shallow (0 - 3 m), intermediate (3.1 - 6.1 m), and deep (6.2 - 12.2 m). Mean bottom slope for perpendicular transects was estimated using transect length and depth of last position. An ANOVA was used to test for differences in habitat quality (% fines) among areas.

RESULTS

Tributary Frv Recruitment

Fishhook Creek fry production increased substantially over that reported in 1992 (Figure 3). Approximately 36,000 fry emerged from Fishhook Creek in 1992 compared to 160,000 in 1993. Interestingly, adult spawning densities in 1992 were only 33% greater than the 1991 estimate (9,600 and 7,200, respectively). Assuming 300 eggs per female (Spaulding 1993) and equal sex ratios, egg to fry survival in Fishhook Creek increased from 3% in 1992 to 12% in 1993. One possible cause for survival differences was increased runoff (see Appendix 3 for Fishhook Creek discharge results).

Although high water conditions prevented sampling from Alturas Lake Creek, few fry likely recruited. The 1992 spawning run for Alturas Lake Creek was estimated at 60 fish (Spaulding 1993). Assuming similar fecundity and egg to fry survival that occurred in Fishhook Creek in 1993, an estimated 1000 young of the year 0. *nerka* would have recruited to Alturas Lake via the inlet stream.

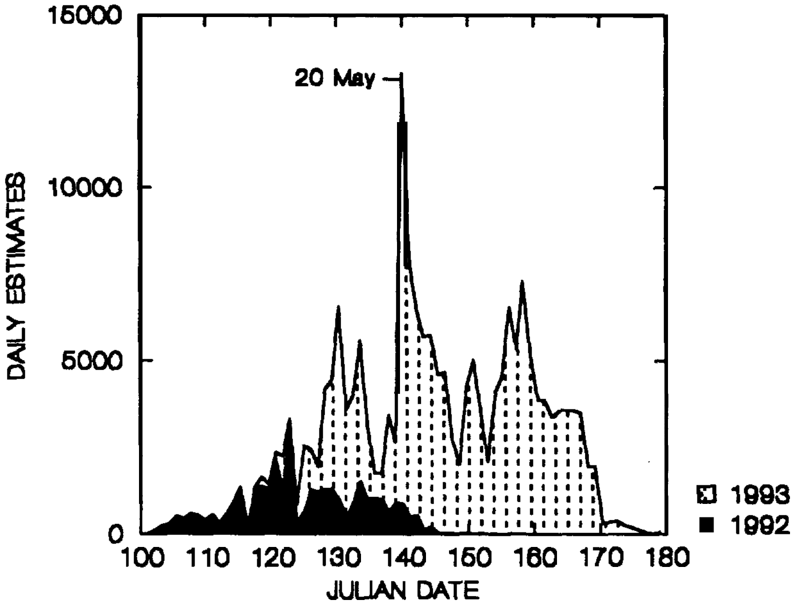

Figure 3. Daily fry emergence estimates in Fishhook Creek for 1992 and 1993.

Smolt Monitoring

In both the Alturas and Stanley Lake Creek monitoring stations no 0. *nerka* smolts were captured. In Alturas, the majority of fish captured were juvenile squawfish, suckers and dace (Table 2). For Stanley Lake Creek, dace made up 69% of all the fish caught followed by 14% squawfish and 11% sculpin.

Table 2. Non-target fish caught in the Alturas screw trap.

	June									
	5	8	9	11	13	16	18	20	25	Total
Sqwf	10	32	56	24	62	1	10	19	5	219
sucker	2	6	4	5	4	15	7	11	21	75
Dace	0	2	4	3	0	8	1	21	26	65
Shiner	0	0	0	0	0	1	5	3	2	11
C. smolt	1	2	0	0	0	0	0	0	0	3
C. fry	0	0	2	1	0	0	0	1	1	5
White F.	6	0	1	0	0	0	0	0	0	7
Sculpin	0	0	3	0	1	0	0	0	0	4

C. = Chinook

Spawning Surveys

Stream Populations: Escapement densities for 1993 in Fishhook and Alturas Lake Creeks were similar to estimates made in 1992 (Figure 4). Escapement'was greatest in Fishhook (10,800) followed by Stanley (1,900) and Alturas Lake Creeks (200). No stream spawning fish were observed in the South or North inlets to Pettit Lake. However, 46 ripe 0. *nerka* were captured from Pettit Lake using gillnets on 23 September. The Pettit 0. *nerka* population may be using shoal areas to spawn.

Run timing for Fishhook and Alturas Lake Creeks was very similar. The first observations of stream spawning occurred on 8 and 9 August in Fishhook and Alturas Lake Creeks, respectively. Spawning in Stanley Lake Creek may have begun earlier and lasted longer than the other stream populations. Our first survey of Stanley Lake Creek occurred on 17 August 1993. At that time, it appeared that spawning activity was

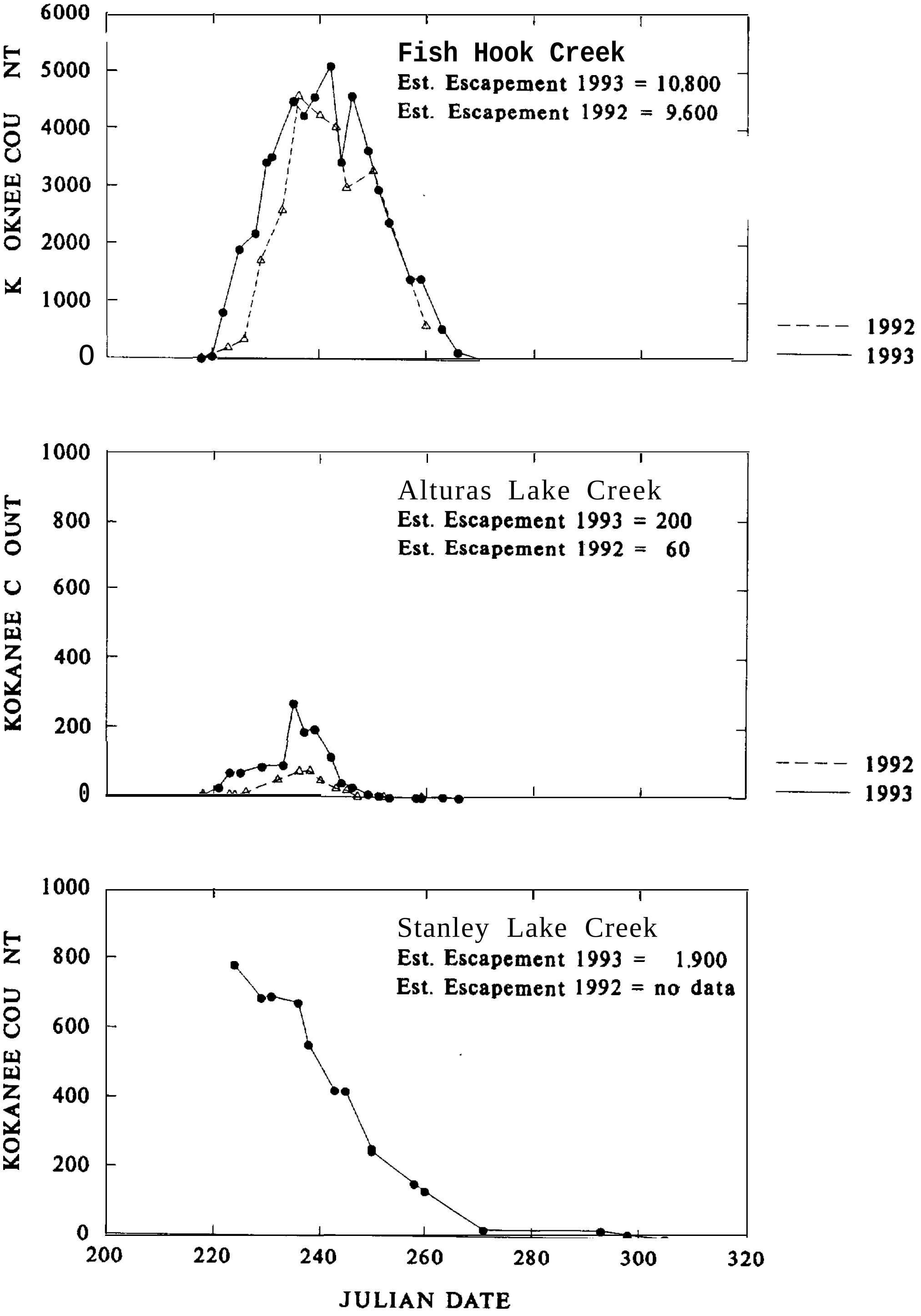

Figure 4. Kokanee escapement in three Stanley Basin tributaries. Notice different y axis for Fish Hook Creek.

advanced compared to the other tributaries (fish were paired and guarding redds). In addition to a possible early start, *0. nerka* continued to enter Stanley Lake Creek as late as 27 October, a month after spawning had ceased in the other streams (Figure 4).

Converse to differences in run timing, age and size composition of Fishhook and Stanley Lake Creek kokanee was similar (Figure 5). The majority of kokanee spawning in Fishhook and Stanley tributaries were age 3. Age at spawning in Alturas ranged from 2 to possibly 4. Age patterns for Alturas, however, are unclear due to the small otolith sample sizes. The mature kokanee captured in gillnets from Pettit Lake were also age 3.

Size and color of spawning fish varied among systems. The largest kokanee came from Pettit Lake, followed by Stanley and Fishhook Creeks. Alturas fish were extremely variable in size (Figure 5). Alturas fish also showed distinctive color differences compared to other stream spawning populations. The largest wild 0. *nerka* in Alturas were bright red in color. The more abundant smaller fish were generally paler. Both the early spawning Stanley fish and Fishhook Creek spawners were bright red. However, late spawners in Stanley Lake Creek (October) were pale in color, similar to small Alturas fish.

The spatial distribution of spawning in stream tributaries is depicted in Figures 1A, 1B, and 1C. In Alturas Lake Creek, most of the wild fish spawned in the lower

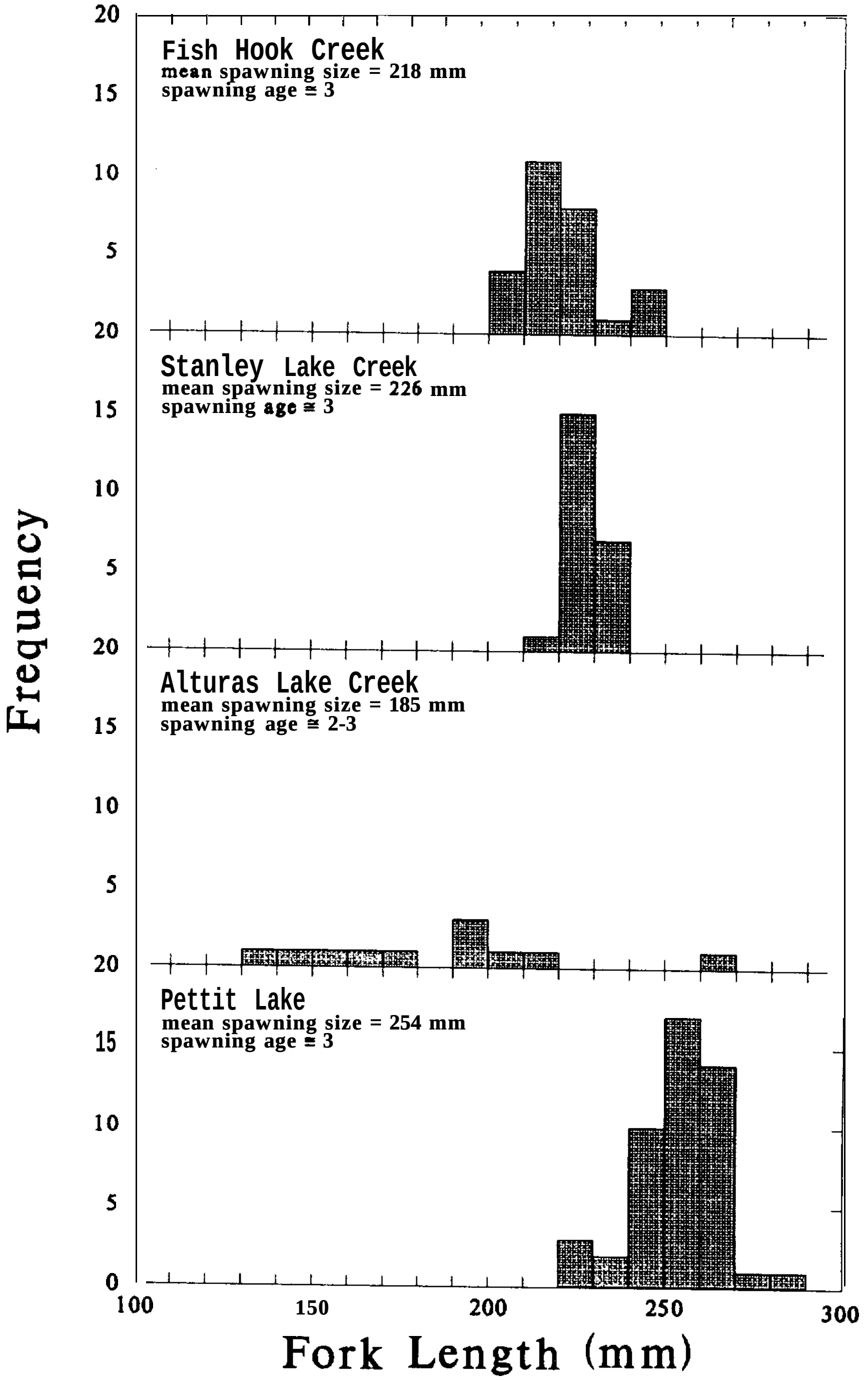

Figure 5. Age and size composition of spawning O. nerka in Stanley Basin tributaries and in Pettit Lake. Age was estimated using otoliths.

sections of the stream (Figure 1C). Outplants from the captive broodstock program moved into the upper reaches and even into Alpine Creek (Figure 1C). A few of the wild 0. *nerka* appeared to follow hatchery outplants upstream. Spawning between released captive broodstock and wild fish occurred. In Stanley Lake Creek, spawning densities were greatest in the middle reaches (Figure 1A). Spawning in Fishhook Creek appeared somewhat homogeneous through transect 8 (Figure 1B). A cascade in transect 9 may impede upstream passage in Fishhook Creek.

Beach Spawning: On 1 November 1993, 32 residuals were observed while snorkeling sockeye beach, in Redfish Lake. On the same day, two hatchery outplants were located by boat. The outplants were spawning in the south end of the lake near the small tributary across from Redfish Lake Creek. One of the fish was in ~ 1.5 m of water the other in ~ 10 m. At similar depth ranges in the same location, we counted approximately 13 large unguarded redds. No residual sockeye were observed in this area of the lake (south end).

Observations of wild 0. *nerka* spawning in Redfish Lake were made only on sockeye beach. A few 0. *nerka* were located in other areas (Table 3), but spawning activity was not confirmed. Similar to Redfish Lake, we observed adults along Pettit Lake shoals but could not confirm spawning sites. Table 3 summarizes results from all beach spawning surveys.

Table 3. 0. nerka counts made during beach spawning surveys. All counts refer to adult sized wild 0. nerka.

Lake	Method	Date	Location	Time	Count
Redfish	Snorkel^1	10/05/93	S. Beach	18:06	3
Redfish	Snorkel^2	10/20/93	S. Beach	20:56	21
Redfish	Snorkel^1	11/01/93	S. Beach	18:55	32
Redfish	Boat	11/01/93	S. Beach	12:00	28
Redfish	Boat	11/17/93	S. Beach	09:15	7
Redfish	Snorkel	10/05/93	Point C.	21:31	2
Redfish	Dive	10/21/93	Point C.	23:41	1
Alturas	Snorkel^1	10/14/93	WP 14	20:08	0
Alturas	Snorkel^1	10/20/93	WP 20	23:17	0
Pettit	Dive	10/05/93	WP 28	18:08	0
Pettit	Dive	10/06/93	WP 28	20:29	2
Pettit	Snorkel^1	10/06/93	WP 28	21:24	0
Pettit	Snorkel^1	10/06/93	WP 26	21:50	0
Pettit	Dive	10/20/93	WP 28	21:10	1

Superscripts refer to the number of transects completed.
Weigh point locations are in Appendix 5.

Habitat Survey

Substrate composition was similar between parallel and perpendicular transects. In both cases, percent fines were greatest in the sockeye beach area (82 and 80%, respectively; Table 4). Percent fines were significantly lower in other areas of the lake (Figure 6; $p < .001$ for parallel and perpendicular transects). Gravel and small cobble substrate (habitat conducive to successful spawning) contributed little to overall substrate composition (Table 4). Areas C and D contained relatively high proportions of large cobble and steep bottom slopes (Table 4). Large substrate and steep bottom slopes may impede spawning activity.

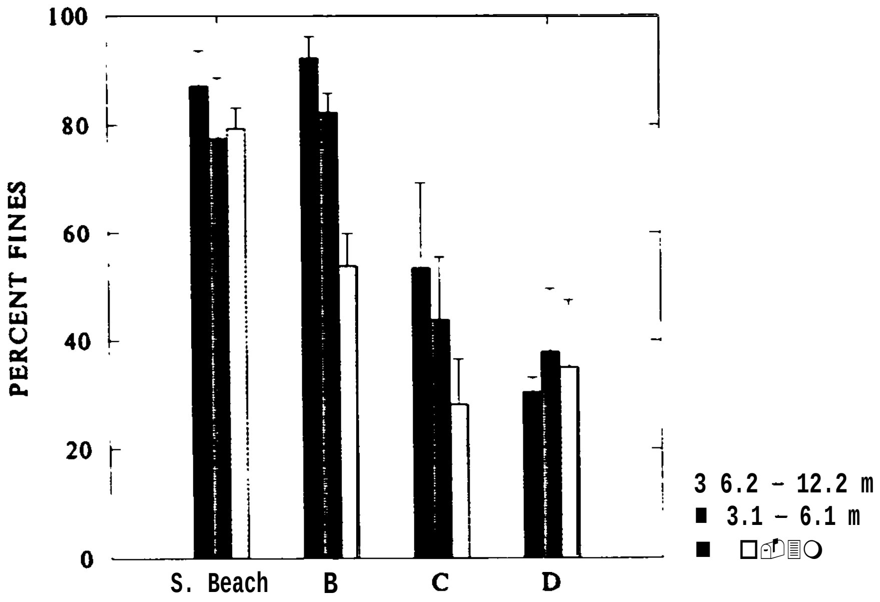

Figure 6. Percent fines in Redfish Lake. See Figure 2 for location of areas sampled. Error bars are standard error.

Table 4. Percent substrate composition, embeddedness, and bottom slope in four areas of Redfish Lake. Data are means and (standard error). Perpendicular (perp.) and parallel (para.) transects are included.

Area	fines	Grav.	S.cobb	L.cobb	Embed.	Slope
Perp.						
A*	82(12)	9 (6)	4 (3)	5 (4)	67(17)	8(4)
B	71(12)	18 (7)	3 (3)	5 (3)	66(18)	11(6)
C	43(23)	19 (3)	4 (3)	23(27)	52(26)	19(9)
D	38(17)	25(12)	6 (1)	22 (7)	43 (7)	20(9)
Para.						
A*	80 (9)	10 (4)	3 (2)	7 (9)	68(11)	
B	69(18)	18(20)	5 (1)	6 (5)	58(15)	
C	34(29)	16 (7)	8 (6)	34(27)	33(16)	
D	33(12)	31 (8)	10 (4)	24(14)	37 (6)	

* Sockeye Beach

DISCUSSION

Spring runoff may sharply influence Fishhook Creek fry recruitment. Fry recruitment increased three fold between 1992 and 1993. This increase occurred when spawning densities were similar (7,200 and 9,600 for 1991 and 1992, respectively). Spring runoff, however, was significantly higher in 1993 (See part II of this document for discharge data). Cooper (1965) reported that embryo survival is positively related' to the amount of water passing through a sockeye salmon redd. Discharge and fry recruitment will be monitored in 1994 so that a third year of comparisons can be made.

The residual population that spawns on sockeye beach in Redfish Lake appears minimal. Our snorkel data indicated that less than 100 sockeye beach residuals spawned in 1993. Assuming equal sex ratios and 300 eggs per female, egg deposition on sockeye beach was 12,500. In 1993, Fishhook Creek kokanee deposited over 1.3 million eggs. It is possible that other residual populations exist in Redfish Lake, but none were found.

Spawn timing was very similar between 1992 and 1993. For both Alturas and Fishhook Creek, spawners entered the streams the first week of August and ceased spawning towards the middle of September. These dates were very similar to those reported 1992 (Spaulding 1993). The only peculiar timing came

DISCUSSION

Spring runoff may sharply influence Fishhook Creek fry recruitment. Fry recruitment increased three fold between 1992 and 1993. This increase occurred when spawning densities were similar (7,200 and 9,600 for 1991 and 1992, respectively). Spring runoff, however, was significantly higher in 1993 (See part II of this document for discharge data). Cooper (1965) reported that embryo survival is positively related to the amount of water passing through a sockeye salmon redd. Discharge and fry recruitment will be monitored in 1994 so that a third year of comparisons can be made.

The residual population that spawns on sockeye beach in Redfish Lake appears minimal. Our snorkel data indicated that less than 100 sockeye beach residuals spawned in 1993. Assuming equal sex ratios and 300 eggs per female, egg deposition on sockeye beach was 12,500. In 1993, Fishhook Creek kokanee deposited over 1.3 million eggs. It is possible that other residual populations exist in Redfish Lake, but none were found.

Spawn timing was very similar between 1992 and 1993. For both Alturas and Fishhook Creek, spawners entered the streams the first week of August and ceased spawning towards the middle of September. These dates were very similar to those reported 1992 (Spaulding 1993). The only peculiar timing came from Stanley Lake Creek where fish entered the stream earlier and continued to spawn in late October. Attempts will be made in 1994 to take genetic samples from the late spawning fish.

The stream spawning populations in Alturas Lake Creek exhibited several differences from the other Stanley Basin populations. First, the spawning population is small (60 and 200 fish for 1992 and 1993, respectively). This factor concerns us because, similar to the residual population in Redfish Lake, the Alturas stock has maintained anadromous characteristics (11,000 smolts in 1991, and recent adult returns; Bowler 1990). If this population should perish, we may lose a critical portion of the Snake River sockeye population. Additionally, the majority of the 0. *nerka* spawning in Alturas Lake Creek were small in size compared to other populations. Small size may reflect poor forage conditions in Alturas Lake, compounding problems related to limited numbers.

Results from the habitat study suggest that substrate composition may not be the limiting factor for beach spawning populations in Redfish Lake. Percent fines were greatest in known spawning areas (sockeye beach). Other factors including ground water inputs, BOD, and temperature effects need to be considered in future studies. Additionally, research comparing egg to fry survival between Fishhook Creek and sockeye beach would help develop a more complete analysis of spawning habitat quality and availability in Redfish Lake.

CHAPTER 2:

COMPETITION, PREDATION, AND FISH COMMUNITY ASSESSMENT

Redfish, Alturas, Pettit, Yellow Belly, and Stanley Lakes (including their inlet and outlet streams) are listed as critical habitat for recovery of Snake River sockeye salmon (FR 68543). In addition to critical habitat distinction, the Stanley Basin lakes provide numerous recreational opportunities. To meet recreational demands, the IDFG has introduced rainbow trout *(Oncorhynchus mykiss)*, brook char *(Salvelinus fontinalis),* and lake trout *(S. namaycush)* in to some of the these lakes. One of our objectives is to ensure that the 0. *nerka* populations in these systems are not deleteriously affected by stocking or by management for non-native species. In addition to possible interactions with introduced species, northern squawfish *(Ptychocheilus oregonensis),* and redside shiners *(Richardsonius balteatus)* may also hinder sockeye recovery efforts. A better understanding of all species interaction, in critical habitat systems, should facilitate future management decisions. This section reviews information collected during gillnet and trawling surveys.

Methods

Stanley, Pettit and Alturas Lakes were sampled in June and September 1993. Multi-panelled sinking gill nets of square mesh sizes 1.90, 2.54, 3.17, 3.81, 5.08, and 6.35 centimeters were used. In each lake, four areas were sampled at depths ranging from 0 – 30 m. See Appendix 5 for sample locations. 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. Fish were identified and measured for fork length.

For diet analysis, fish stomachs were removed and placed in 70% ethanol. Stomach lavage was used to collect non-lethal diet samples from bull char and lake trout. 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). Diet overlap indices were calculated using equations described by Koenings et al. (1987).

Fish collected in trawl surveys were also sampled for stomach contents. Trawling was completed in June and September 1993. Stomach contents were handled as described above.

RESULTS

Species Comoosition

The benthic fish community in Alturas Lake is dominated by norhtern squawfish and suckers. In September, catch rates were 2.02 and 1.89 fish / hr, respectively. Hatchery rainbow trout, bull char, *0. nerka,* and whitefish were sampled at lower frequencies (.14, .09, .01, and .04 fish / hr, respectively) Spring trends were similar (Table 5).

Table 5. Gillnet catch rates (fish / hr) in Alturas, Stanley and Pettit Lakes in 1993.

	Alturas June, Sep.	Stanley June. Sep.	Pettit June, Sep.
Bull Char	0.03, 0.09	0.00, 0.00	0.00, 0.00
Brook Char	0.00, 0.00	0.06, 0.17	0.05, 0.15
Lake Trout	0.00, 0.00	0.19, 0.18	0.00, 0.00
Rainbow T.	0.14, 0.14	0.42, 0.29	0.02, 0.64
0. nerka	0.00, 0.01	0.55, 0.12	0.00, 0.59
Suckers	0.69, 1.89	0.00, 0.00	0.00, 0.00
Squawfish	1.06, 2.02	0.00, 0.00	0.01, 0.00
Redside S.	0.00, 0.00	0.00, 0.00	0.22, 0.59
Whitefish	0.09, 0.04	0.00, 0.00	0.00, 0.00

In Stanley and Pettit Lakes, fish communities were dominated by salmonids (Table 5). Rainbow trout and 0. *nerka* were consistently the most abundant fish caught in Stanley Lake. Stanley Lake also appeared to have a modest Lake trout population (catch rates of .19 and .18 fish / hr; Table 5). In Pettit Lake, rainbow trout, *0. nerka* and redside shiners were well represented (September data; Table 5). A few brook char were captured from both lakes.

Length frequency distributions for most of the species caught in gillnets are shown in Figures 7A, B, and C. Lengths of rainbow trout ranged from 160 to 420 mm fork length. Mean fork lengths for rainbow trout in September for Alturas, Pettit, and Stanley were 266, 266, and 255 mm, respectively. Rainbow trout sizes were similar between June and September sample periods (Figures 7A, B, and C). Most of the rainbow trout exhibited frayed and missing fins indicating recent introduction. Only one rainbow trout (420 mm, caught in June from Pettit Lake) was suspected to be a carryover from a previous year. *0. nerka* sizes ranged from 160 to 280 mm fork length, with Pettit Lake producing the largest fish (Figure 7C). Length frequency distributions for squawfish, suckers, bull char and lake trout are shown in Figures 7A-C.

Gillnets were not set in Redfish or Yellow Belly Lakes by the Shoshone-Bannock tribes. However, Redfish Lake was sampled by IDFG personnel who were targeting bull char. Squawfish and suckers were the most abundant fish caught (R. Dillinger, IDFG personal communication). Interestingly, Redfish and Alturas Lakes, two systems without non-game fish barriers, exhibit similar fish community structure. Yellow Belly Lake was sampled by IDFG in 1992. Cutthroat trout and brook char were captured (IDFG, unpublished data), and it is unlikely that any 0. *nerka* survived recent rotenone programs.

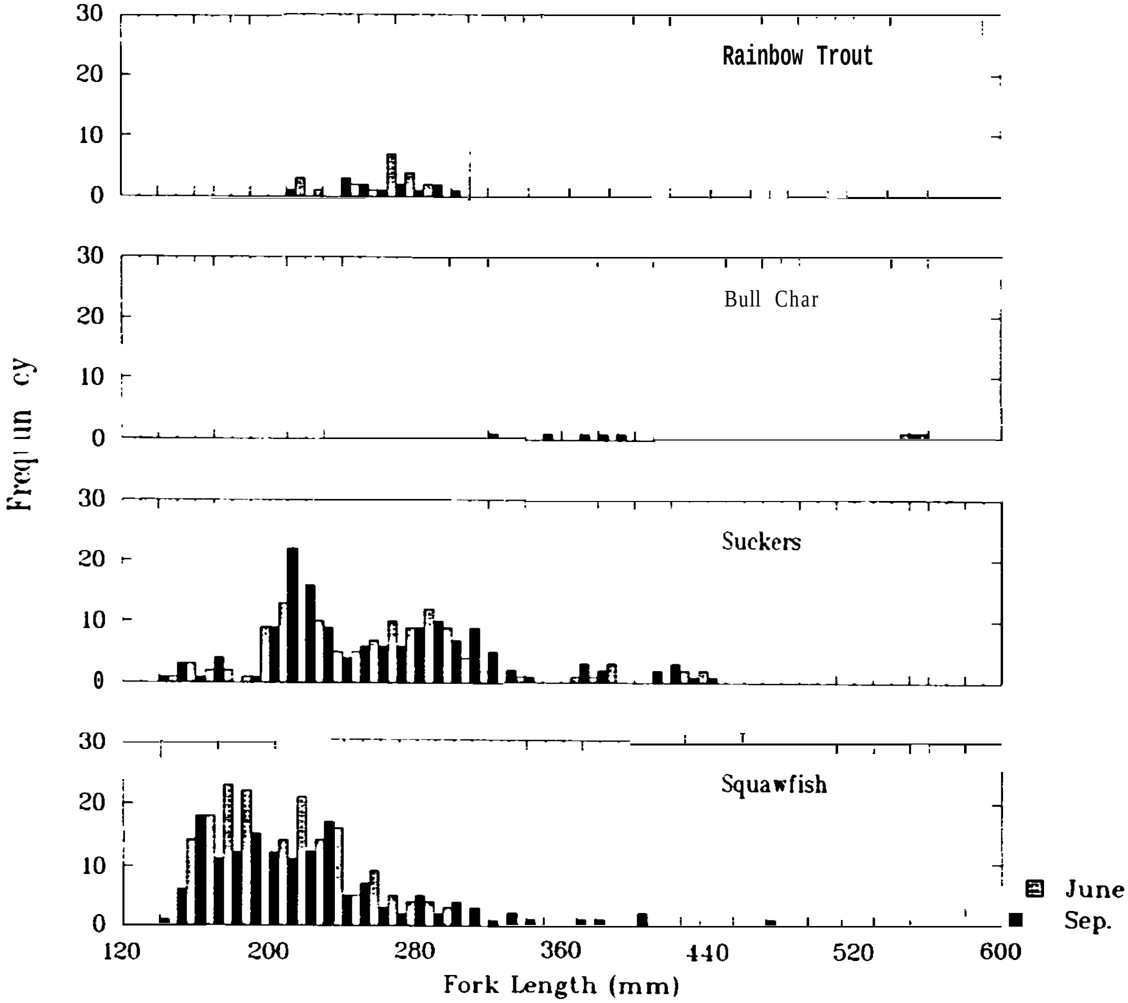

Figure 7A. Length frequency distributions for fish caught in gillnets in Alturas Lake.

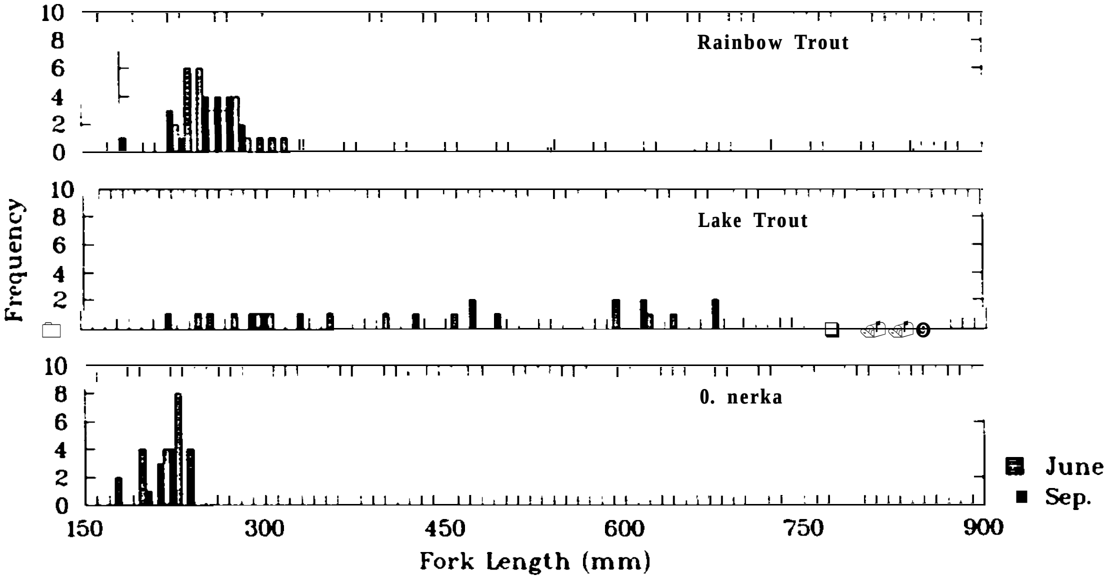

Figure 7B. Length frequency distribution for fish caught in gillnets in Stanley Lake.

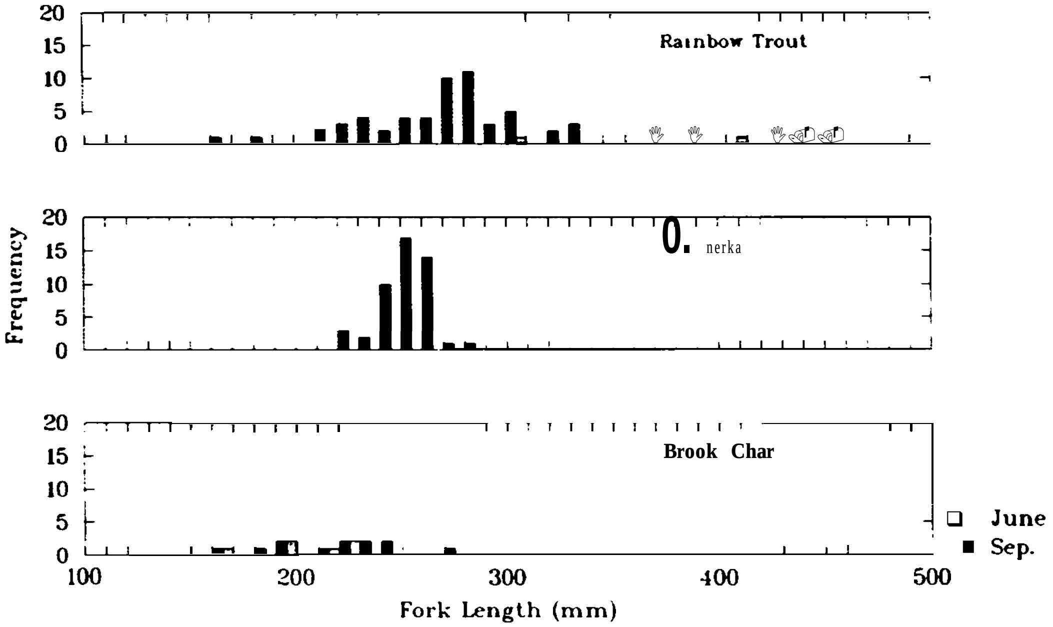

Figure 7C. Length Frequency distributions for fish caught in gillnets in Pettit Lake.

Foraging Habits

Competitors: Given the extreme oligotrophic nature of the Stanley Basin Lakes (Luecke and Wurtsbaugh 1993), food resources for many of the fish species may be limiting. Competitive interactions within and among species may cause additional foraging stress resulting in poor 0. nerka growth and survival. This section reviews foraging habitats of 0. *nerka* and potential competitors.

0. nerka populations among the Stanley Basin Lakes share similar temporal trends in foraging behavior. In June, chironomid pupa made up 93%, 98%, and *80%* of 0. *nerka* stomach contents by weight for Stanley, Alturas, and Redfish Lakes, respectively. By September, *0. nerka* shifted their diets to more zooplankton prey (Figure 8). *Daphnia* was the most important zooplankton prey in Redfish, Pettit and Stanley Lakes. In Alturas Lake, *Polyphemus* made the largest proportion of zooplankton prey (Figure 8). Preliminary winter analysis indicated that Stanley Lake *0. nerka* continued to feed on zooplankton. The dominant prey item in December and January samples was Copepods.

In Alturas Lake, rainbow trout diets were markedly different from those of 0. *nerka* diets. In June, rainbow trout favored terrestrial insects (71% by weight), whereas 0. *nerka* preferred chironomid pupa (98%). When 0. *nerka* shifted to zooplankton prey (in the fall), rainbow trout stomachs were filled with plant material. Diet overlap between 0. *nerka* and

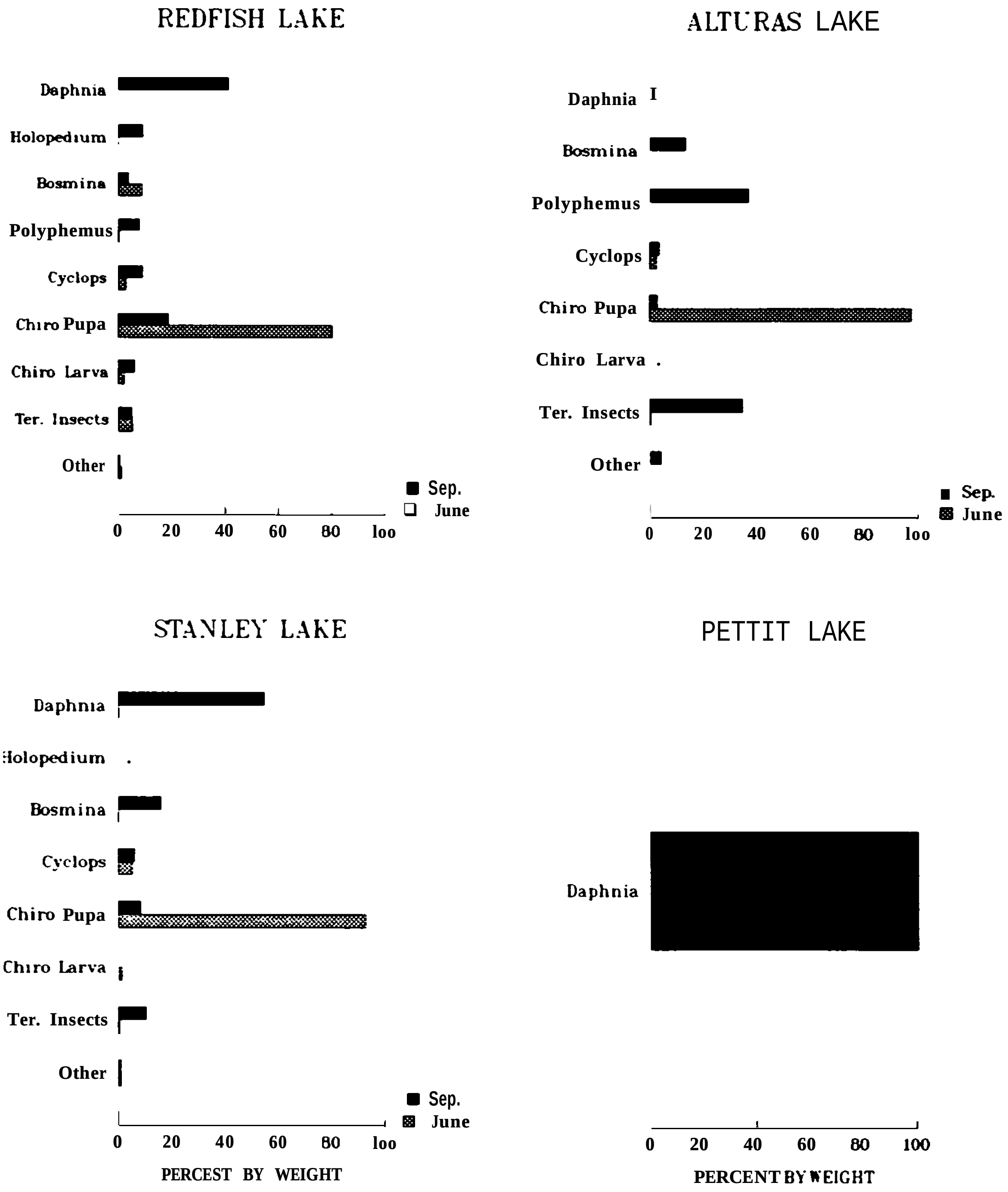

Figure 8. O. *nerka* diet composition (percent by weight) in June (hashed bars) and September (solid bars). Samples were collected during gillnet and trawling surveys.

rainbow trout was 20% in June and .04% in September. Other species that may compete with 0. *nerka* for food in Alturas Lake include whitefish and young of the year squawfish. Diet results for these fish are not available. In 1994, we hope to collect such information.

Pettit Lake, diet overlap among species was limited. Redside shiners predominantly foraged on Odonata larva (70% and 91% in June and September, respectively). Rainbow trout consumed a broad range of invertebrates as well as a few redside shiners (September data; Table 6). 0. *nerka* diets in September consisted of 100% *Daphnia* prey, yielding 0% diet overlap with redside shiners and rainbow trout. Small sample sizes precluded spring comparisons.

In Stanley Lake, rainbow trout and 0. *nerka* shared similar food resources. Both species consumed chironomid pupa in June and zooplankton prey in September (See Figure 8 and Table 6). Diet overlap between 0. *nerka* and rainbow trout was 94% and 47% for June and September, respectively.

Predators: Predation losses in the natural environment may frustrate sockeye recovery efforts as much as mortalities in the brood stock program or in the migration corridor. Numerous predaceous species exist in the Stanley Basin Lakes (Spaulding 1993). However, little is known about their foraging habits or densities. In this section, foraging habits of northern squawfish, bull char, rainbow trout, and lake trout are reported. Future research will be directed at estimating predator densities and quantifying the magnitude of *O. nerka* consumption.

Table 6. Rainbow trout stomach contents (percent by weight) for Stanley, Alturas, and Pettit Lakes.

Prey	Stanley	Alturas	Pettit
June			
Daphnia	0.0	0.0	
Chironomid Pupa	71.1	15.4	
Chironomid Larva	0.0	0.0	
Terrestrial Insects	10.3	71.4	
Aquatic Insects	0.0	5.8	
Mollusca	0.0	1.1	
Plant	13.4	6.3	
Fish	0.0	0.0	
Other	5.2	0.0	
September			
Daphnia	17.2	0.0	0.0
Chironomid Pupa	2.9	0.5	31.0
Chironomid Larva	3.9	6.8	0.0
Terrestrial Insects	34.0	3.0	10.8
Aquatic Insects	3.4	22.5	21.6
Mollusca	22.8	0.0	18.5
Plant	3.3	59.2	8.1
Fish	10.8	5.6	9.8
Other	1.6	2.2	0.2

Northern squawfish currently reside in significant numbers in Redfish and Alturas Lakes as well as in companion tributaries. Because of permit constraints, however, diet samples were collected only from Alturas Lake. A total of 138 samples were analyzed from squawfish ranging in length between 110 and 480 mm fork length. Samples were collected in June (n = 50), September (n = 62), and December (n = 26). Because of relatively small sample sizes and no apparent differences in prey selection, data for all size classes were pooled.

Squawfish diets were very different among seasons. In June, squawfish prey was dominated by oligochaetes (28% by weight), terrestrial insects (20%), *O. nerka* (16%), and sculpins (11%). In September, squawfish diets consisted of mostly snails (34%), suckers (23%) and O. *nerka* (16%). In December, sculpins and aquatic insects combined for 75% of squawfish prey (Figure 9). *O. nerka* were absent from squawfish diets in winter samples.

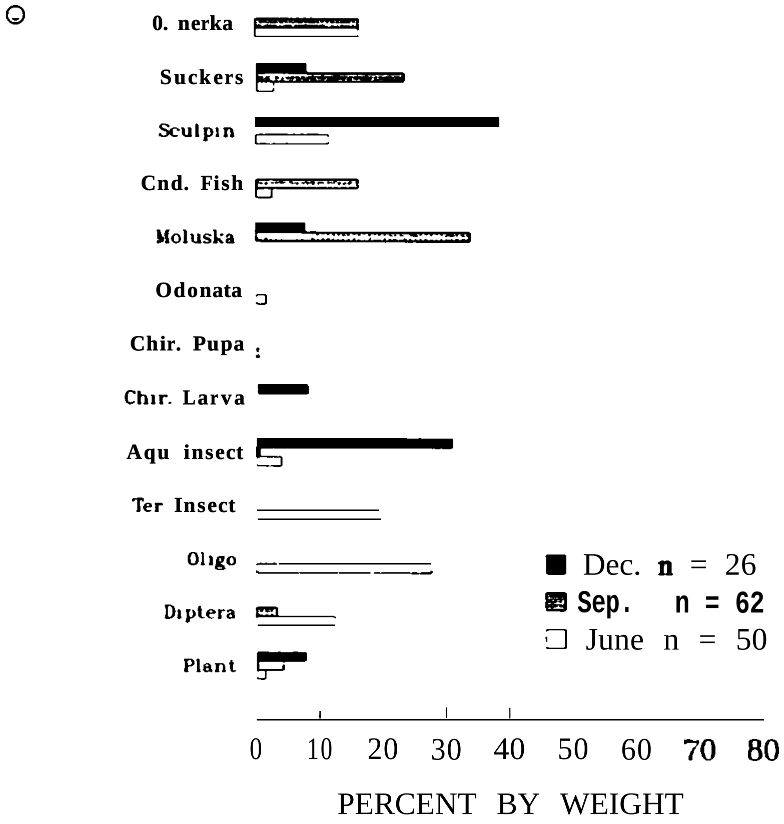

Figure 9. Squawfish diet composition (percent by weight) for fish caught in gillnets in Alturas Lake.

Eleven stomach samples were analyzed from bull char. Unlike squawfish, bull char preferred 0. *nerka* prey. 0. *nerka* made up 89% of bull char stomach content. The remaining 11% was unidentified fish prey. Because no other fish were identified in the stomachs of bull char, the unidentified portion was likely 0. *nerka.*

Lake trout in Stanley Lake showed seasonal changes in diet composition. A total of 17 stomachs were analyzed from lake trout ranging in size between 220 to 800+ mm fork length (n = 10 in June, n = 7 in September). In June, lake trout diets consisted of 66% chironomid pupa, 19% redside shiners, and only 3% 0. *nerka.* In September, *0. nerka* made up 83% of the diet followed by 10% chironomid larva and 7% unidentified fish.

Piscivory by hatchery rainbow trout was limited, but noted in Alturas, Pettit, and Stanley Lakes (Table 6). Redside shiners were the major fish prey for hatchery rainbow trout in Pettit and Stanley Lakes. In Alturas Lake, sculpins were the dominant fish prey. Interestingly, rainbow trout piscivory occurred only in September. The absence of fish in June stomach samples may have resulted from a lack of predation skill (fish were stocked in June), or changes in food resources (e.g., reductions in the abundance of invertebrate prey caused the shift to fish).

DISCUSSION

Before discussing the competition and predation results, we caution the reader on two important facts. First, these results represent two snapshots in time (one sample in June and one sample in September 1993). These relationships may not hold for other periods (i.e., winter). Secondly, diet analysis is a preliminary step in understanding the impacts of competitive or predatory interactions in fish communities. The magnitude of these interactions can not be determined from diet analysis alone.

Competition: In general, competitive interaction between hatchery rainbow trout and wild 0. *nerka* populations may not occur. Our diet analysis suggested that foraging habits of 0. *nerka* and hatchery rainbow trout were sharply different in Alturas and Pettit Lakes. In Stanley Lake, however, rainbow trout and 0. *nerka* shared similar food resources, but common prey items may not have been limiting. For example, June samples showed that chironomid pupa were highly selected by both rainbow trout and 0. *nerka.* However, *0. nerka* gut fullness was very high during the June sample period indicating that rainbow trout were not limiting the available forage. Additionally, virtually all fish species in Stanley Lake were consuming chironomid pupa in June. If chironomids were in short supply, it is unlikely that voracious predators like lake trout would spend time foraging on such small prey.

Concerns over high diet overlap would be much greater in a system like Alturas Lake, where food resources are scarce and growth is poor (it generally takes two years to reach smolt size in Alturas Lake compared to one year in Redfish Lake). Hence, the 20% diet overlap between rainbow trout and *0. nerka* in Alturas may be more detrimental to the 0. *nerka* population than the 98% found in Stanley.

Intraspecific competition may have a greater impact on sockeye recovery than interspecific interactions. Bjornn et al. (1968) reported significant declines in Redfish Lake smolt size as densities increased. Additionally, in limnocorral enclosure experiments, kokanee densities had a greater impact on growth than nutrient addition (see Part 2 of this document). These data demonstrate the importance of careful management of brood stock introductions to Stanley Basin Lakes. Supplementation of 0. *nerka* populations under current conditions may crash food resources which may in turn produce small, potentially inferior smolts. Therefore, steps to increase lake productivity or control kokanee populations are being considered as methods of offsetting increases in foraging pressure brought on by broodstock supplementation. Feasibility studies directed at increasing lake productivity are ongoing, but early results indicate that fertilization experiments may not yield desired results (see Part 2 of this report). Productivity experiments will continue in 1994.

Controlling kokanee populations may be necessary to expedite sockeye recovery efforts. In 1993, an estimated

10,800 adult kokanee spawned in Fishhook Creek, Redfish Lake. Under similar lake productivity, and assuming each spawning kokanee in Fishhook Creek weighed 100 g, Redfish Lake could have produced 108,000 10 g smolts. The largest smolt population estimated leaving Redfish Lake during Bjornn et al.'s study was 65,000 (mean between 1955 and 1961 was 27,000). Bjornn et al. (1968) also monitored kokanee spawning in Fishhook Creek. In 1962 no kokanee were observed spawning in Fishhook Creek. We suggest that the decline in sockeye over the last forty years was paralleled with increases in Fishhook Creek kokanee. Kokanee filled the void left by sockeye. Therefore, controlling Fishhook Creek spawning escapement or emergent fry densities should be considered as one alternative to insure that forage resources are available forage sockeye juveniles. Alternatively, sockeye progeny could be reared in hatchery facilities until they reach smolt stage, and then released into Redfish Lake. This strategy can not and should not be used exclusively. It can not be used exclusively because (under current policy) mature sockeye from the brood stock program will be released to spawn naturally. The progeny from those fish will exert additional foraging pressure on food resources. Secondly, in addition to limited hatchery rearing we should utilize natural environments and processes where possible.

Predation: *0. nerka* populations are being preyed on by Lake trout, bull char, and squawfish. Although the magnitude of losses can not be determined from diet analysis, several generalities can be mentioned. First, even though squawfish in Alturas Lake do not appear to select for any one fish species, their numbers are so great that they probably exert greater impacts on 0. *nerka* populations than the less abundant bull char. Secondly, the lake trout population in Stanley Lake may be controlling 0. *nerka* growth. Forage resources (zooplankton and benthic production) are greatest in Stanley Lake, but 0. *nerka* growth is comparable to systems with lower forage resources (i.e., Redfish Lake; see Part 2 of this report for limnology data).

Additionally, the lake trout population in Stanley Lake may be reproducing. Stanley Lake was stocked with 15,000 lake trout in 1975. In 1986, IDFG completed a creel and gillnet survey of Stanley Lake (Reingold and Davis 1987). The mean size and age of lake trout taken was 680 mm and ten years old, respectively. Reingold and Davis (1987) concluded that the lake trout were not reproducing successfully. Our conclusions differ because, in 1993, we collected several lake trout less than 300 mm fork length (Figure 7B). Age analysis was not completed on these fish, but it is unlikely that they were nineteen years old. In 1994, we plan to estimate the lake trout population. Once the population estimate is complete, a bioenergetics model will be used to estimate 0. *nerka* consumption rates. This information will help direct future management and recovery goals for Stanley Lake.

1994 GOALS AND MANAGEMENT SUGGESTIONS

GOALS

- ★ Collect diet information on whitefish and young of the year squawfish from Alturas Lake.
- ★ Set vertical gillnets in Alturas, Stanley, and Pettit Lakes to evaluate pelagic fish composition and feeding habits.
- ★ Estimate Lake trout population size in Stanley Lake-.
- ★ Use bioenergetics modeling to estimates lake trout consumption of 0. *nerka* in Stanley Lake.
- ★ Monitor fry recruitment in all the Stanley Basin tributaries that support spawning populations.
- ★ Continue monitoring stream spawning densities.
- ★ Locate beach spawning locations in Pettit and Alturas Lakes (if any).
- ★ Use core sample techniques to collaborate substrate findings from the pilot habitat study completed in 1993.

MANAGEMENT SUGGESTIONS

- ★ Develop a contingency plan for controlling Fishhook Creek kokanee population size in the event that fertilization experiments prove unsuccessful.
- ★ Close fishing in Alturas Lake Creek during spawning periods.
- ★ Maximize adult broodstock releases in Redfish Lake. An expected 800 sockeye from the brood stock program have the potential to mature in the fall of 1994.
- ★ Take steps to begin the process of incorporating Alturas Lake outmigrants into the brood stock program. Develop broodstock objectives through discussion with IDFG and seek Stanley Basin TOC approval.
- ★ Take steps to begin the process to reintroduce sockeye into Pettit Lake.

ACKNOWLEDGEMENTS

Kenneth Ariwite of the Shoshone-Bannock Tribes was instrumental in collecting field data and completing diet analysis. Scott Spaulding initiated the spring fry trapping and gillnet surveys, as well as the field season objectives. Bob Griswold helped design the spawning habitat study and collect field data. Wes Stonecypher, Marcus Colby, and Hal Hayball assisted in gillnet surveys and spawning counts. We also thank Rob Dillinger and Bob Griswold for their intellectual guidance during the 1993 field season.

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APPENDICES

Appendix 1. Sample locations for gillnet sites and habitat surveys.

Lake	Site	Gillnet	Habitat	WP	location
Redfish	RFL-1		X	1	lat. N44 08.400 lon.W114 54.812
Redfish	RFL-2		X	2	lat. N44 08.161 lon.W114 55.096
Redfish	RFL-3		X	3	lat. N44 07.900 lon.W114 55.176
Redfish	RFL-4		X	4	lat. N44 07.805 lon.W114 55.171
Redfish	RFL-5		X	5	lat. N44 06.473 lon.W114 55.248
Redfish	RFL-6		X	6	lat. N44 06.473 lon.W114 55.550
Redfish	RFL-7		X	7	lat. N44 06.038 lon.W114 56.159
Redfish	RFL-8		X	8	lat. N44 05.613 lon.W114 57.241
Redfish	RFL-9		X	9	lat. N44 06.170 lon.W114 57.053
Redfish	RFL-9		X	10	lat. N44 06.609 lon.W114 56.428
Redfish	RFL-10		X	11	lat. N44 07.158 lon.W114 56.011
Redfish	RFL-11		X	12	lat. N44 07.975 lon.W114 55.820
Redfish	RFL-12		X	13	lat. N44 08.447 lon.W114 55.470
Alturas	ALT-1		X	14	lat. N43 55.435 lon.W114 51.352
Alturas	ALT-2		X	15	lat. N43 55.513 lon.W114 51.055
Alturas	ALT-3		X	16	lat. N43 55.219 lon.W114 50.707
Alturas	ALT-4		X	17	lat. N43 54.877 lon.W114 50.979
Alturas	ALT-5		X	18	lat. N43 54.558 lon.W114 51.290
Alturas	ALT-6	X	X	19	lat. N43 54.350 lon.W114 51.891
Alturas	ALT-7	X	X	20	lat. N43 54.348 lon.W114 52.670
Alturas	ALT-8	X	X	21	lat. N43 54.355 lon.W114 52.266
Pettit	PET-l		X	22	lat. N43 58.622 lon.W114 51.959
Pettit	PET-2		X	23	lat. N43 58.617 lon.W114 52.387
Pettit	PET-3	X	X	24	lat. N43 58.563 lon.W114 52.979
Pettit	PET-4	X	X	25	lat. N43 58.652

					lon.W114 53.515
Pettit	PET-5	X	X	26	lat. N43 58.894 lon.W114 53.539
Pettit	PET-6	X	X	27	lat. N43 58.952 lon.W114 53.052
Pettit	PET-7	X	X	28	lat. N43 59.015 lon.W114 52.404
Pettit	PET-8		X	29	lat. N43 58.906 lon.W114 52.055
Stanley	STN-1	X		30	lat. N44 14.526 lon.W115 03.833
Stanley	STN-2	X		31	lat. N44 14.564 lon.W115 03.415
Stanley	STN-3	X		32	lat. N44 14.779 lon.W115 03.072
Stanley	STN-4	X		33	lat. N44 14.800 lon.W115 03.581

KEY:
Daph = *Daphnia*
Hol = *Holopedium*
Bos = *Bosmina*
Pol = *Polyphemus*
Cal = Calanoid
Odo = Odonota

Cyc = Cyclopoid
CP = Chironomid Pupa
CL = Chironomid Larva
TI = Terrestrial Insects
AI = Aquatic Insects
Olg = Oligochaet

Eur = *Eurycerus*
Amp = Amphipod
zoo = zooplankton
O.n = 0. *nerka*
Rss = redside shiner
Misc = Miscellaneous

SC1 = Sculpin
Suk = Sucker
SQWF = Squawfish
BULT = Bull trout
LKT = Lake trout

Appendix 2A. 0. *nerka* diet (percent by weight) in Sawtooth Valley Lakes.

		PREY										
Lake	Date	Dap.	Hol.	Bos.	Pol.	Cal.	cyc.	C.P.	C.L.	Eur.	Amp.	T.I.
Redtish	6/93	0.0	0.0	8.8	0.2	0.0	2.8	80.1	2.2	1.0	0.0	4.9
Redtish	9/9341 .o	9.1	3.7	7.7	0.4	8.8	18.7	5.8	0.0p2382X0.0		4.9	
Stanley	6/93	0.0	0.0	0.0	0.0	0.0	5.2	92.7	1.0	0.6	0.2	0.4
Stanley	9/93	54.4	4.7	15.9	0.0	0.5	6.0	‘8.4	0.0	0.0	0.0	10.2
Stanley	12/93	6.6	0.0	1.3	0.5	71.5	20.1	0.0	0.0	0.0	0.0	0.0
Stanley	1/94	0.0	0.0	0.2	0.0	33.7	66.0	0.0	0.0	0.0	0.0	0.0
Alturas	6/93	0.0	0.0	0.0	0.0	0.0	2.3	97.6	0.0	0.0	0.0	0.0
Al turas	9/93	2. I	0.0	13.1	36.6	0.0	3.3	2.8	3.9	3.7	0.0	34.5
Pettit	9/93	100.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

Appendix 2B. Rainbow trout diet (percent by weight) in Sawtooth Valley Lakes.

		PREY									
Lk	Date	Dap.	CP.	CL.	TI.	AI.	Mol.	Odo.	Plt.	Scl.	Rss.	Misc.
Alt.	6/930.0	15.4	0.0	71.4	5.8	1.1	0.0	6.3	0.0	0.0	0.0	
Alt.	9/930.0	0.4	6.5	2.6	26.7	0.0	0.0	57.0	4.7	0.0	1.8	
Stn.	6/930.0	71.1	0.0	10.3	0.6	0.0	0.0	13.4	0.0	0.0	4.6	
Stn.	91981.2	2.9	3.9	34.0	2.7	22.8	0.7	3.3	0.0	10.8	1.6	
Pet.	9/930.0	31.0	0.0	10.8	0.6	18.5	21.0	8.1	0.0	9.9	0.2	

Appendix 2C. Diet content (percent by weight) for species collected in gillnets.

			PREY															
Lk	Spec.	Date	O.n	Suk.	Scl.	Rss.	Udf.	mol.	Odo.	C.P	C.L	A.1	T.I	Olg.	Dip.	Zoo.	Plt.	Misc.
Alt.	SQWF	6/93	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	0.0
Alt.	SQWF	9/93	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	0.0
Alt.	SQWF	12193	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	93	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
Stn.	LKT	6/93	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
Stn.	LKT	9/93	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
Pet.	RSS	6/93	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
Pet.	RSS	9/93	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

PART TWO:

LIMNOLOCICAL ANALYSES AND FIELD EXPERIMENTS TO ASSESS MANAGEMENT STRATEGIES FOR ENDANGERED SOCKEYE SALMON IN SAWTOOTH VALLEY LAKES

LIMNOLOGICAL ANALYSES AND FIELD EXPERIMENTS TO ASSESS MANAGEMENT STRATEGIES FOR ENDANGERED SOCXEYE SALMON IN SAWTOOTH VALLEY LAKES

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Executive Summary

Limnological investigations were made on five Sawtooth Basin lakes and their drainages in 1992 and 1993 to evaluate their capacity for recovery of the endangered Snake River sockeye salmon. This report provides additional data to that reported after our 1992 investigation (Luecke & Wurtsbaugh 1993). Here we report on five main components of the work: (1) The nutrient and water budgets of the lakes (Redfish, Pettit, Alturas, Stanley and Yellow Belly Lakes); (2) Physical, chemical, plankton, and fish population monitoring in the lakes; (3) Limnocorral experiments in Redfish Lake designed to determine how low and high levels of fertilization would affect plankton and fish production; (4) Similar limnocorral experiments in Pettit Lake designed to evaluate how surface and deep-water fertilizations could affect lake productivity, and; (5) Net-pen experiments and modeling to evaluate how zooplankton food resources and temperature would influence the growth of sockeye salmon.

The increased precipitation during 1993, compared to that during the 1992 drought year, greatly increased both the water and the nutrient budgets of the lakes. Water inflow into the lakes in 1993 was near the long-term average, and it was 2-3 times greater than in 1992. Water residence times with normal precipitation varied between 0.3 yr (Stanley Lake) and 3.0 years in Redfish lake. Calculated phosphorus and nitrogen budgets for Redfish Lake were 2-3 times higher in the normal precipitation year than during the drought. Areal nutrient loadings in 1993 were nevertheless very low: phosphorus loadings ranged from 4 mg/m^2 in Pettit Lake to 11 mg/m^2 in Stanley Lake. Water and nutrients entering in streams plunge to deeper waters in the lake because stream water temperatures are cooler and denser than those in the lakes over much of the year. These plunging inflows thus divert nutrient inputs to the surface of the lakes and this probably contributes to their unproductive state. Returning sockeye salmon were estimated to have contributed 3% of phosphorus loading when 4,400 adults spawning population of 25,000 adults.

Limnological monitoring indicated that the lakes were cooler in 1993 than in 1992 and that thermal stratification was less pronounced. Conductivity and temperature profiles indicated that some of the lakes do not mix every fall and/or spring, thus explaining why they sometimes have very low oxygen levels in the lowest strata despite their low productivity. Dissolved oxygen and temperature are nevertheless suitable for sockeye salmon growth throughout most of the water column in all of the lakes. Higher nutrient loading in 1993 than in 1992 increased phytoplankton in the lakes over those observed in 1992. Nevertheless, mean summer surface chlorophyll a concentrations in 1993 ranged between 0.5 and 1.0 mg m^{-3}, indicating that they are very unproductive. Mean annual chlorophyll levels were highly correlated with nutrient loading levels in the different lakes. Because phytoplankton populations are low, water transparency is high, with Secchi depth readings varying between 8 and 15 m during midsummer. The high transparencies permit phytoplankton populations to develop in deeper strata, and ^{14}C primary production measurements indicated that these algae contribute over 50% of the productivity in the lakes. The phytoplankton populations in most of the lakes are dominated by diatoms, small chlorophyta, *Dinobryon sp.*. Cyanobacteria are present in low numbers. Zooplankton biomasses were lower in the lakes in 1993 than in 1992, despite the higher nutrient loading and algal populations.

Nutrient additions to the surface waters of 340 m^{-3} polyethylene tubes (limnocorrals) containing Redfish Lake water stimulated phytoplankton biomass, chlorophyll a, primary productivity, and zooplankton egg production over control treatments. The experiments did not demonstrate, however, higher zooplankton abundances or higher growth rates of juvenile kokanee salmon in the fertilized limnocorrals.

Pettit Lake limnocorral experiments demonstrated that phytoplankton and zooplankton egg production could be stimulated with either surface (epilimnetic) or deeper (metalimnetic) nutrient additions. When nutrients were added to the metalimnia of the limnocorrals, algal growth and production was stimulated primarily in the deeper waters, thus maintaining more transparent surface layer.

Net-pen experiments in Stanley and Redfish Lake indicated that both temperature and zooplankton food abundance will interact to control the growth of salmon. When fish were held in the warmer surface water they only grew when food was abundant (Stanley Lake) but not when it was limited (Redfish Lake). Bioenergetic simulation models were moderately successful in predicting salmon growth rates in the different temperature x food experiments.

Empirical and simulation modeling excercises indicated that increasing nutrient loadings to the lakes has the potential to double the production of sockeye salmon. Adult sockeye salmon returning to the Sawtooth Valley lakes would not contribute substantially to the nutrient budgets of the lakes unless outmigrant survival was enhanced considerably.

Chapter 1

Water and Nutrient Budgets of the Sawtooth Valley Lakes

by

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INTRODUCTION

Hydrologic studies in 1992 and 1993 by Utah State University focused on the quantification of water budgets and nutrient loading of the historical rearing lakes of the Snake River sockeye salmon *(Oncorhynchus nerka)* in the Sawtooth Valley of southcentral Idaho – Redfish, Alturas, Pettit, Stanley, and Yellow Belly Lakes. The lakes' watersheds formed logical and convenient landscape units for these investigations (Likens 1985, Rigler 1977). These studies have provided data essential for understanding the lake systems in three areas: (1) flushing rates, which help us understand how quickly the lakes would respond to perturbations in nutrient loading; (2) the bottom-up controls on lake primary productivity, including temporal and spatial variations; and (3) the effect of the decline of sockeye runs on the nutrient budgets, and thus primary productivity.

Lake productivity and nutrient loading are important factors to consider when assessing strategies for the release of the endangered Snake River sockeye salmon back into one or more of the lakes. Our studies have shown these lakes to be highly oligotrophic (Budy et al. 1993 and Chapter 2, this report), with both N and P co-limiting algal production (Gross et al. 1993). In turn, nutrient loading is positively correlated with algal biomass (Dillon and Rigler 1974, Vollenweider 1976). Finally, algal standing crop has been shown to be positively related to *Oncorhynchus nerka* biomass in ten of Idaho's oligotrophic lakes (Rieman and Myers 1992).

One strategy being considered to increase the fitness of outplanted 0. *nerka* pre-smolts is lake fertilization. The decline of anadromous 0. *nerka* has most likely resulted in decreased nutrient loading from spawned-out salmon to the five Sawtooth Valley lakes. Spawned-out salmon can be an important part of a lake's nutrient budget (Krokhin 1967, Stockner 1987, Koenings and Burkett 1987). For example, when overfishing reduced spawning sockeye salmon by 97% in Lake Dalnee, phosphate levels fell by 75% and primary production by 40% (Krogius 1979, as cited by Thorpe

1986). A nutrient "deficit" caused by loss of salmon to fishing or other causes can be made up with nutrient additions to stimulate plankton production and, subsequently, fish growth and survival (Nelson 1958, Stockner and Shortreed 1985, Kyle et al. In press). Our hydrologic investigations have focused on providing data to **evaluate the need for lake fertilization.** This data is also necessary to compare annual **lake nutrient** inputs **from the watershed, with nutrients returned to the lakes by spawning** *O. nerka.*

Each of our five study lakes exhibit a deep chlorophyll layer, **with** meta- and hypolimnetic chlorophyll a levels 1.2 to 9.4 times the level in the epilimnion during the ice-free stratified periods (Budy et al. 1992 and Chapter 2, this report). Several reasons have been proposed to explain this phenomena in other lakes - light attenuation in the water column, reduced grazing pressure, **differential sinking rates of** algal cells, and/or higher nutrient concentrations at depth (Fee 1976, Richardson et al. 1978, Shortreed and Stockner 1990, Gasol et al. 1992).

Our **hydrologic investigations have focused on evaluating** another possible factor responsible for the formation and/or maintenance of deep chlorophyll maxima - *plunging inflows.* In the Sawtooth Valley lake systems in 1992, temperature measurements showed stream inflow temperatures to be similar to those of each lakes' metalimnion and upper hypolimnion. We hypothesized that these colder, and thus denser, inflows would result in an interflow to the lakes, or cause a *plunging inflow,* that delivers nutrients to the metalimnion and hypolimnion. This may contribute to the ultraoligotrophic status of the epilimnion and the formation of a deep chlorophyll maximum in each lake. Vincent et al. (1991) described how plunging inflows in a New Zealand lake deprived the **epilimnion of nutrients and affected eutrophication.**

To meet these objectives, we have measured water budgets and nutrient loading for total nitrogen (TN) and total phosphorus (TP) for the Sawtooth Valley Lakes in 1992 and 1993 to quantify lake water residence time and annual and seasonal fluxes of nutrients.

This provided a good contrast between an extreme drought year (1992) and a more normal water year (1993). The water budgets for both 1992 and 1993 are reported here. A water budget for Yellow Belly Lake was not developed for 1993. Nutrient budgets for Alturas, Pettit, Redfish, and Stanley Lakes for 1993 are also reported, as well as for Redfish Lake for 1992. Additionally, we intensively measured stream and lake temperatures to determine if plunging inflows occurred. The ongoing analyses of the plunging inflow investigations are discussed briefly.

STUDY AREA

The five study lakes are located in the Sawtooth Valley National Recreation Area in southcentral Idaho (lat. 44°, long. 115°) at elevations between 1985 and 2157 m (Fig. 1, Table 1). The lake's watersheds lie mostly within the Sawtooth Wilderness Area, and drain the east side of the granitic Sawtooth and Smoky Mountains. The U-shaped lake basins were heavily glaciated during the Pleistocene when glaciers advanced just beyond the mouths of the mountain valleys, depositing large moraines behind which the lakes are impounded (Killsgaard et al. 1970, Alt and Hyndman 1989). Detailed investigations into the streamflow hydraulics and geometry and the water quality of the channels of the Upper Salmon River area were conducted in the early 1970's by Emmett (1975).

Table 1. Sawtooth Valley Lakes physical and morphometric data.

Lake	Elev. (m)	Surface Area (km^2)	Mean Depth (m)	Max. Depth (m)	Drainage Area (km^2)	Max. Drainage Elev. (m)
Redfish	1996	6.15	44	91	108.1	3277
Alturas	2138	3.38	32	53	75.7	3246
Pettit	2132	1.62	28	52	27.4	3246
Stanley	1985	0.81	13	26	39.4	3005
Y. Belly	2157	0.73	14	26	30.4	3246

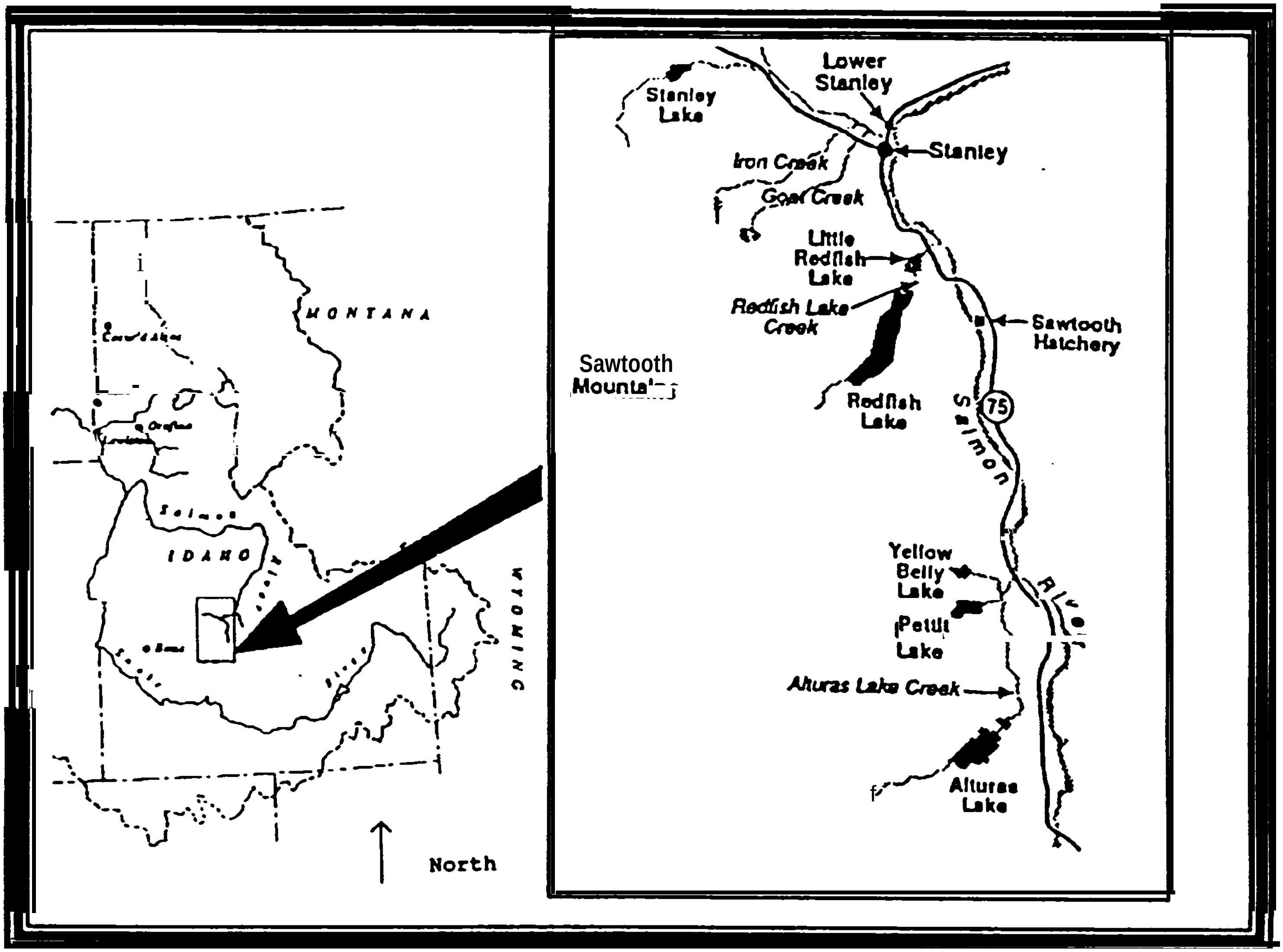


Figure 1. Location of Sawtooth Valley Lakes, ID.

The study lakes are popular regional destinations for boating, fishing and water sports, hiking, camping, and picnicking. The development on the lakes consists of a cabin/lodge complex on Redfish Lake, approximately twenty vacation cabins on Pettit Lake, organizational camp complexes directly below Alturas Lake, and a developed campgrounds at Redfish, Alturas, and Stanley Lakes. All lakes except Yellow Belly Lake are accessible by hard-surface roads and have boat ramps.

Alturas and Stanley Lakes each have one perennial inflow. Redfish Lake has two perennial inflows, Redfish Lake Inflow and Fishhook Creek. Pettit Lake has two perennial inflows, called (for the purposes of this report) Pettit North Inflow and Pettit South Inflow. In the 1992 annual report (Spaulding 1993), these streams were referred to as Pettit Main Inflow and Pettit Intermittent Inflow, respectively.

METHODS

Lake Water Budgets

A lake's annual water budget can be expressed by the following (Likens 1985):

$$I + P + R + G = O + E + S_p + \delta S_t, \quad (1)$$

where I = stream inflows,
P = precipitation on lake surface,
R = non-channelized runoff,
G = groundwater inflows,
O = stream outflows,
E = evaporation,
S_p = seepage, and
δS_t = annual change in lake storage.

G and S_p were not quantified in this investigation; δS_t is negligible for the Sawtooth Valley Lakes.

Stream Inflows and Outflows. Staff gauges were installed in

1992 on the inflows and outflows of the five lakes. Discharge was measured using a Marsh-McBirney Flo-Mate 2000 electromagnetic flow meter. Measurements were taken at 15-20 points along a cross-section. If stream depth was $\leq$ 0.75 m, flow was measured at six-tenths of depth; if stream depth was $>$ 0.75 m, flow was averaged between measurements taken at two- and eight-tenths of depth (Marsh-McBirney 1990). A gauge reading of stream stage was recorded with each discharge measurement, allowing the development of a stage-discharge relationship for each stream (e.g., Fig. 2). This allowed discharges to be determined on several dates by recording the stage height. In 1993 we directly measured flows or took gauge readings to estimate discharge at least weekly for each stream from late-April through September, with more intensive measurements made during the peak runoff between mid-May and late June. In 1992, these methods were used to estimate discharge at least 3 times per month from May through September. Several additional measurements were also made during baseflow conditions both years. Additionally, l-3 gauge readings were taken daily on Fishhook Creek in 1993 and 1992 and on Alturas Lake Inflow in 1992 during the kokanee fry emergence period (Shoshone-Bannock Tribes, Dept. of Fisheries). This period encompassed the rising limb, peak, and falling limb of the local hydrograph each year.

Missing discharge information for some of the streams were estimated from regressions for those creeks with the more intensively measured discharges in Fishhook Creek and Alturas Lake Inflow. Linear regressions with suitable r^2 values ($\geq$0.93) between these two streams and the other study inflows were used to derive these values. Comparisons to USGS streamflow data (USGS, Boise, ID) for the Salmon River at Salmon, ID (Station ID 13302500) were used to augment the hydrographs during winter baseflow periods.

Precipitation. Since the study lakes are ice-covered several months each year, precipitation input was partitioned into precipitation received during the ice-free season, P,, and during the ice-cover season, P,. P_f was an immediate input into the lake. P_f was derived from recordings made at a weather station in Stanley

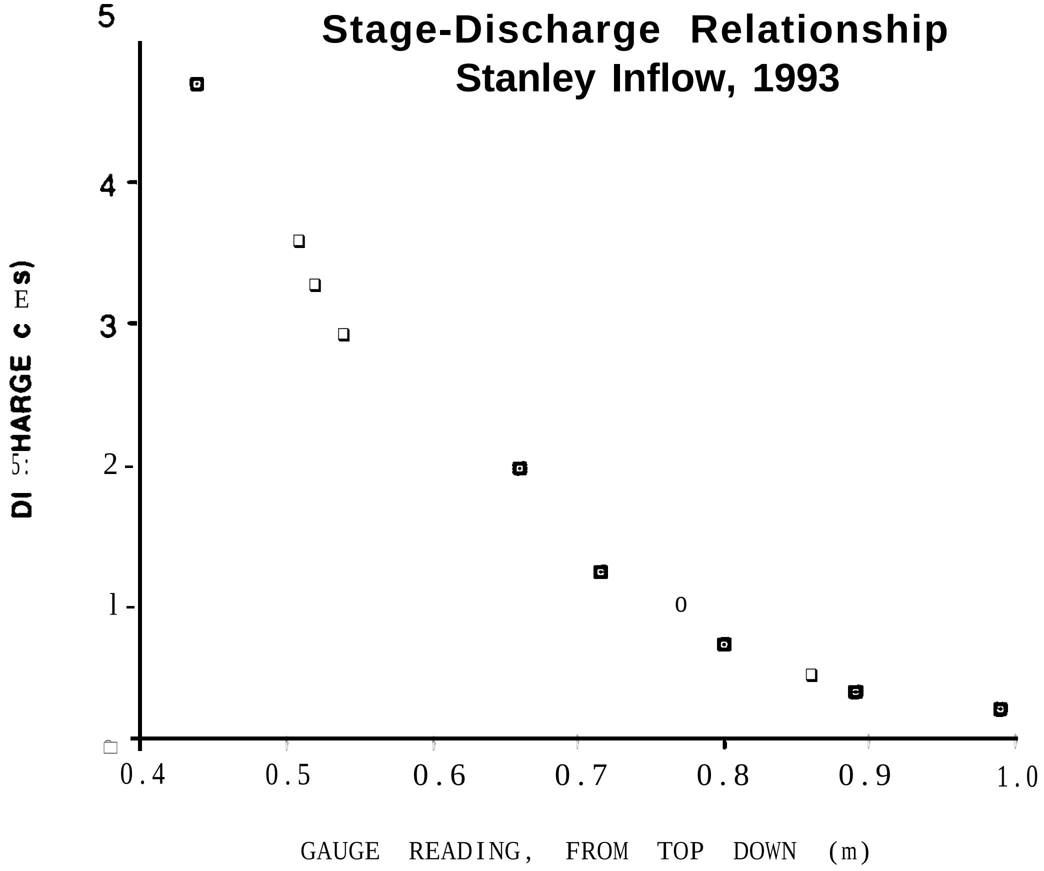


Figure 2. Stage-discharge relationship for Stanley Lake Inflow, 1993.

(Station ID 108676) which is part of the State Climatologist's network (M. Molnau, State Climatologist, University of Idaho, Moscow). In addition, P_f was recorded from a gauge near the Stanley Ranger Station (Mark Moulton, U.S. Forest Service) and the authors. The later recordings were used for months for which data from the Idaho State Climatologist were not available. For months with recordings for both gauges, results were similar.

P_1 accumulates on the frozen lake; some of it sublimates or evaporates. The rest does not become a direct water input to the lake until the ice cover breaks up in the spring. In order to determine this amount, data from the Soil Conservation Service (SCS) snow courses was used (SCS, Boise, ID). The snow water equivalent (SWE) of the snowpack for April 1 at the Redfish Lake Flat (RFL) site was used to estimate the quantity of the precipitation input which accumulated on the lake before ice-out. Because the RFL snow course became inactive in 1990, we predicted the SWE for this site using regressions based on 30 years of data (1961-1990) from two other SCS snow courses, Banner Summit (BS) and Vienna (VM), in the region. Linear regressions between each site and the RFL site yielded the following equations:

$$RFL_{SWE1} = 0.492*BS_{SWE} - 1.58,\ r^2 = 0.82 \quad (2), \text{ and}$$
$$RFL_{SWE2} = 0.346*VM_{SWE} - 0.561,\ r^2 = 0.85 \quad (3).$$

The means of RFL_{SWE1} and RFL_{SWE2} were used. In 1992, RFL_{SWE1} and RFL_{SWE2} were equal; in 1993 they differed by 0.5 cm.

Non-channelized hillslope runoff (NHR). The watersheds for each lake have portions which drain directly into the lakes and are not accounted for when lake inflows are measured. Most of these areas are forested, rising only a few hundred meters above the lakes they immediately surround. Unit runoff from these areas was computed as precipitation for Stanley minus the percent lost to evapotranspiration for a mature forest (38% for Hubbard Brook Experimental Forest, NH, Likens et al. 1977).

Lake evaporation. We used the maps of Dunne and Leopold (1978) and Wolman (1990) to estimate lake evaporation for the Sawtooth Valley Lakes. Both of these sources gave annual figures of 70 cm.

Percent difference between *inputs and outputs* to the lakes was calculated as:

$$[(I+P+R)-(O+E)] * [O+E]^{-1} \times 100 \qquad (4).$$

Lake flushing rates. Lake volume divided by annual average outflow yields lake flushing rate. Lake volumes were calculated from hypsographic curves reported in Spaulding (1993). Outflows were integrated through time from our discharge measurements on the outlet creeks.

Lake Nutrient Loading

We computed TN and TP fluxes for each input of the water budget by multiplying nutrient concentration by water quantity. The results were integrated over time to provide volume-weighted annual fluxes. Since we have gauge readings and discharge measurements for more dates than we collected stream samples for nutrient analyses (see below), interpolation was used to determine some of the nutrient concentrations used in the flux computations.

Stream inflows. In 1993, we collected inflow stream water samples weekly for each stream from late-April through September, with additional samples collected during the snowmelt season. In 1992, we collected samples at least 3 times per month from May through September. Additional samples were also taken during baseflow conditions both years.

Depth-integrated stream samples were stored in Nalgene polyethylene bottles. Bottles were rinsed first with 0.1 N HCl and then three times with aliquots of the actual sample. Samples were stored in an ice cooler and then frozen upon return to our field laboratory.

Each sample was analyzed for total nitrogen (TN) and total phosphorus (TP). TN was calculated from the sum of Total Kjeldahl Nitrogen (TKN) and nitrate+nitrite nitrogen (NO_3-N). Unfiltered water was used for TKN and TP analyses; samples analyzed for (NO,-N) were filtered through a 0.45-µm filter. TP samples underwent a persulfate digestion and were then analyzed colorimetrically in our lab (Utah State University Limnology Laboratory) using the molybdate - absorbic acid method. Nitrogen analyses were done by colorimetrically using a Kjeldahl digestion for TKN and the hydrazine method for NO, at the University of California at Davis Limnology Laboratory. Replicates, spikes, and standard solutions were used for quality control/quality assurance.

Precipitation. The nitrogen species in precipitation are NO, and NH,, which are summed to calculate TN. The mean volume-weighted TN concentration of six sites in the National Atmospheric Deposition Program/National Trends Network (NADP/NTN 1994) were used to determine precipitation nutrient concentration for the Sawtooth Valley (Table 2). These particular sites were chosen because of their proximity to the Sawtooth Valley.

The NADP/NTN monitors PO, concentrations, but states that over 95% of the measurements are below the current detection limit (20 µg/L). Since PO, concentrations lower than the detection limit can result in a TP load which is above trace amounts in the Sawtooth Valley Lakes, a concentration of 8 µg/L (for total wet and dry deposition, from Hubbard Brook, Likens et al. 1977) was used to estimate inputs.

We collected our own precipitation samples between April and December of 1993. When analyzed, data from analysis of these samples will allow us to refine the precipitation nutrient fluxes into the Sawtooth Valley Lakes.

Table 2. Sites used from the National Atmospheric Deposition Program/National Trends Network (**NADP/NTN** 1994) to determine TN concentration of precipitation for the Sawtooth Valley Lakes, ID.

Station Name	County	State	Elev. (m)	TN (μg/L)
Craters of the Moon	Butte	ID	1807	880
Headquarters	Clearwater	ID	969	390
Reynolds Creek	Owyhee	ID	1198	690
Smiths Ferry	Valley	ID	1442	440
Lost Trail Pass	Ravalli	MT	2414	270
Yellowstone National Park	Park	WY	1912	640
	Mean TN of all sites:			550

Non-channelized *hillslope* runoff *(NHR)*. We did not measure nutrient concentrations in NHR, but rather estimated them by using the mean annual nutrient concentration for the inflow(s) to each lake (annual nutrient load ÷ annual water discharge). NHR volume was multiplied these concentrations to determine NHR nutrient loads.

Plunging Inflows

The temperatures of the six inflows to **Redfish**, Alturas, Pettit, and Stanley Lakes were recorded by placing **Hobo-temp®** thermographs (Onset Computers, Boston, MA) in the stream channels SO-200 m from the lakes. These thermographs were accurate to 0.2°C and recorded at 36 min to 2.4 hr intervals. Lake temperature profiles were measured every two weeks. Temperatures were recorded from when the lakes turned over in mid-May through the end of October, except for **Redfish** Lake Inflow, where recording started in mid-April.

The slope of the lake bottom at the inlets determines how far plunging water will need to flow (and diffuse) before it reaches water of equal density. We measured these slopes in each lake using a meter line and an echosounder on our boat.

To test whether temperature of the streams accurately predicted the depth to which the water would plunge, we placed fluoroscene dye (Rhodamine WT) in the Pettit South Inflow on 9 October. The dye was visually traced by SCUBA divers who measured the depth to which the dye plunged.

RESULTS

Water budgets

The water budgets in the lakes were dominated by stream inflows and outflows (Table 3). Inflows represented 89-96% of the inputs and outflows were 87-98% of the measured losses (seepage was not measured). The percent difference between inputs and outputs ranged from -13 to 21% in 1992 and from -3 to -1% in 1993, when discharges were more frequently measured.

Water inputs to the lakes in 1993 were between 194% and 268% of the 1992 inputs; output differences were similar. The onset of the snowmelt period was approximately 10 days later in 1993 than 1992 (Fig. 3a). The highest flows during 1993 occurred in late June, due to a rain-on-snow event. Near-base flow was reached in mid-September in 1993 and in mid-August in 1992. Hydrographs for the lakes' inflows and outflows are shown in Appendix 3.

The ice-free season for the lakes was from early-May in 1992 or mid-May in 1993 through early December. Precipitation during the ice-free period (P_f) was 23.7 cm in 1993 and 10.7 cm in 1992. Precipitation inputs when the lakes are frozen (P_i) was 29.2 cm in 1993 and 16.2 cm in 1992. The 30-yr averages of P_i and P_f are 30.2 cm and 18.7 cm, respectively. Thus, for 1993, precipitation for the basin was very close to average.

Lake flushing rates based on 1993 outflows were between 0.3 and 3.0 yr, which were 36-55% of those based on 1992 outflows (Table 4; Gross et al. 1993).

Table 3. Water budgets for Sawtooth Valley Lakes, 1992 and 1993. All quantities are 10^6 m^3, except for A, which is in % (A = [Sum of inputs-sum of outputs] x [sum of outputs]$^{-1}$ x 100). NHR = non-channelized hillslope runoff.

	Gains			Losses		
Lake	Inflows	Ppt.	NHR	Evap.	Outflow	A (%)
		1992				
Redfish	43.2	1.6	3.5	4.3	50.3	-11
Alturas	20.3	0.9	1.3	2.4	23.6	-13
Pettit	7.4	0.4	0.5	1.1	7.4	- 2
Stanley	13.0	0.2	1.0	0.6	11.2	+21
Y. Belly	13.2	0.2	0.3	0.5	14.6	- 9
		1993				
Redfish	83.4	3.3	6.9	4.3	89.9	- 1
Alturas	55.9	1.8	2.6	2.4	59.8	- 3
Pettit	19.8	0.9	1.0	1.1	20.8	- 1
Stanley	30.0	0.4	2.0	0.6	32.4	- 2

Table 4. Computation of flushing rates for the Sawtooth Valley Lakes, based on 1992 and 1993 outflows. Flushing rate = lake volume ÷ annual outflow.

Lake	Lake Volume (10^6 m^3)	Annual Outflow (10^6 m^3)		Flushing Rate (Yr)	
		1992	1993	1992	1993
Redfish	269.9	50.3	89.9	5.4	3.0
Alturas	108.2	23.6	59.8	4.6	1.8
Pettit	45.0	7.4	20.8	6.1	2.2
Stanley	10.4	11.2	32.4	0.9	0.3
Y. Belly	10.3	14.6	--	0.7	--

Nutrient budgets. Nutrient loading to the Sawtooth Valley Lakes varied from 110-930 kg TP and 2,380-11,900 kg TN in 1993 (Table 5). TP nutrient loading to Redfish Lake in 1993 was 930 kg, almost twice as much as the 470 kg in 1992 (Table 6). TN nutrient loading to Redfish Lake in 1993 was 11,900 kg, nearly three times the 4080 kg P in 1992. Nutrient concentrations used to derive these figures are in Appendix 1.

As expected, the majority of the nutrient loading occurred during the snowmelt runoff period. Between 12 May and 30 June, 1993, only 13% of the calendar year, the streams delivered an average of 69% (range: 60-78%) of their annual TP load and 67% (range: 54-79%) of their annual TN load (Fig. 3b; Appendix lb).'

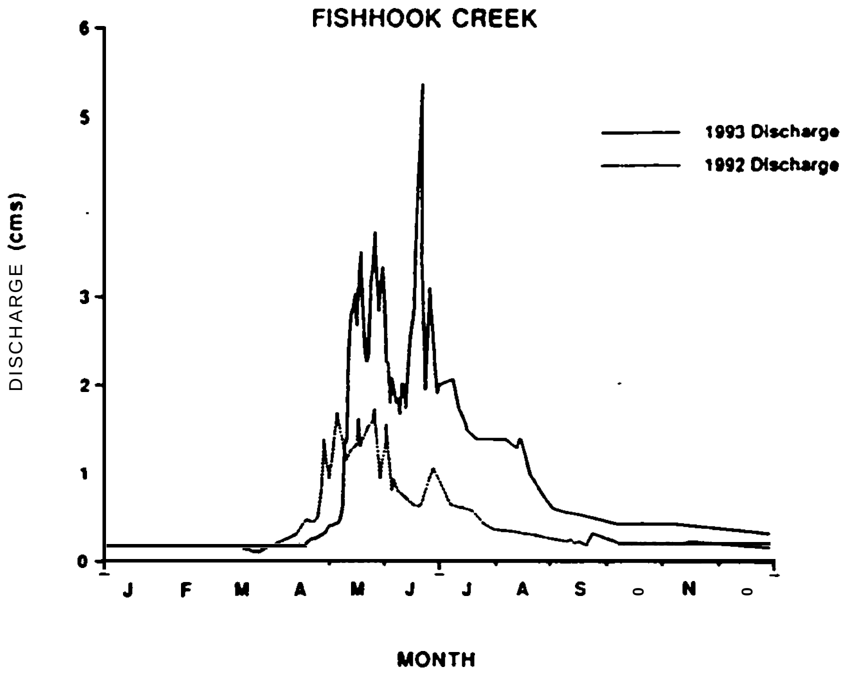
Table 5. Nutrient budgets for the Sawtooth Valley Lakes in 1993. Ppt. = precipitation, NHR = non-channelized hillslope runoff.

	Total (kg)		% of TP			% of TN		
Lake	TP	TN	Inflows	NHR	Ppt.	Inflows	NHR	Ppt.
Redfish	930	11900	89	8	3	78	7	15
Alturas	720	6570	94	4	2	81	4	15
Pettit	110	2380	89	5	6	76	4	20
Stanley	390	3140	93	6	1	87	6	7

Table 6. Comparison of nutrient budgets for Redfish Lake, 1993 and 1992. All figures are in kg. Ppt. = precipitation, NHR = non-channelized hillslope runoff.

Lake	Inflows		NHR		Ppt.		Total	
	TP	TN	TP	TN	TP	TN	TP	TN
1993	830	9330	69	775	26	1790	930	11900
Fishhook Creek	450	4060						
Redfish Inflow	380	5270						
1992	420	2930	38	239	13	910	470	4080
Fishhook Creek	130	950						
Redfish Inflow	290	1980						

Plunging inflows. Mean daily stream temperatures in 1993 ranged from <1 to 6°C in the spring and fall and from 8 to 11°C during midsummer. However, stream temperatures exhibited a diel fluctuation of between 4 and 8°C. With the onset of lake



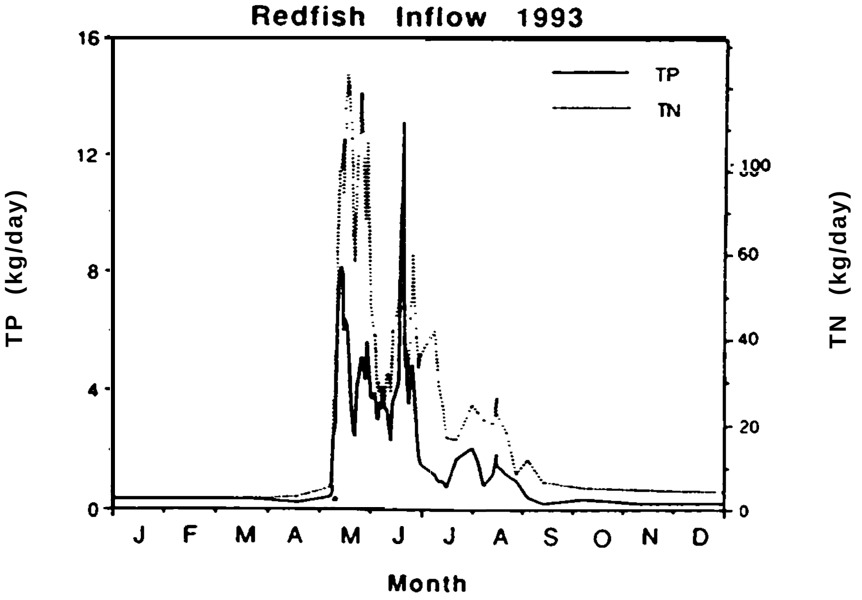


Figure 3. A) Comparison of 1992 and 1993 hydrographs for Fishhook Creek, a tributary of Redfish Lake. B) Temporal variation in total nitrogen and total phosphorus loading from Redfish Creek inflow in 1993. Additional nutrient-loading plots and hydrographs for other streams are shown in Appendices 1B and 3, respectively.

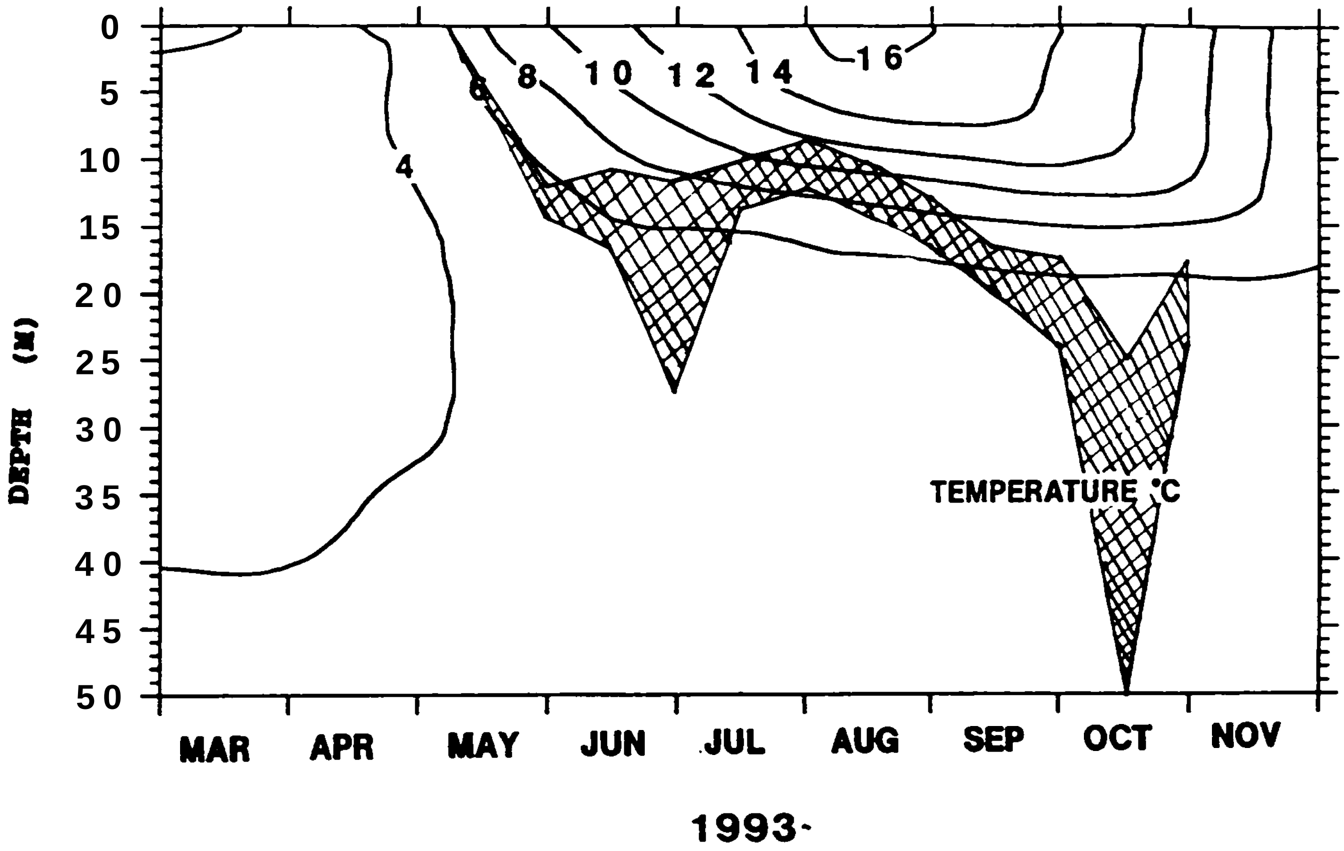
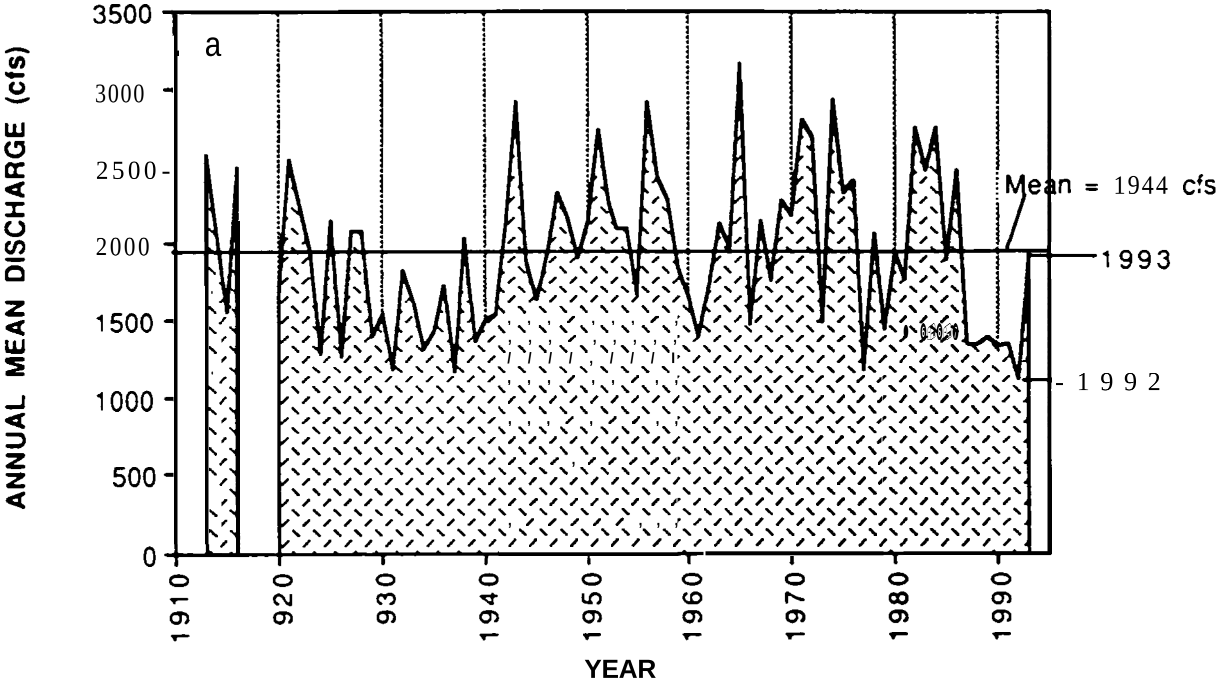


Figure 4. Stream inflow temperatures for Pettit South Inflow overlayed on the Pettit Lake temperature isopleths, 1993, showing at what depth the lake water density was equal to the stream water daily minimum and maximum density. The hatched area represents the depth to which water would plunge based on temperatures in the streams and at different depths in the lakes.

Annaul Mean Daily Flow for the Salmon River at Salmon, ID (19134916, 1920-I 993)



Daily Flow for Salmon River at Salmon, ID (USGS Sta. ID 13302500)

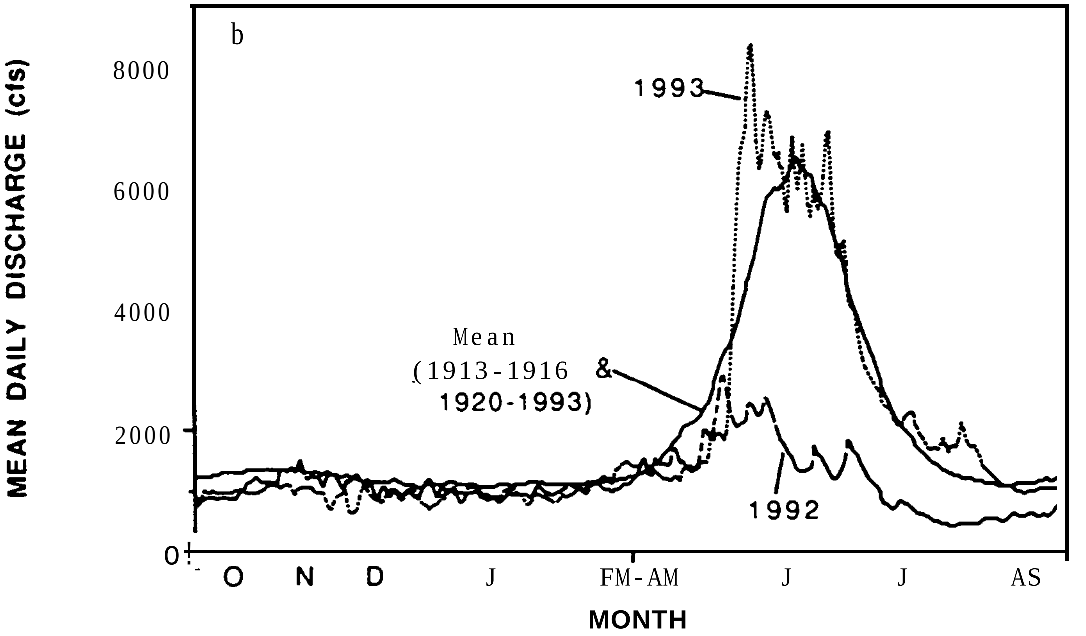


Figure 5. (a) Annual mean daily flows for Salmon River at Salmon, ID, for all years of record (1913-1916, 1920-1993). (b) Comparison of mean, 1992, and 1993 hydrographs for Salmon River at Salmon, ID, USGS Station ID 13302500.

20% less than for a "normal" year, and peak runoff was -3 weeks earlier than normal. The hydrograph in 1993 contrasted sharply with that in 1992, which was the sixth year of a drought period and had the lowest stream flows in 78 years. The differences in hydrograph characteristics between 1993 and 1992 for the Sawtooth Valley streams were similar to those at the USGS gauging station on the Salmon River at Salmon, ID (cf. Fig. 3 and 5b, Appendix 3).

The two- and three-fold increases in TP and TN loading to Redfish Lake in 1993 over 1992 can largely be attributed to increased stream flows (Table 6). The largest increase (four-fold) in nutrient loading was from Fishhook Creek, probably the result of flushing of the wetland system through which Fishhook Creek passes just before entering the lake. Several intermittent channels allowed this area to be flushed by overbank flows, clearing much organic debris which accumulated during the 6-year drought.

Variation in the magnitude of nutrient-loading among the Sawtooth Valley Lakes is partially explained by the differences in sizes of the lakes and their watersheds (Table 1). Lake size directly affects the amount of nutrients contributed by precipitation. Nutrient loading from streams is also affected by watershed size. Nutrient export from the watersheds varied substantially between lakes (Table 7). Redfish Lake's watershed yielded higher TN/m^2 than other basins; this may be due to the flushing of nutrients accumulated in the wetland by Fishhook Creek overbank flow. The calculated area1 yield of TP from Pettit Lake's watershed was less than half of that in the other watersheds, largely because we did not find a pulse in the nutrient concentrations during spring runnoff (Appendix 1).

Area1 nutrient loading to the lakes was influenced by the ratio of watershed area to lake area (Table 7) . A higher watershed to lake area ratio signifies focusing of a greater quantity of nutrient runoff per unit area of lake surface. Accordingly, Stanley Lake had the highest watershed to lake area ratio and the highest TP and TN area1 loading rates. Alturas and Redfish Lakes had similar ratios and loading rates, all of which were 30-50% of the values for Stanley Lake. TP loading to Pettit Lake was low as the result of the low area1 export from the watershed, and its' low watershed to lake area ratio.

Table 7. Nutrient contribution from each stream, normalized by drainage area, and total nutrient loading per surface area, of the Sawtooth Valley Lakes during 1993.

Lake	Watershed ÷ Lake Area	mg/m² of drainase area		g/m² of lake surface	
		TP	TN	TP	TN
Redfish	17.6	10.3	115	0.15	1.93
Alturas	22.4	10.4	82	0.21	1.94
Pettit	16.9	4.3	80	0.07	1.47
Stanley	48.6	11.2	84	0.48	3.87

The measurement of Redfish Lake's nutrient budget in 1993, during what appears to be a "normal" flow year, allows us to compare what percent of nutrients were contributed to the lake by adult sockeye in the past. In 1955, Bjornn et al. (1968) counted 4,361 adult sockeye returning to Redfish Lake. Based on the calculations in Table 8, Scenario 1, if this number of fish returned in 1993, they would have contributed 3% of the TP and 2% of the TN budget. However, the number of sockeye that returned in 1955 was, according to Bjornn et al., "probably only a small fraction of the number which returned during the 1800's....There is no reliable information on the numbers of sockeye salmon spawning in Redfish Lake at those early times." To demonstrate another possible scenario, we have therefore assumed a return of 25,000 adults, which gives a spawner density (4065/km²) similar to that reported for Karluk Lake, Alaska, a system with some limnological characteristics similar to those in Redfish Lake. (Table 8, Scenario 2). If 25,000 spawners returned to Redfish Lake we estimate that they would have contributed 17% of the phosphorus and 9% of the nitrogen coming into the lake.

One must also consider what portion of the nutrients entering the lake are recruited into the aquatic food web. Bioavailable phosphorus entering via streams usually does not exceed 60% of total phosphorus, and is often considerably less (Sonzogni et al. 1982). A similar proportion of the nutrients brought to the lakes by returning fish can be lost to sedimentation, particularly by immobilization of phosphorus in bones and scales (Kitchell et al. 1975; Parmenter and Lamarra 1991).

Table 8. Annual total phosphorus (TP) and total nitrogen (TN) budgets for Redfish Lake, using 1993 inflows as an average year, and including two scenarios of adult sockeye contributing nutrients.

Input:	TP	%	TN	%
Scenario 1: 4,361 adults in 1955 (Bjornn et al. 1968)				
Fluvial inputs	900	94	10,100	83
Precipitation	26	3	1,790	15
Adult sockeye	32[1,2]	3	200[1,3]	2
Total	960	100	12,100	100
Scenario 2: 25,000 adults, pre-1900's, hypothetical				
Fluvial inputs	900	81	10,100	77
Precipitation	26	2	1,790	14
Adult sockeye	190[1,2]	17	1,150[1,3]	9
Total	1,100	100	13,000	100

[1] Using a mean weight of sockeye adults returning to Redfish Lake of 2.2 kg, based on a mean fork length of 561 mm (Bjornn et al. 1968) and the length-weight relationship derived from Burgner (1992).
[2] Based on 0.3364% P in an adult sockeye (Koenings and Burkett 1987).
[3] Based on 10.12% N (% dry mass) in an adult rainbow trout (Parmenter and Lamarra, 1991), and 20.6% dry mass in an adult sockeye after miqration to Lake Babine, B.C. (Brett 1983).

The water budgets for 1993 appear to be more accurate than those for 1992, with inputs and outputs differing by only -3 to -1% in 1993 versus -13 to +21% in 1992. The improved accuracy in 1993 is most likely a result of the greater number of discharge measurements and gauge readings taken (including diurnal comparisons during the snowmelt period) compared to 1992. The agreement between inflow and outflow in 1993 is quite good considering that neither groundwater inflows for non-NHR areas nor seepage losses from the lakes were considered in these budgets.

Very low nutrient loading to the Sawtooth Valley Lakes (Table 7) may be exacerbated because fluvial inputs appear to plunge below the epilimnia of the lakes during much of the year (Fig. 4). This may contribute substantially to the growth of phytoplankton in the deeper layers, but because light intensity in this region is low, many of the nutrients may not be used as effectively as if they entered directly into the epilimnion (see Chapter 2).

Further work is planned on quantifying the spatial and temporal variation in nutrient loading resulting from the plunging inflows. The results from the dye-tracing SCUBA dive in 1993 verified that the Pettit South Inflow was plunging upon entering Pettit Lake, depriving the epilimnion of the nutrient load. However, since the dye-tracing showed that the depth of plunging can not be determined solely by temperature, we are currently considering the use of a reservoir mixing model for this purpose. The mixing model considers the geometry and temperature stratification of the lakes along with the temperature, discharge, and nutrient concentration of the inflows.

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Chapter 2

Limnological Investigations and Hydroacoustic Surveys of Sawtooth Valley Lakes

by

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INTRODUCTION

The Snake River Sockeye salmon, *Oncorhynchus nerka,* begin their anadromous life cycle in central Idaho. Now listed as endangered, these salmon once returned to five mountain lakes in the Sawtooth Valley. Currently, Redfish lake is the only one of the lakes that still has sockeye returning to it. Sockeye spawn in freshwater, and the juveniles spend at least one season growing in the lake before their migration. The decline of the Snake River sockeye is largely attributed to the dams along the Columbia, Snake, and Salmon Rivers, but degradation of spawning and rearing habitats may also contribute.

The work presented here is a continuation of research begun in the fall of 1991 to assess the limnological conditions of the five lakes that historically contained sockeye salmon (Redfish, Alturas, Pettit, Stanley, and Yellow Belly). Examining the physical and biological characteristics of the lakes over this period will allow us to better determine which lakes are potentially better rearing habitats. This research will be useful in developing strategies for releasing juveniles from the broodstock program into the lakes.

Although the Snake River sockeye are endangered, the Sawtooth Valley lakes contain healthy populations of kokanee, *Oncorhynchus nerka.* Consequently, hydroacoustic surveys were conducted in September 1993 to assess abundance, size, and spatial distributions of pelagic fish populations. The surveys were designed to compare kokanee populations among lakes. The data from these surveys will help assess abundances of pelagic fishes and determine depth distributions of kokanee, which will assist in the analyses of the growth potential of sockeye in the lakes.

In 1993 temperature, oxygen, conductivity, light level, water chemistry, chlorophyll, phytoplankton, primary production, and zooplankton were measured in the lakes. Inter-year fluctuations in climate could potentially impact the limnology and consequently the sockeye salmon rearing potential of the Sawtooth Valley lakes. In this chapter we present limnological and hydroacoustic data collected in 1993, a comparatively wet year, and compare these results with data collected in 1992 at the end of a prolonged drought.

METHODS

In 1993 sampling of the Sawtooth lakes began in March when all the lakes were still ice covered. Redfish, Alturas, Pettit, and Stanley were sampled biweekly from mid-May until the beginning of October. These four lakes were sampled again in November, and all but Pettit were sampled in December. The December sampling of Stanley lake was through the ice. Yellow Belly lake was sampled in March, and monthly June through September.

At each lake, we sampled temperature, oxygen, conductivity, light intensity, water chemistry, chlorophyll concentration, and phytoplankton and zooplankton population at a deep station. During the summer we collected additional chlorophyll and zooplankton samples at two nearby stations. During the March, under-ice sampling, all data were collected through a hole sawed through 50-70 cm of ice.

Temperature, Oxygen, Conductivity and Light

A Hydrolab H20 Multiparameter Water Quality Data Transmitter was used to take vertical profiles of temperature, oxygen, and conductivity. Occasionally, a Yellow Springs Instrument Model 58 Dissolved Oxygen Meter was used to measure oxygen and temperature. Seasonal isopleths of temperature and oxygen as a function of depth were calculated using SYGRAPH (Wilkinson (1990).

Vertical profiles of light extinction were measured using a Li-Cor Model LI-1000 DataLogger, a deck cell and an underwater spherical sensor that measured photosynthetic active radiation (400-700 nm; PAR). Measurements were taken at 2 m intervals until 36 m or the bottom of the lake. The extinction coefficient was then calculated as the slope of the regression of ln (% surface intensity) against depth (Wetzel 1983). Water transparency measurements were made with a 25-cm black and white Secchi disk.

Water Chemistry

The lakes were sampled once through the ice in March. Starting at ice-out (mid-May), sampling occurred at two to four weeks intervals, depending on the parameter. The epilimnion was sampled with a 6 m depth-integrating Tygon tube. Samples were also collected from the 1% light level and 3-5 m above the lake bottom with a 4-L Van Dorn bottle. On some dates, additional depths were sampled. The 1% light level is significant because it the nominal depth of the bottom of a lake's photic zone. Samples collected for nutrient analyses were placed in polyethylene bottles which were first rinsed with 0.1 N HCl and then 3 times with aliquots of the actual sample. Nutrient samples were stored in an ice cooler and then frozen upon return to our field laboratory. Any necessary filtration occurred prior to freezing.

Samples for dissolved inorganic nutrient analyses [nitrate+nitrite nitrogen (NO_3-N), ammonia-nitrogen (NH,-N), and soluble reactive phosphate (SRP)] were filtered through a 0.45-µm membrane filter 2-6 hours after collection at our field laboratory. Samples were then frozen until analyzed. Unfiltered samples were analyzed for total phosphorus (TP) and total Kjeldahl nitrogen (TKN). Total nitrogen (TN) was calculated from the sum of TKN and NO,-N. NO,-N, NH,-N, and SRP provide estimates of inorganic nutrients readily available for phytoplankton uptake. TP and TN provide estimates of the reserves of these nutrients in the water column.

TP and SRP were analyzed colorimetrically in our lab (Utah State University Limnology Laboratory) using the molybdate - absorbic acid method. TP samples were first digested with a persulfate digestion. Nitrogen analyses were conducted by the University of California (Davis] Limnology Laboratory using a Kjeldahl digestion for TKN, the hydrazine method for NO,, and the indophenol method for NH,. Replicates, spikes, and standard solutions were used for quality control/quality assurance.

Chlorophyll, Phytoplankton and Primary Production

In most of the lakes chlorophyll samples were collected biweekly from the epilimnion at three stations separated by 200-500 m near the deepest part of each lake. At the central index station additional samples were taken at the 1% light level, and 3-5 m off the bottom. At monthly intervals, samples from 4-13 other depths were collected at the index station. Two 50-ml aliquots per sample were filtered through 0.45-μm cellulose acetate membrane filters. Filters were either temporarily frozen or placed directly into 6-ml of 100% methanol for chlorophyll *a* pigment extraction in the dark for 24-48 hours. The extracts were then analyzed before and after acidification (Holm-Hansen and Riemann 1978) using a Turner model 111 fluorometer. Corrections were made for phaopigments. The fluorometer was calibrated using commercial chlorophyll *a* standards which were verified spectrophotometrically.

Samples for phytoplankton enumeration were collected monthly from three depths at the deepest station in each lake (the epilimnion, 1% light level, and 3-5 m above the bottom). Additional epilimnetic samples were collected at biweekly intervals from some lakes. Samples were preserved using Lugol's iodine solution. A 100-ml aliquot from each sample was filtered through a 0.45 μm mixed-ester filter (Millipore HAWP). The filters were cleared and permanently mounted, according to the method of 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 Likens 1991). Phytoplankton were taxonomically classified as follows: Cyanophyta (blue-green algae), Chlorophyta (green algae), Chrysophyta (*Dinobryon* sp.), Bacillariophyta (Diatoms), and Dinophyta (*Peridinium* sp.).

In *situ* primary production rates (PPR) of phytoplankton were measured with the "C-technique (Wetzel and Likens 1990). Water from each of 8-9 depths was placed into three 25-ml glass scintillation vials, taking care not to expose the plankton to direct sunlight. Each vial was inoculated with 80 μL of 25 μCi/ml of $^{14}CHO_3$. To measure non-photosynthetic ^{14}C uptake, we inoculated one vial from each depth with 150 µL with Diuron (dichloro-phenyl-dimethylurea; DCMU), a photosynthetic inhibitor. The vials were resuspended in the water column in clear acrylic plastic tubes hung from an incubation line. Incubations were normally conducted from 1000 to 1400 hrs (Mountain Standard time). Within 2 h of the end of the incubation the entire contents of each vial was filtered through 0.45 µm cellulose nitrate filters (Micro Filtration Systems) and rinsed with 0.1N HCl. They were then air dried, and subsequently counted by liquid scintillation spectrometry using Readysafe' cocktail. Production rates were calculated by subtracting carbon uptake in the DCMU treatments from the light treatments. Dissolved inorganic carbon was estimated from pH, and alkalinity measurements determined with the Gran procedure (Wetzel and Likens 1990). Productivity in the water column was partitioned into that occurring in the epilimnion and in the lower strata. Because stratification was not clearly defined for much of the summer, we used a nominal depth strata of O-7.5 m for calculating the production that occurred in the epilimnion.

Zooplankton

Zooplankton were normally collected at three stations on each date using a Wisconsin style zooplankton net (35 cm diameter by 80 cm long with 80 µm mesh size) equipped with a messenger-operated closing device. A General Oceanics flow meter modified to prevent reverse flow was used to determine net efficiency and volume sampled. In the shallow Stanley and Yellow Belly lakes, zooplankton were sampled at three stations and over 2 depth ranges: a 10 m to surface tow and a bottom to 10 m tow. Tows were made over three depth ranges in the deeper Pettit, Alturas, and Redfish Lakes: 10 m to surface; 30 m to 10 m, and; bottom to 30 m. The net was rinsed and zooplankton were preserved in 5% sucrose-formalin solution. Revolutions per tow were recorded and densities were corrected for net efficiency and volume sampled.

The zooplankton taxa identified included five cladocera (*Daphnia rosea, Bosmina longirostris, Holopedium gibberum, Polyphemus pediculus*), one calanoid copepod (*Epischuira nevadensis*) and at least two species of cyclopoid copepods. In this report we refer to each species by genus. Zooplankton were enumerated and measured in replicated l-5 ml sub-samples taken with a Hensen-Stempel pipette and placed in a circular counting dish. Individual lengths of each species were measured from each sample and mean length for each taxa was calculated. Biomass was determined for each species using the procedures and linear regression equations described by McCauley (1984) and Koenings et al. (1987).

The diel vertical migration of crustacean zooplankton was examined in Stanley and Redfish Lakes at the beginning of September. Depth-stratified collections were made every 4 h for a consecutive 24-h period with the zooplankton net described above. In Stanley Lake we sampled with 5-m long zooplankton tows distributed over the 25-m water column. In Redfish Lake 5-m tows were made between 0 and 20 m, and then 10-m strata were sampled to the bottom (87 m). Duplicate tows were made in each strata. Each sample was preserved and counted according to methods described above.

Hydroacoustic estimates of Fish Abundance

Hydroacoustic surveys were designed to compare kokanee populations among lakes. As *0. nerka* are present in the pelagic regions of lakes at night (Narver 1966; Rieman and Meyers 1992), when most other salmonid fishes are resting on the bottom, we conducted our surveys during nighttime hours. Surveys occurred during new moon periods in September to avoid problems of assessing fish density when fish move closer to the bottom under moon-lit conditions (Luecke and Wurtsbaugh 1993). Between seven and eleven cross-lake transects were sampled on each of four of the lakes using a Biosonics dual-beam echo sounder. Because acoustic sampling of Yellow Belly Lake in 1992 indicated that virtually no pelagic targets were present during nighttime hours, it was not sampled in 1993.

Pettit, Alturas, Redfish and Stanley Lakes were sampled 14-18 September 1993. Surveys began at approximately 22:00 h and ended between midnight and 02:00 h the following morning. Transects established in 1992 (Beauchamp et al. 1993) were followed during the 1993 survey. The nighttime acoustic data were collected with a BioSonics model 105 echosounder equipped with a 420 Khz dual-beam (6x15") transducer that allowed us to estimate fish sizes. We sampled at a rate of 2 pings per second traveling at a boat speed of 4-6 m/s. The region where fish targets were indistinguishable from the bottom (bottom window) was set at 1.0 m. Data were recorded directly into computer files, as well as being stored on digital audio tape.

Data were processed by counting echoes using dual-beam information processed with Biosonics ESP Dual Beam Processor (Model 281) and software. Targets within 4" of the center axis of the sound pulse were examined for fish target criteria and used for density analysis. In this report we present echo count data of fish targets ranging from -59 to -30 db, which represents fish of approximately 30-420 mm (Love 1971). These targets were divided into three size classes; small fishes (-59 to -51 db, 30-80 mm TL), medium sized fishes (-51 to -43 db, 80-170 mm TL), and larger fishes (170-420 mm). These size classes approximate age-O, age-l and age-2 and older kokanee, respectively. The small and medium size classes will also contain redside shiners, and the large size class will contain piscivorous fishes. In addition to reporting these size classes, we converted each target strength to fish total length using Love (1971). Length-frequency distributions are reported for each lake. Care should be taken when interpreting these size distributions in that acoustic target strengths can easily vary by ± 30% for individual fish. Only echoes that met single-target criteria of the analysis software were used to calculate densities. This procedure may underestimate fish densities, but will not likely affect distribution patterns. TO calculate total pelagic fish abundance we multiplied the hydroacoustics estimates of fish density in individual depth strata by the water volume in each strata (calculated with hypsographic curves; Luecke and Wurtsbaugh 1993).

On 17 September, a daytime survey was conducted on Redfish Lake with an Eagle Accura fish finder to estimate the depth distribution of fish targets during daylight. Previous sampling using the Biosonics echosounder indicated that daytime acoustic surveys sampled few fish, and those that were sampled tended to be in large schools (Beauchamp et al. 1993). In 1993, we used the Eagle echosounder with a wider acoustic beam to sample a greater volume of water. During this survey, the depth of each target, bottom depth, and time of sample were recorded. The relative number of fish sampled in different depth strata were calculated correcting for differences in sample effort among strata. No estimates of absolute fish density were calculated.

RESULTS

Temperature, Oxygen, Conductivity & Light

Seasonal lake temperatures were lower in 1993 than 1992 (Figs. l-2, and Appendix A Figure 1). In 1993 maximum surface temperatures barely edged above 16°C in early August. In 1992 temperatures were above 16°C for over a month, and peaked at over 18°C. In 1993 the lakes began to thermally stratify in mid-May but in contrast to 1992, a distinctive epilimnion did not develop until mid-August. The final depth of the thermocline was similar to 1992: either between 8-20 m (Redfish, Alturas, and Pettit) or 8-15 m (Stanley and Yellow Belly).

Dissolved oxygen levels in 1993 were comparable to those in 1992 (Fig. 1, Fig. 2, and Appendix A, Fig. 1). Surface oxygen levels averaged between 7 mg/L and 10 mg/L in all lakes. However, oxygen levels at depth varied between lakes. Oxygen levels in Redfish rarely dropped below 5 mg/L (the oxygen concentration below which salmonids show behavioral avoidance), even at the bottom. In

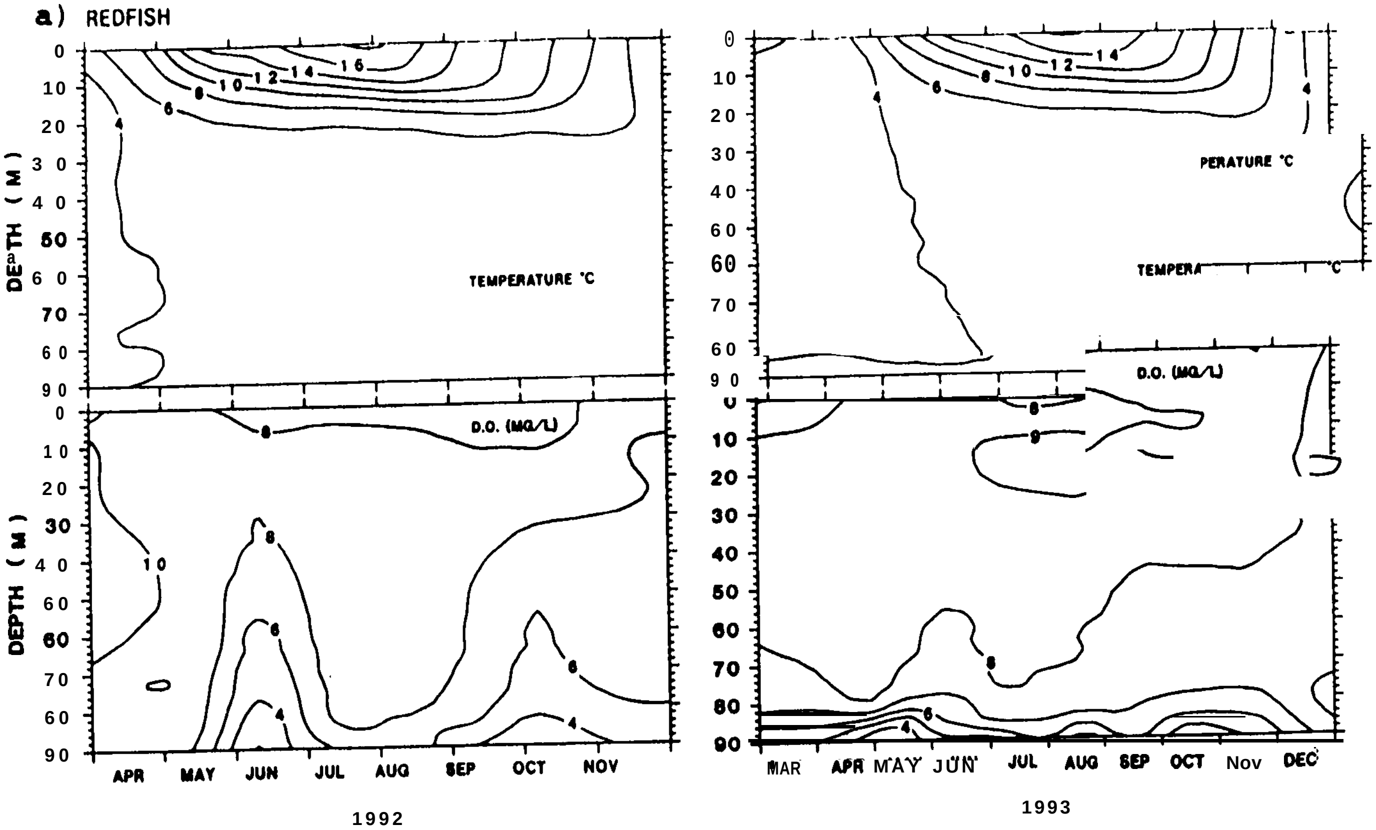
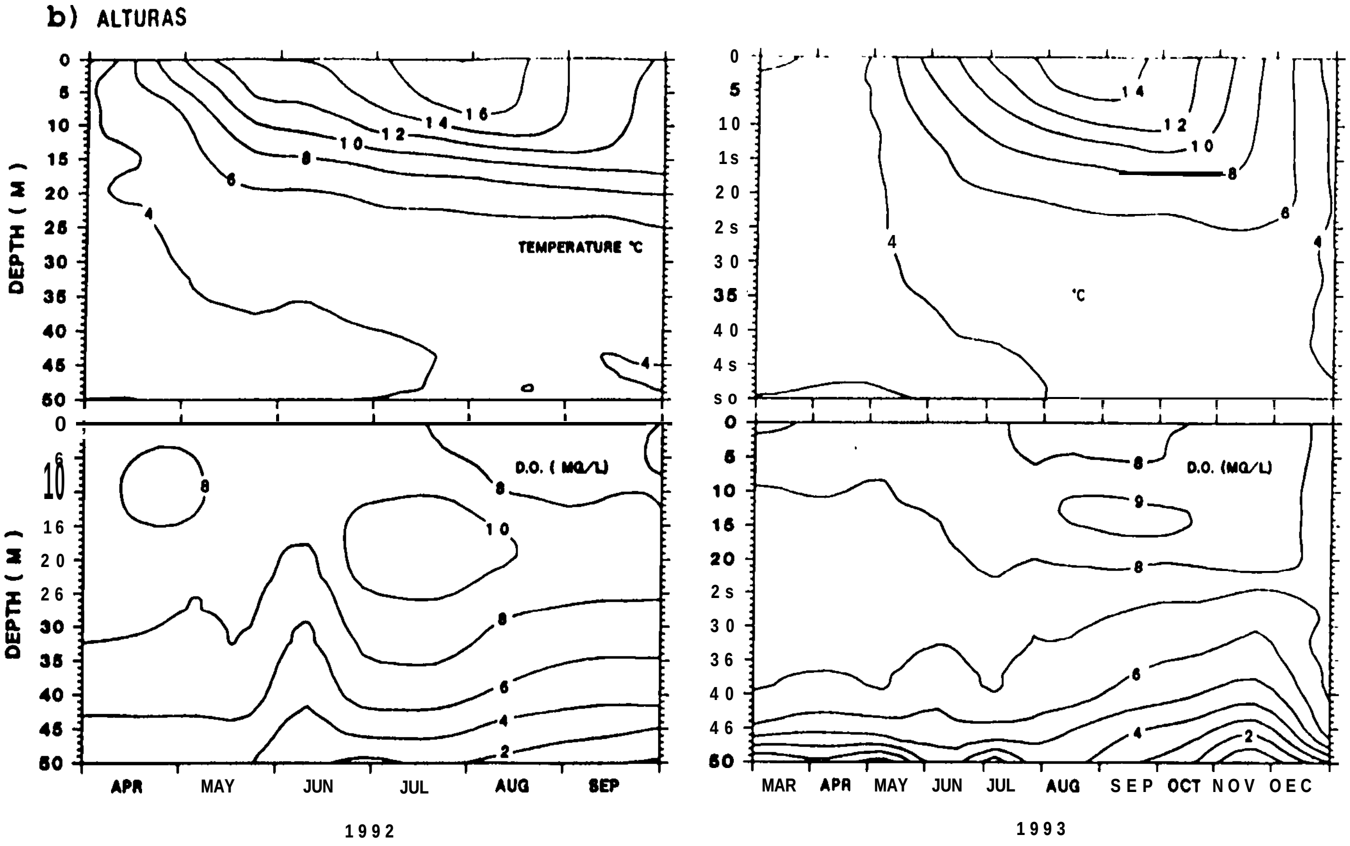


Figure 1. Temperature and oxygen isopleths for the Sawtooth Valley Lakes in 1992 and 1993. a) **Redfish** **b)** Alturas **c)** Pettit **d)** Stanley **e)** Yellow Belly.



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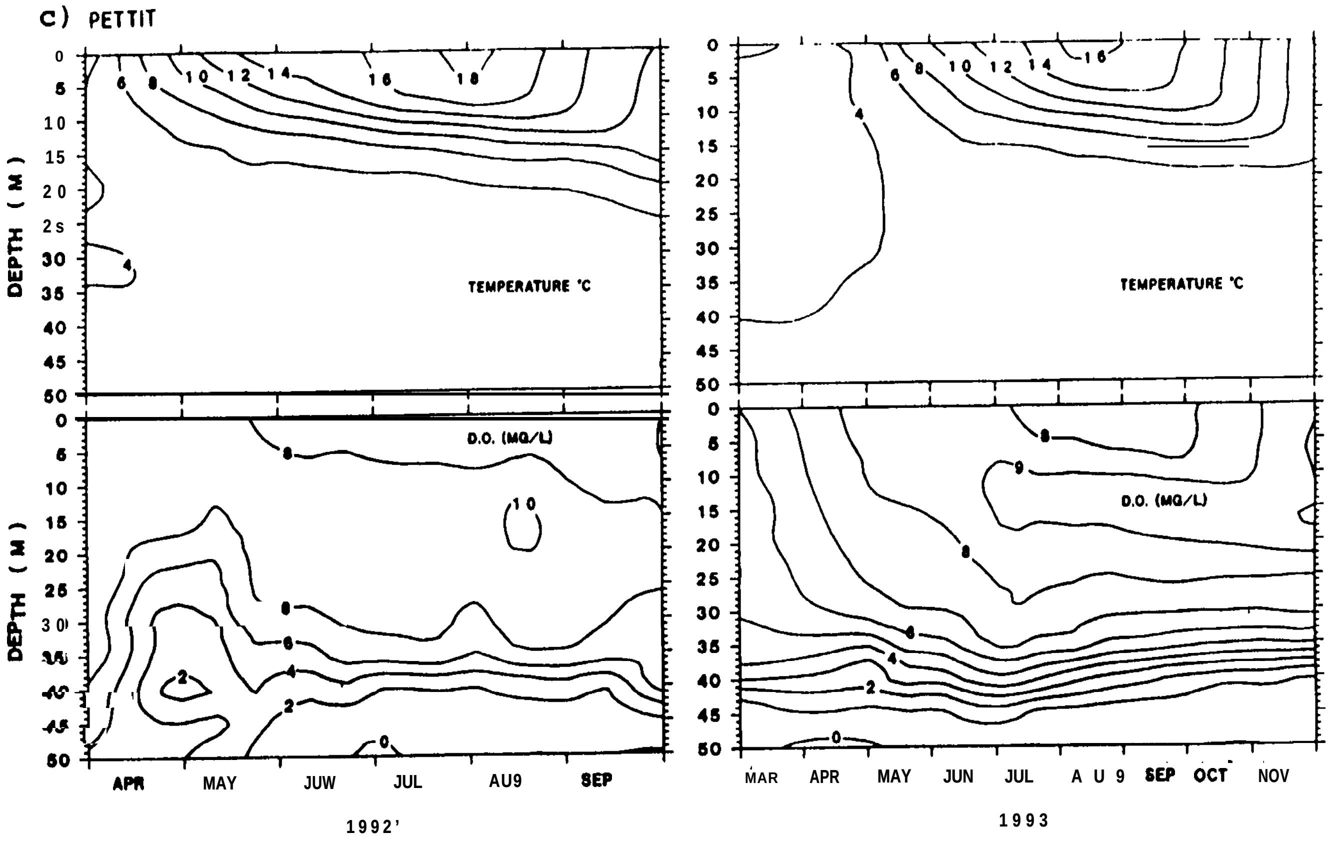
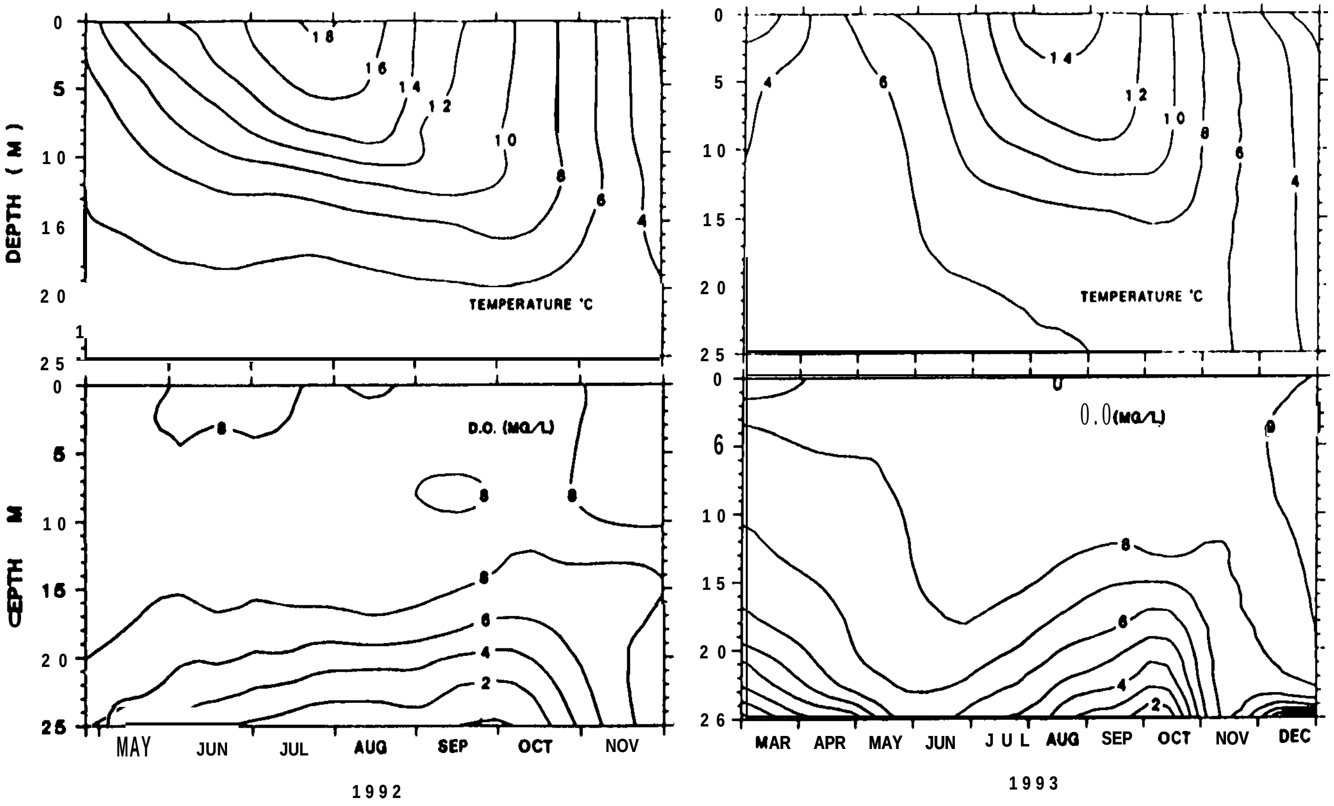


Figure 1. Continued.

d) STANLEY



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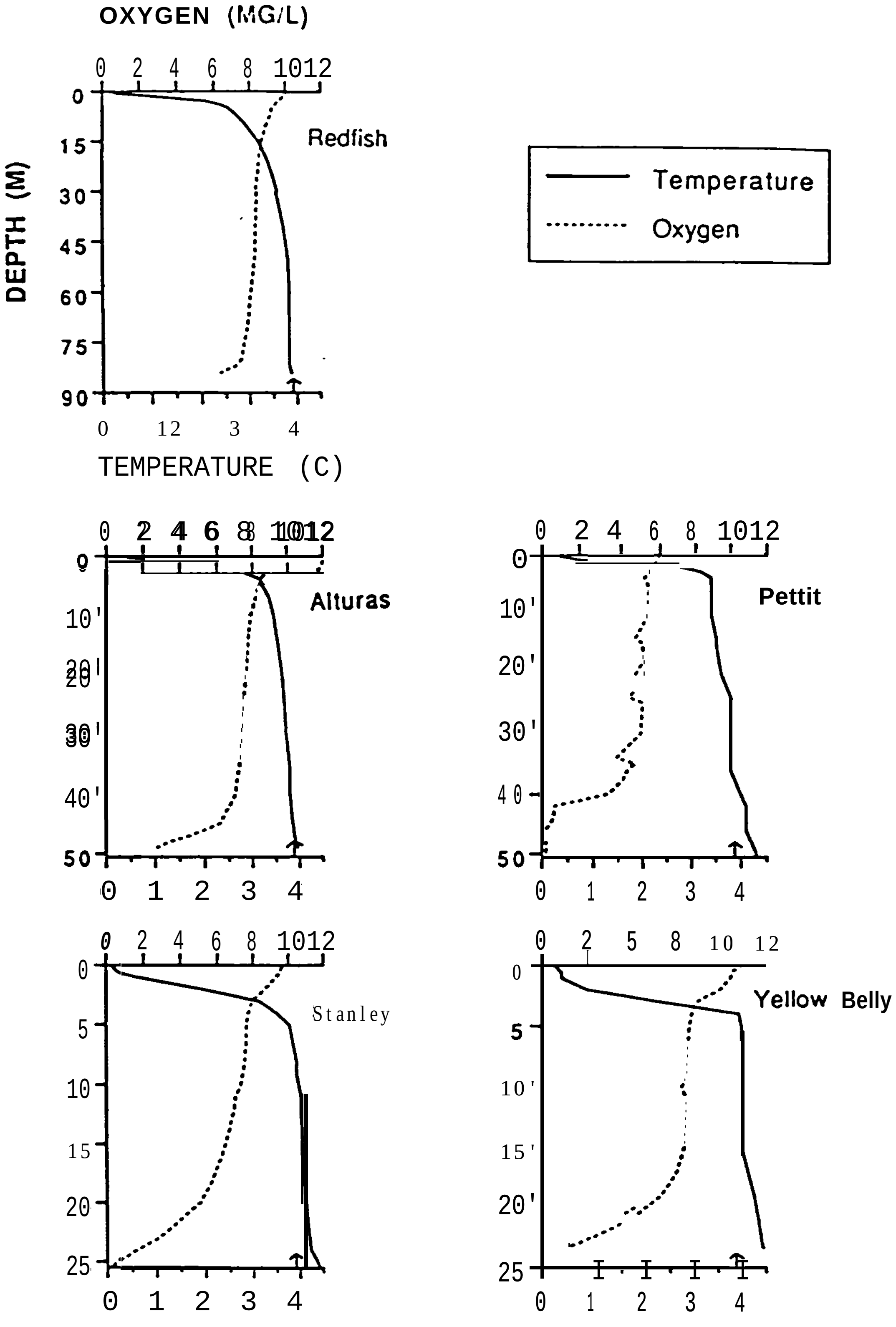


Figure 2. a) Temperature and oxygen profiles ***for*** the Sawtooth Valley Lakes during ice-cover in March (19-21 March 1993). Arrows indicate the temperature of the maximum density of freshwater at depth for each lake.

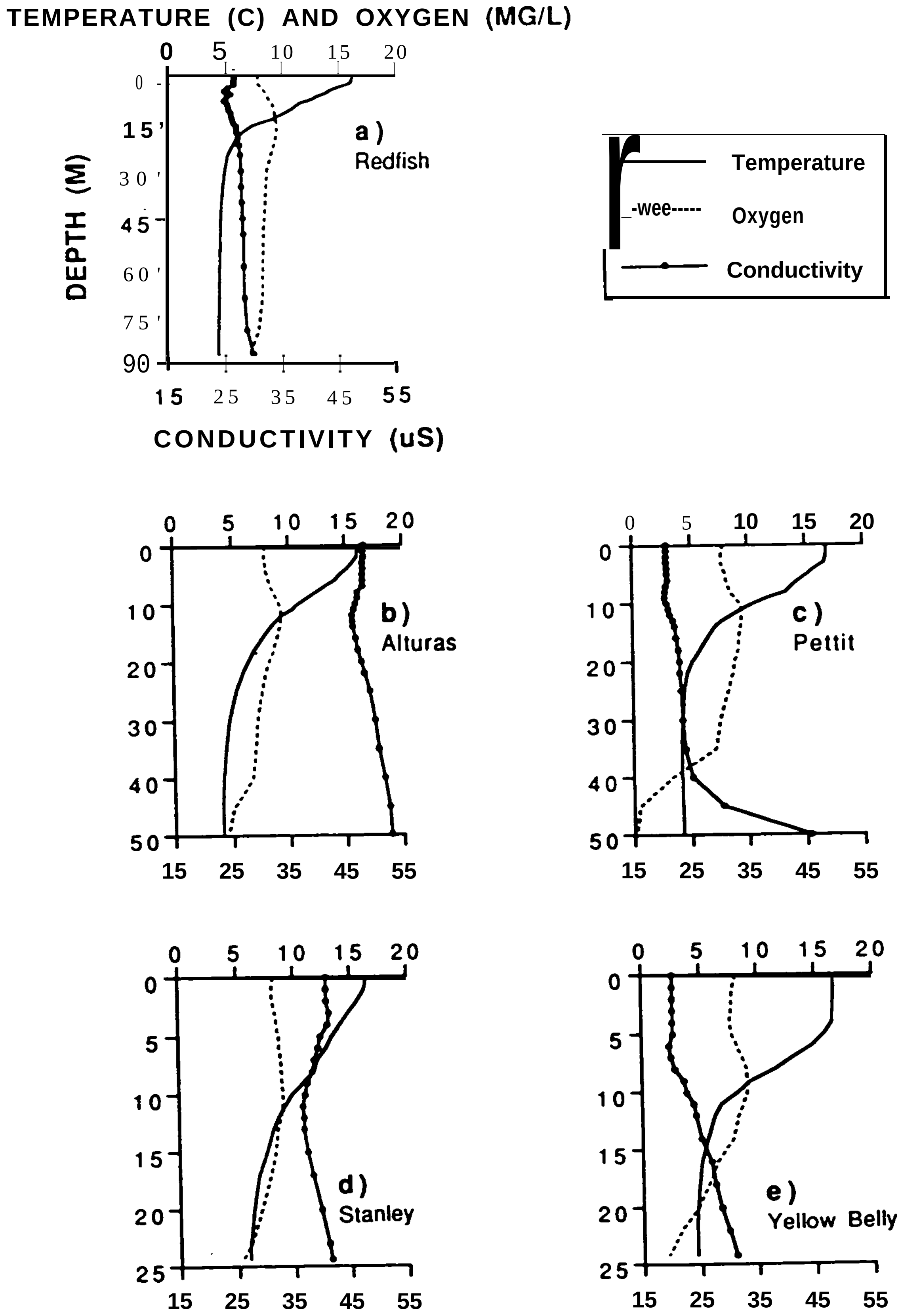


Figure 2. Temperature, oxygen, and conductivity profiles for the Sawtooth Valley Lakes during midsummer (5-9 August 1993) in: a) **Redfish,** b) Alturas, c) Pettit, d) Stanley, and e) Yellow Belly Lakes.

Alturas, Stanley, and Yellow Belly the oxygen was often below 5 mg/L close to the bottom. Pettit had oxygen levels below 5 mg/L as shallow as 29 m, and very low oxygen levels (<3 mg/L) 10-15 m off the bottom. These low oxygen levels were maintained-in the lakes for most of the year.

A metalimnetic peak in oxygen was present in all five lakes during at least part of the summer stratification (Fig. 1). Redfish and Pettit Lakes had the highest metalimnetic oxygen levels (10.0 and 10.2 mg/L, respectively). The oxygen bulges lasted from mid-July until mid-October. The oxygen peak in Stanley was below 9 mg/L, and was not well developed. In all of the lakes the metalimnetic oxygen maxima started near the top of the thermocline, and descended as the summer progressed to a maximum depth of 18 m in Redfish Lake.

Conductivities varied among the lakes (Pigs. 2 a-e). Alturas and Stanley Lakes had conductivities from 40 µS to 50 µS. Redfish and Yellow Belly had conductivities ranging from 20 µS to 30 µS. In these lakes, conductivity remained constant for most of the water column before it increased within 1-5 m of the bottom. In Pettit Lake conductivities were between 20 and 25 µS in the top 35 m of the water column, but increased to nearly 50 µS below this depth. Conductivities increased dramatically as far as 10-15 m off the bottom.

Temperature profiles measured under the ice in March of 1993 indicated that Pettit, Stanley and Yellow Lakes were meromictic (salt-stratified) below, 35 m, 20 m, and 17 m, respectively. Temperatures in the deep waters of these lakes were above 3.96°C (the maximum density of fresh water), indicating these layers must contain dissolved salts that would increase the overall density. Oxygen levels were < 4 mg/L in these deep waters (Fig. 2a). Temperature, oxygen and conductivity profiles measured in May and June indicated that Stanley Lake had mixed to the bottom, but that Pettit and Yellow Belly Lakes did not turn over.

Although the Sawtooth Basin lakes were generally very clear, there was a strong seasonal pattern in water transparency. Water transparency was lowest in the spring and increased through the summer and then decreased in the fall. The depth of the compensation point (1% of surface intensity) gradually deepened over the summer (Fig. 3). The compensation point reached 26 and 32 m in Pettit and Redfish Lakes, respectively. In Alturas Lake compensation depths were between 17 and 22 m, while in Stanley and Yellow Belly Lakes the photic zone usually extended to between 12 and 17 m. The depth of the compensation point was usually 1.5 - 2.5 times the Secchi depth. The 1% light extinction plane was related to Secchi depth (m) by the following equation:

$$1\%\ \text{light extinction} = 0.92*(\text{Secchi Depth}) + 8.5 \qquad R^2 = 0.61;\ n=28$$

This equation was different than in 1992, when Secchi depths predicted a deeper 1% light intensity.

Secchi depths varied in a similar pattern to the compensation depth (Fig. 4). Stanley Lake had the lowest water transparency, with Secchi depths ranging from 3 m in May to 10 m in September. In the clearest waters, Pettit and Redfish Lakes, transparencies ranged from near 9 m in June to near 20 m in the fall. In 1993 Secchi depths were shallower than in 1992 during early summer, but deeper in the fall. Mean summer Secchi depths were shallower in 1993 than in 1992 (Table 1).

Nutrients

In 1993 TP and TN concentrations at spring overturn ranged from 9-16 and 63-167 µg/L, respectively (Table 2). These levels were higher than in 1992, except for TN in Redfish Lake, which was similar in the two years. Mean seasonal TP and TN concentrations in 1993 ranged from 5.9-8.3 and 51-110 µg/L, respectively. TN:TP ratios of the five lakes were lowest in Redfish Lake in both 1992 and 1993.

Mean seasonal concentrations of dissolved nutrients were low in both years, with mean soluble reactive phosphorus (SRP) and nitrate+nitrite nitrogen (NO,-N) for the lakes ranging from 1.0-2.3

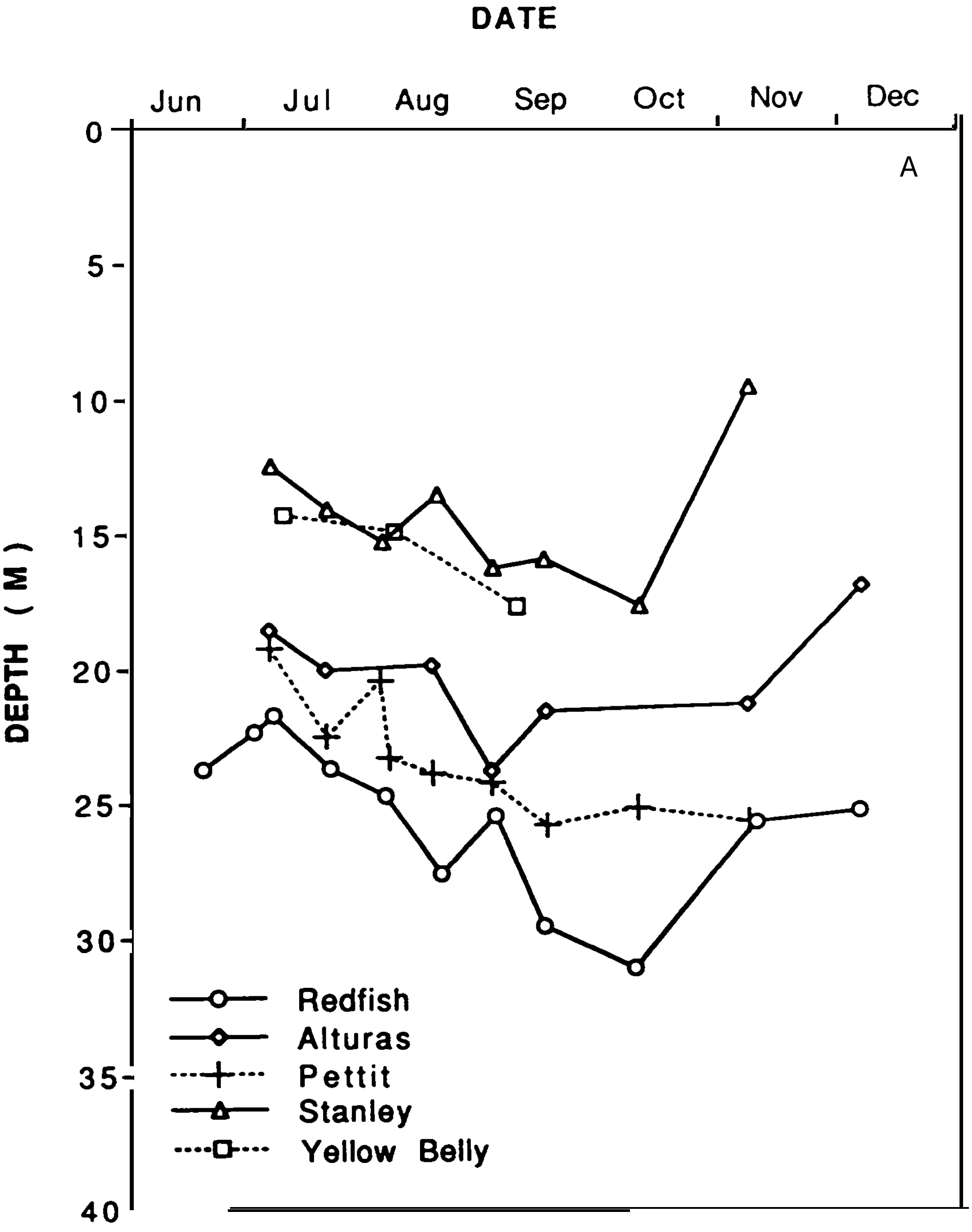


Figure 3. Seasonal changes in the depth of the compensation point (1% of surface light intensity) in each of the Sawtooth Valley Lakes in 1993.

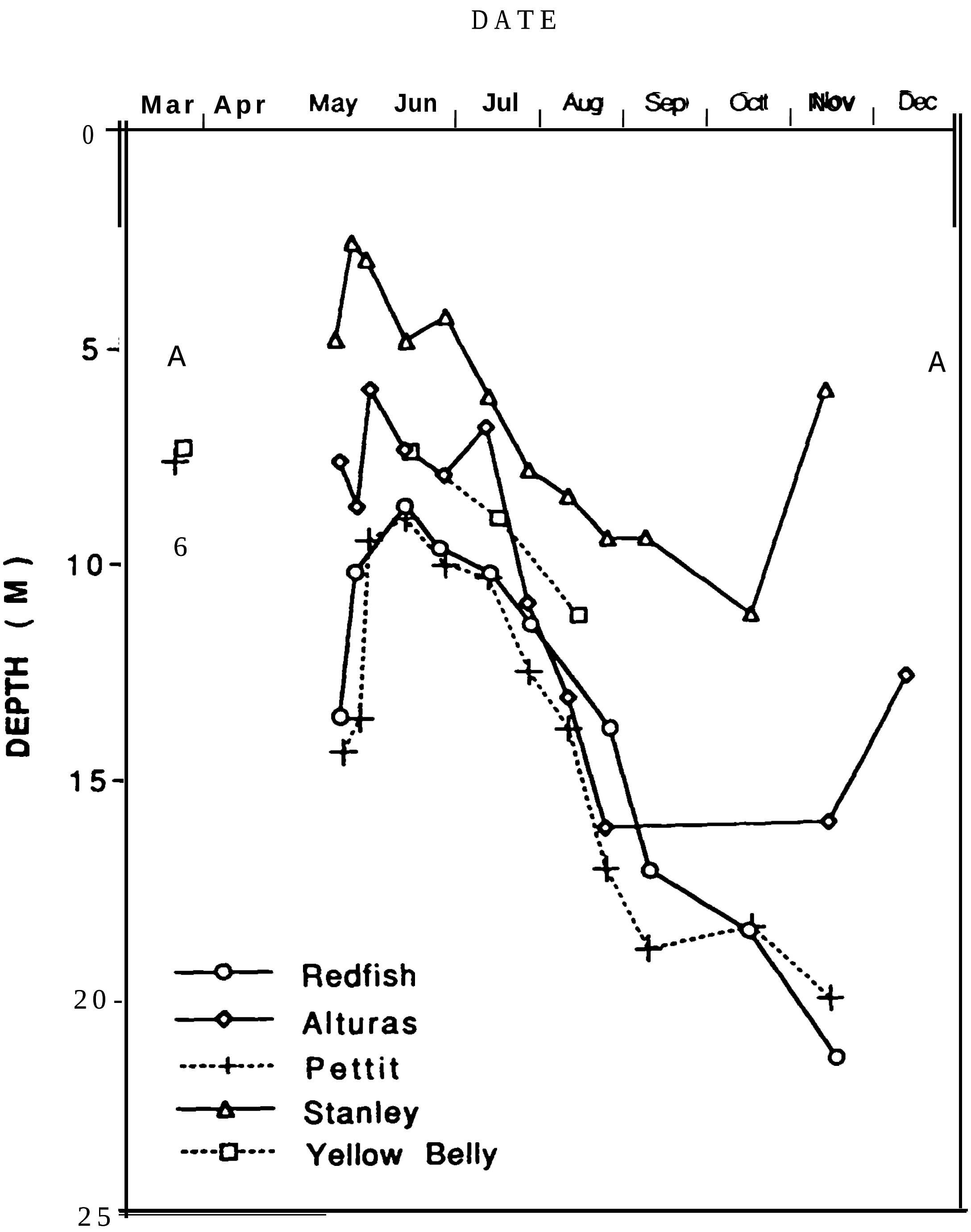


Figure 4. Seasonal distribution of Secchi depths for the Sawtooth Valley Lakes during 1993.

and 2.1-9.4 µg/L, respectively (Table 2; Appendix 1). NH,-N was also low, ranging between 2.6-3.2 for Redfish, Alturas, and Pettit Lakes.

Chlorophyll, Phytoplankton and Primary Production

Chlorophyll a concentrations were 35-75% higher in the Sawtooth Valley Lakes in 1993 than in 1992. In 1993, mean summer epilimnetic chlorophyll a levels in the five lakes ranged from 0.54-1.05 µg/L, compared to 0.40-0.60 µg/L in 1992 (Table 1). According to mean epilimnetic chlorophyll levels, the lakes ranked from highest to lowest in the same order in 1993 as in 1992: Stanley, Alturas, Yellow Belly, Redfish, and Pettit.

Spring overturn chlorophyll *a* levels were slightly higher in 1993, ranging from 0.85-2.85 µg/L, compared to 0.80-2.38 µg/L in 1992 (Fig. 5). The seasonal progression of epilimnetic chlorophyll *a* was similar in 1993 to 1992: the highest values for the ice-free season occurred just after spring overturn in May, followed by a gradual decline to a summer minimum (between late July and early August), after which levels increased (Fig. 5). However, in 1993, the summer lows occurred an average of 10 d later for each lake.

Each lake had its highest chlorophyll *a* levels in the hypolimnion, near and frequently below the 1% light level. The deep chlorophyll layer (DCL) of the lakes deepened and became thicker as the growing season progressed (Fig. 6, Appendix A Figures 2-5). The mean value of the DCL peak during the growing season for each lake ranged from 2.40 to 6.57 µg/L - 130-940% greater than the mean epilimnetic chlorophyll *a* concentration (Table 3). The lakes with the highest epilimnetic chlorophyll *a* concentrations, Stanley (1.05 µg/L) and Alturas (0.76 µg/L), had the lowest mean DCL peaks, 2.40 and 2.70 µg/L respectively. Conversely, the lakes with the lowest epilimnetic chlorophyll *a* concentrations, Pettit (0.54 µg/L), Yellow Belly (0.61 µg/L), and Redfish (0.63 µg/L), had the highest mean DCL peaks of 3.31, 6.57, and 3.09 µg/L respectively. The DCL persisted late into the

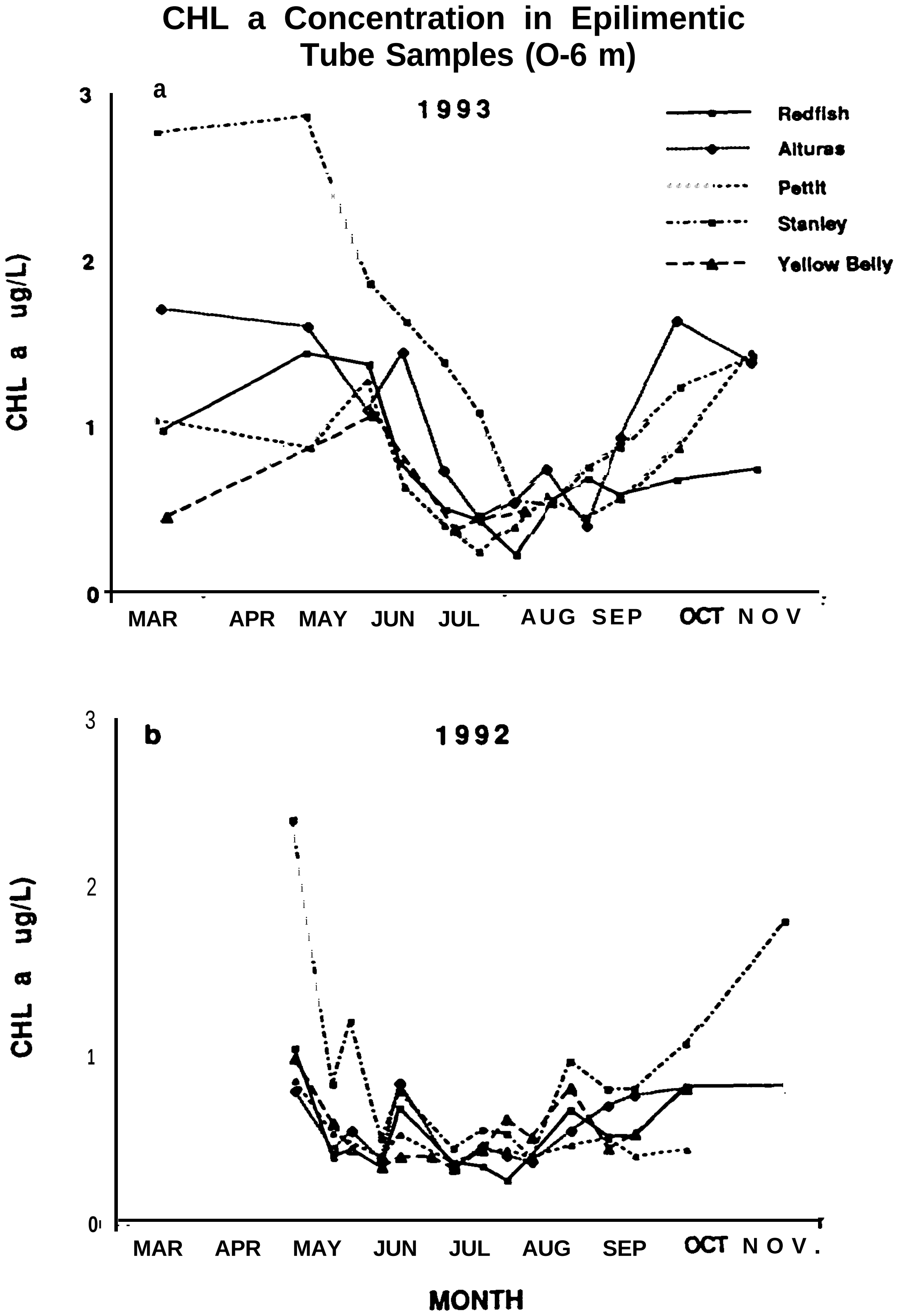


Figure 5. Seasonal changes in the epilimnetic chlorophyll a levels **for** the Sawtooth Valley Lakes in 1992 and 1993.

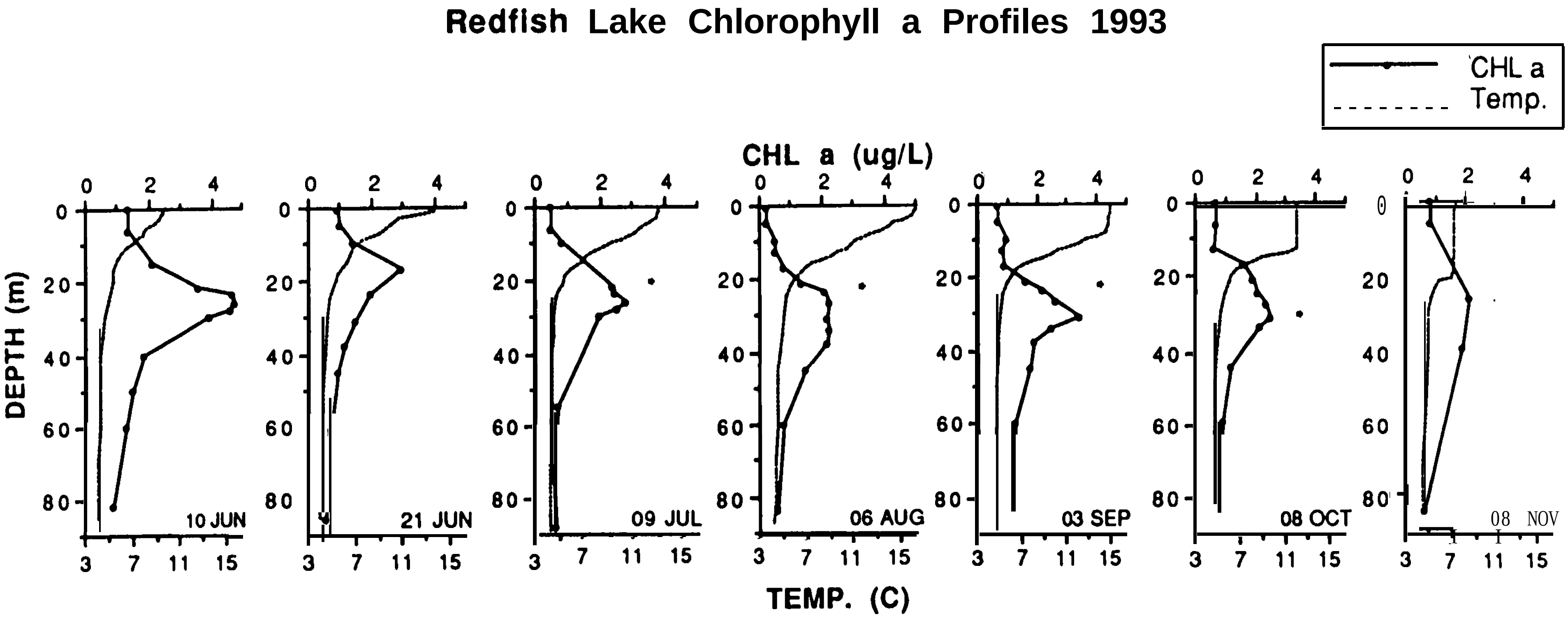


Figure 6. Chlorophyll a and temperature profiles for **Redfish** Lake, 1993. The '*' represents the 1% light level.

growing season, eroding as the epilimnion deepened and each lake approached fall overturn.

The taxa of phytoplankton observed in the Sawtooth Valley Lakes in 1993 are listed in Table 4. Mean equivalent spherical diameters of the taxa listed range from 4.0-21.7 µm, considered to be a size fraction grazable by zooplankton. However, some of the diatoms occurred in chains or colonies (e.g. Melosira, *Tabellaria*, and *Asterionella*). In Alturas, Pettit, Stanley, and Yellow Belly Lakes, Chlorophyta species dominated the biovolume for most of the sampling period (Figs. 7-11). *Chlorella* sp., Oocystis sp., and smaller Chlorococcales spp. comprised most of this; *Chlamydomonas* sp. and the Desmidiaceae spp. contributed minor amounts. In Redfish Lake, diatoms, *Dinobryon* sp. and *Peridinium* sp. made up a substantial part of the phytoplankton community along with the Chlorophyta.

Mean epilimnetic phytoplankton biovolumes were higher in 1993 than in 1992, except in Yellow Belly Lake (Pig. 12). In Yellow Belly Lake, Chlorophyta biovolumes were higher in 1993 than 1992, while *Dinobryon* biovolumes were higher in 1992 than 1993. This was most likely responsible for the mean chlorophyll a value being higher in 1993 than in 1992, even though total algal biovolume was not. Also, note that *Peridinium* (Dinophyta) were observed in the lakes in 1993, but not in 1992.

As the growing season progressed, greater algal biovolumes were supported in the deeper waters of the DCL, near the 1% light level (Figs. 7b-11b), than were supported by the epilimnetic waters. In all lakes except Redfish, this deep algal stratum consisted primarily of *Oocystis* and *Chlorella*. In Redfish Lake, *Dinobryon* and the diatoms *Synedra* and *Tabellaria* made up a considerable portion of the biovolume of this stratum.

The major diatoms in the lakes (*Synedra*, *Cyclotella*, Melosira, and *Tabellaria*) occurred early in the growing season in the epilimnion. By late July, diatoms mainly occurred in samples from the 1% light level or lake bottom. *Dinobryon were* also less numerous later in the growing season and did not persist as long as

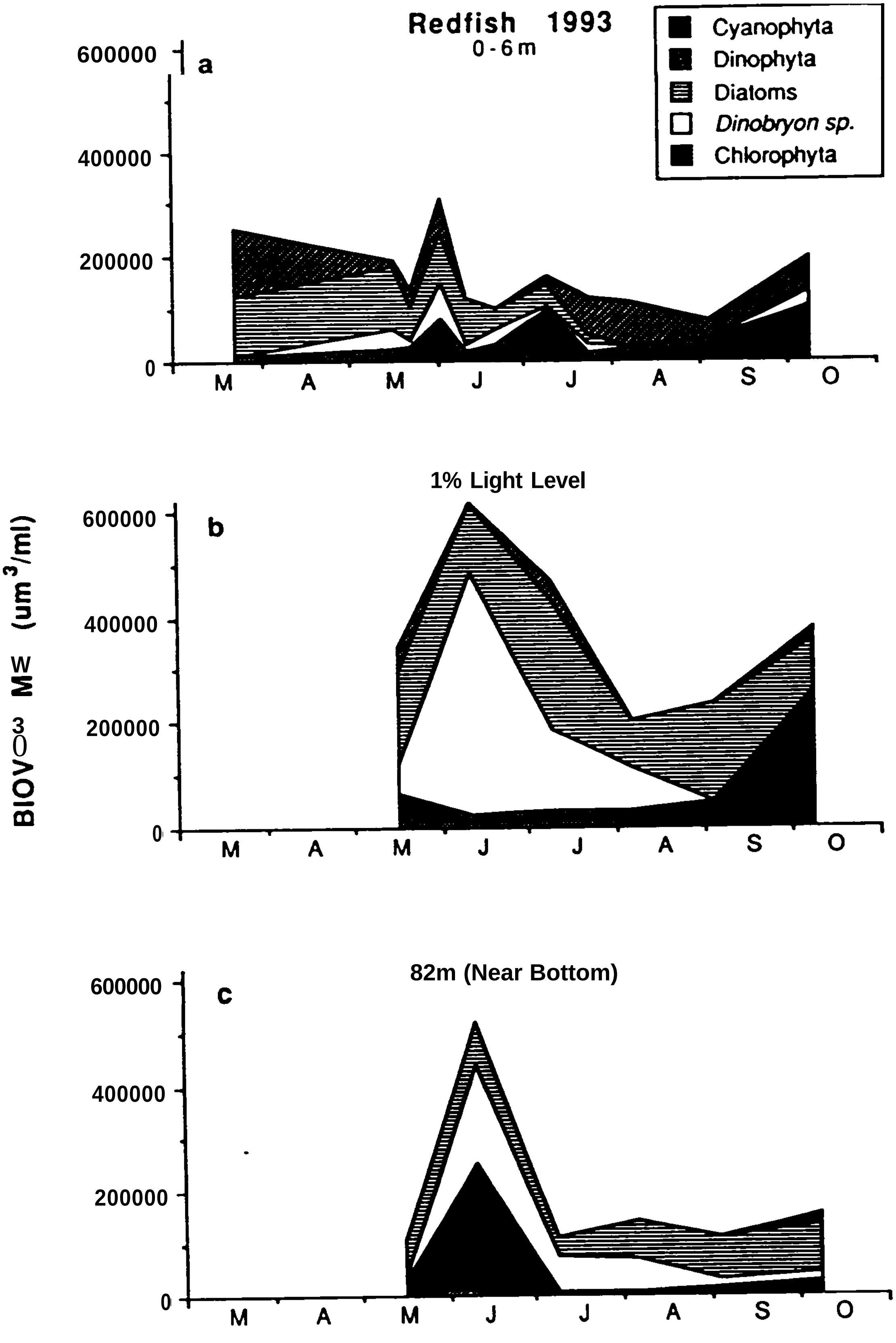


Figure 7. Seasonal changes in the biovolume of different phytoplankton **taxa** in **Redfish** Lake, 1993. **a)** epilimnetic water **b)** the 1% light level **c)** 82 **m** (near-bottom).

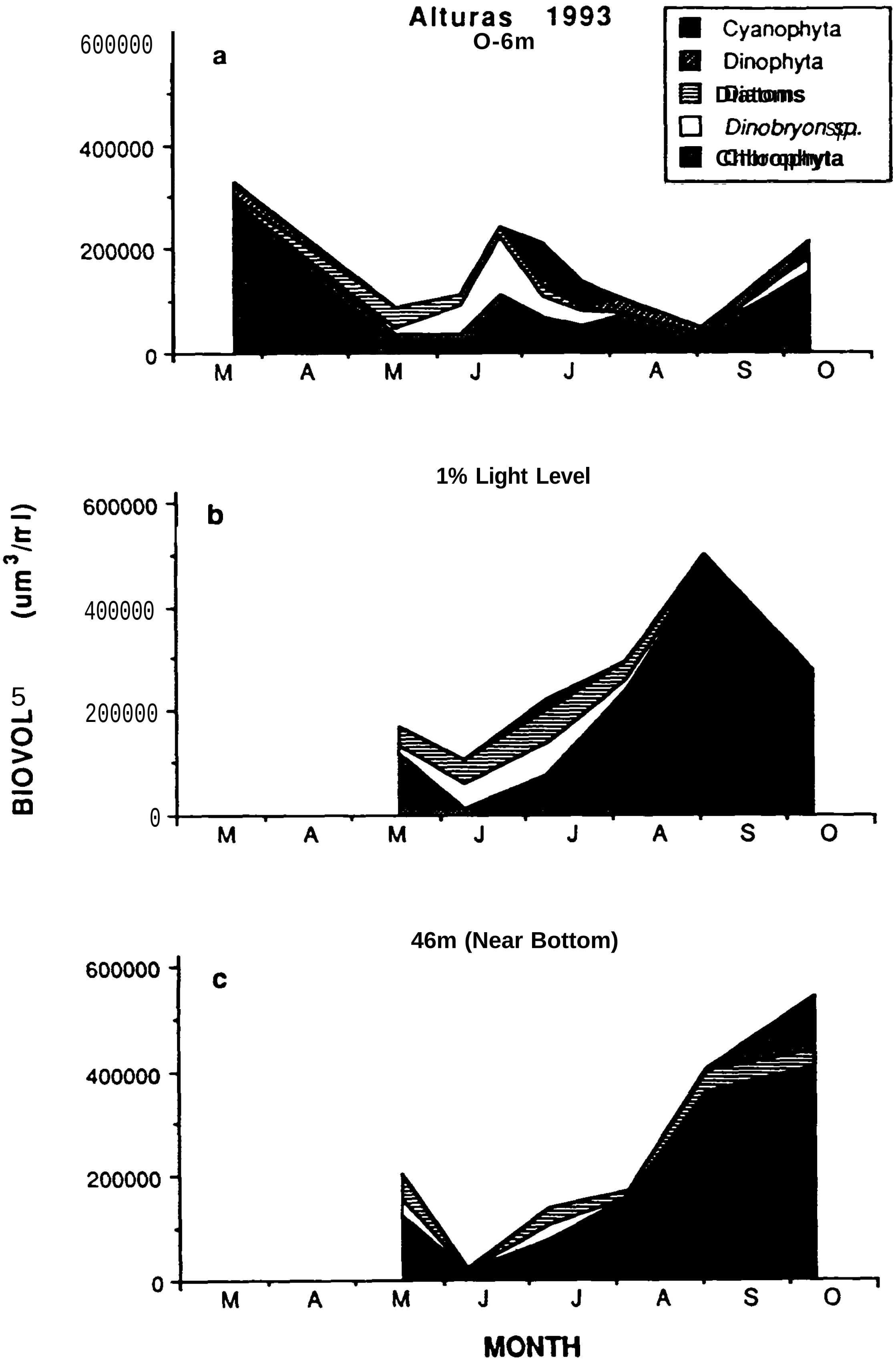


Figure 8. Seasonal changes in the biovolume of different phytoplankton taxa in Alturas Lake during 1993. a) epilimnetic water b) the 1% light level c) 46 m (near bottom).

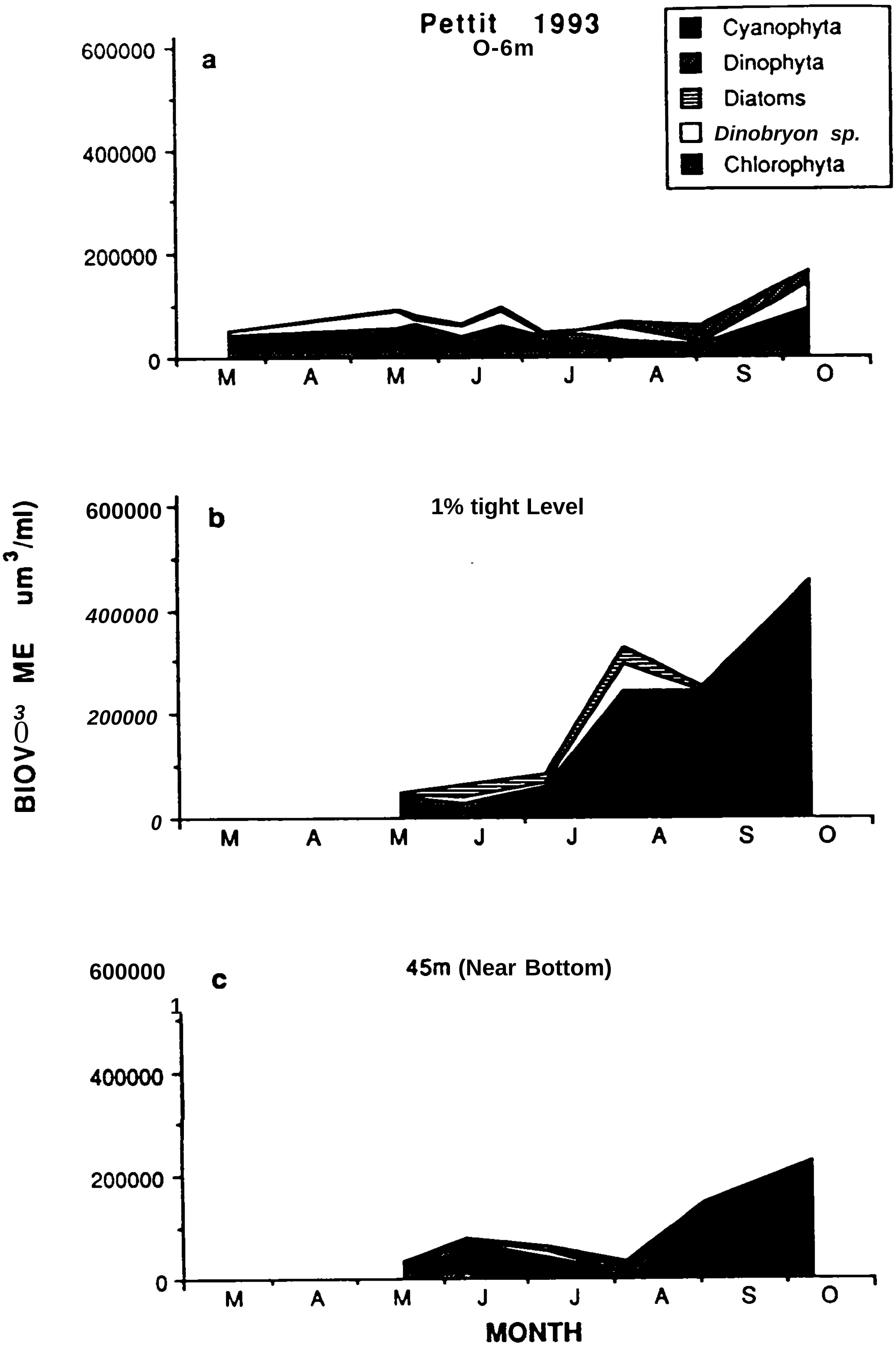


Figure 9. Seasonal changes in the biovolume of different phytoplankton **taxa** in Pettit Lake, 1993. a) epilimnetic water b) the 1% light level c) 45 m (near bottom).

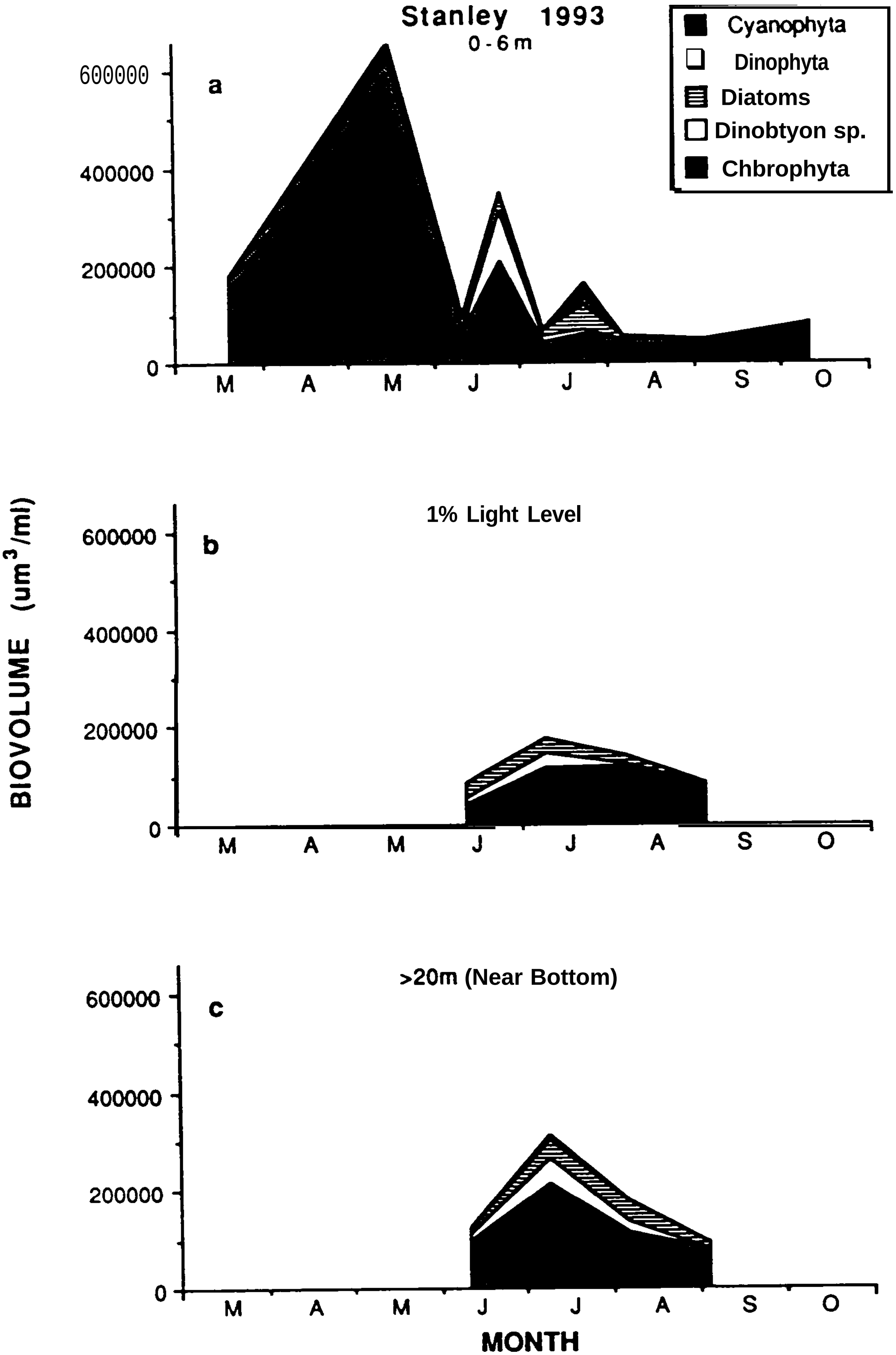


Figure 10. Seasonal changes in the biovolume of different phytoplankton **taxa** in Stanley Lake, 1993. a) epilimnetic water **b)** the 1% light level **c) >20 m** (near bottom).

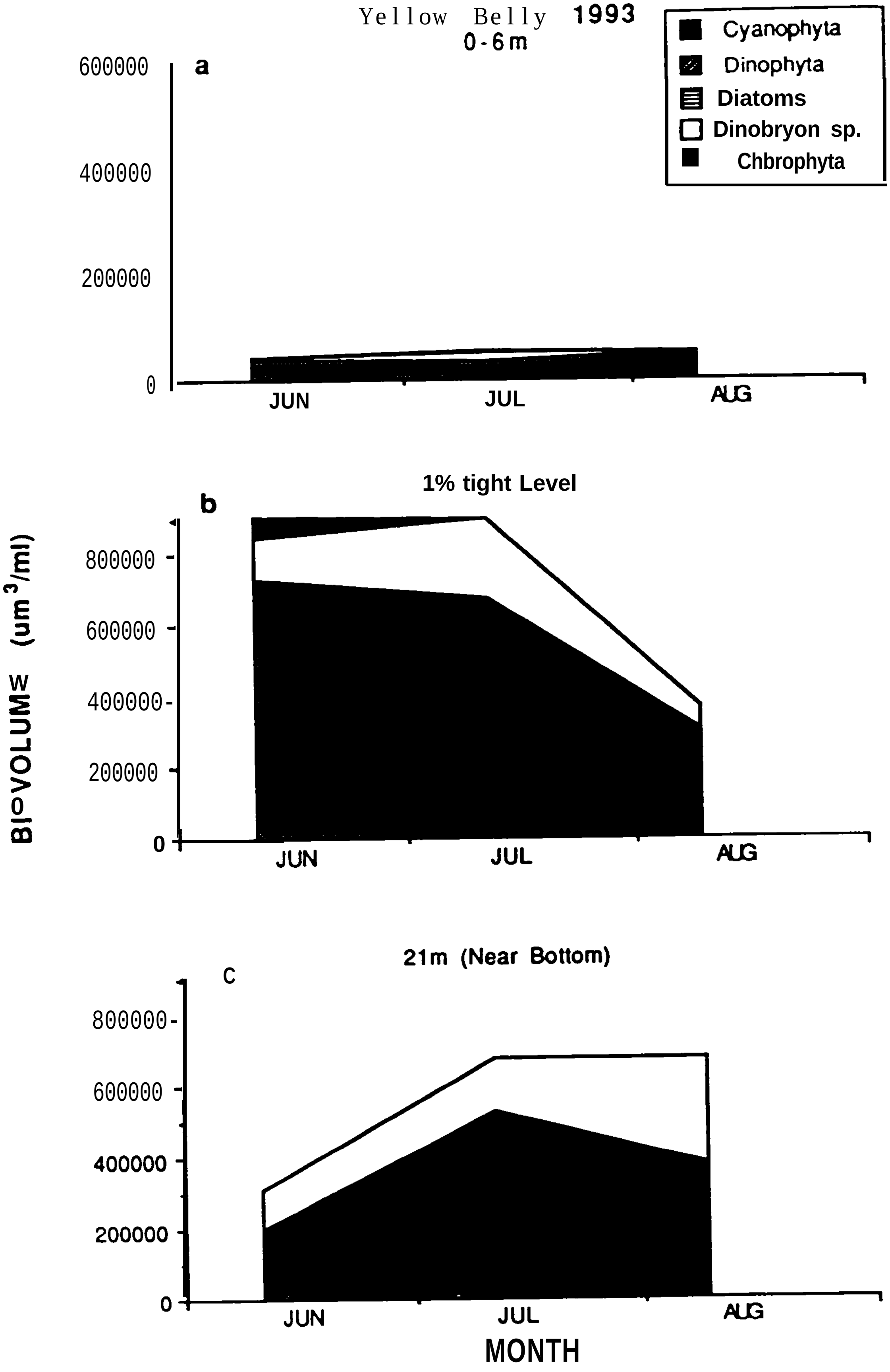


Figure 11. Seasonal changes in the biovolume of different phytoplankton taxa in Yellow Belly Lake, 1993. a) epilimnetic water b) the 1% light level c) 21 m (near bottom). Note that the range of the biovolume axes for b) and c) are larger than that of a) and the other lakes (Figures 7-11).

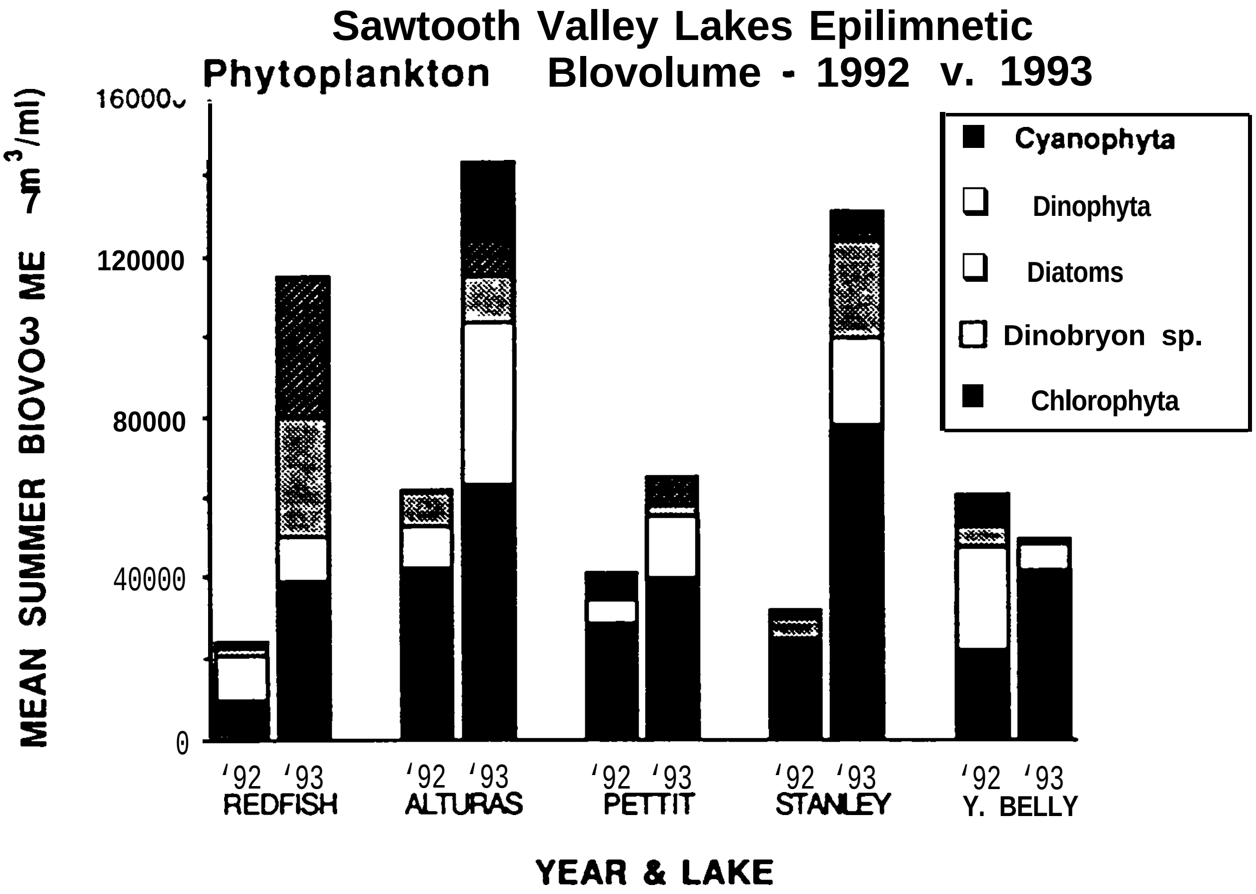


Figure 12. **Mean** summer **epilimnetic biovolume** of different phytoplankton **taxa** for the Sawtooth Valley Lakes in 1992 and 1993.

the diatoms did in the deeper waters. Diatoms were nearly absent from Yellow Belly Lake; however, *Dinobryon* were abundant in the deeper waters. In contrast to *Dinobyron*, *Peridinium*, when present, bloomed in the second half of the growing season (mid-July through October). Cyanophyta were not abundant in any of the lakes.

Rates of primary production in the Sawtooth Basin lakes were very low. Integrated water column productivity ranged from 6 to 22 mg C m^{-2} h^{-1} on the nine occasions it was measured:

Lake	*n*	Mean Carbon Fixation (mg C m^{-3} h^{-1})	Range	Mean % in Epilimnion
Redfish Lake	4	16	6-22	29%
Pettit Lake	3	13	9-16	35%
Alturas	1	27		30%
Stanley	1	13		62%

On most sampling dates, only a small proportion of the primary production occurred in the epilimnions of the lakes (Figs. 13-14, Appendix 2). In Redfish Lake primary production was usually maximal near the top of the thermocline at depths of 5-10 m (Fig. 13). In June the productivity curve was low and irregular, while in July there was a classical curve with lowered productivity at the surface (photoinhibition), a peak near 10 m, and then a gradual decline with depth. Productivity in June was measurable to 31 m, considerably below the 1% light-level depth of 22 m (Fig. 13). In August and September the productivity profiles indicated a secondary peak in productivity in the deep chlorophyll layer near the top of the hypolimnion (22 m). On these dates, photosynthesis was measurable to near 40 m. Secondary peaks in productivity were also found in Pettit Lake on the three dates when we made measurements (Fig. 14). In Alturas and Stanley lakes secondary production peaks were absent (Fig. 14), but photosynthesis in the

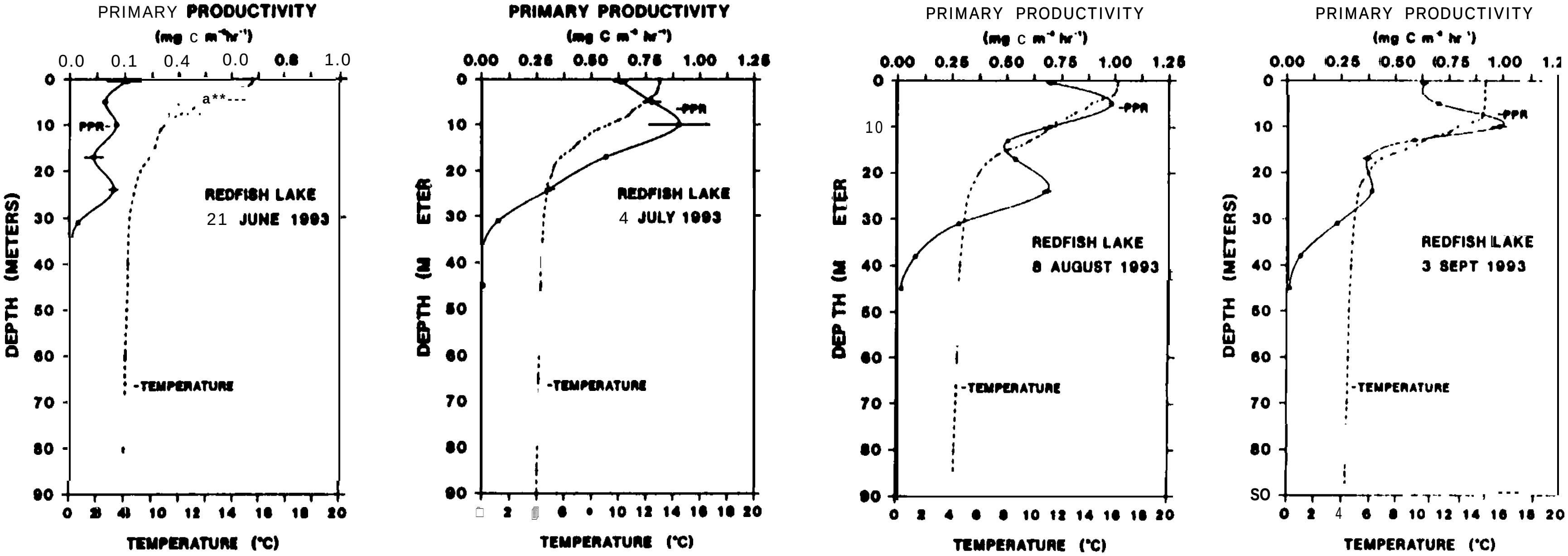


Figure 13. Depth profiles of primary production (PPR) and temperature in Redfish Lake on four dates in 1993. Mean and ranges of duplicate measurements of primary production are shown. The curves for primary production were fit with a spline function.

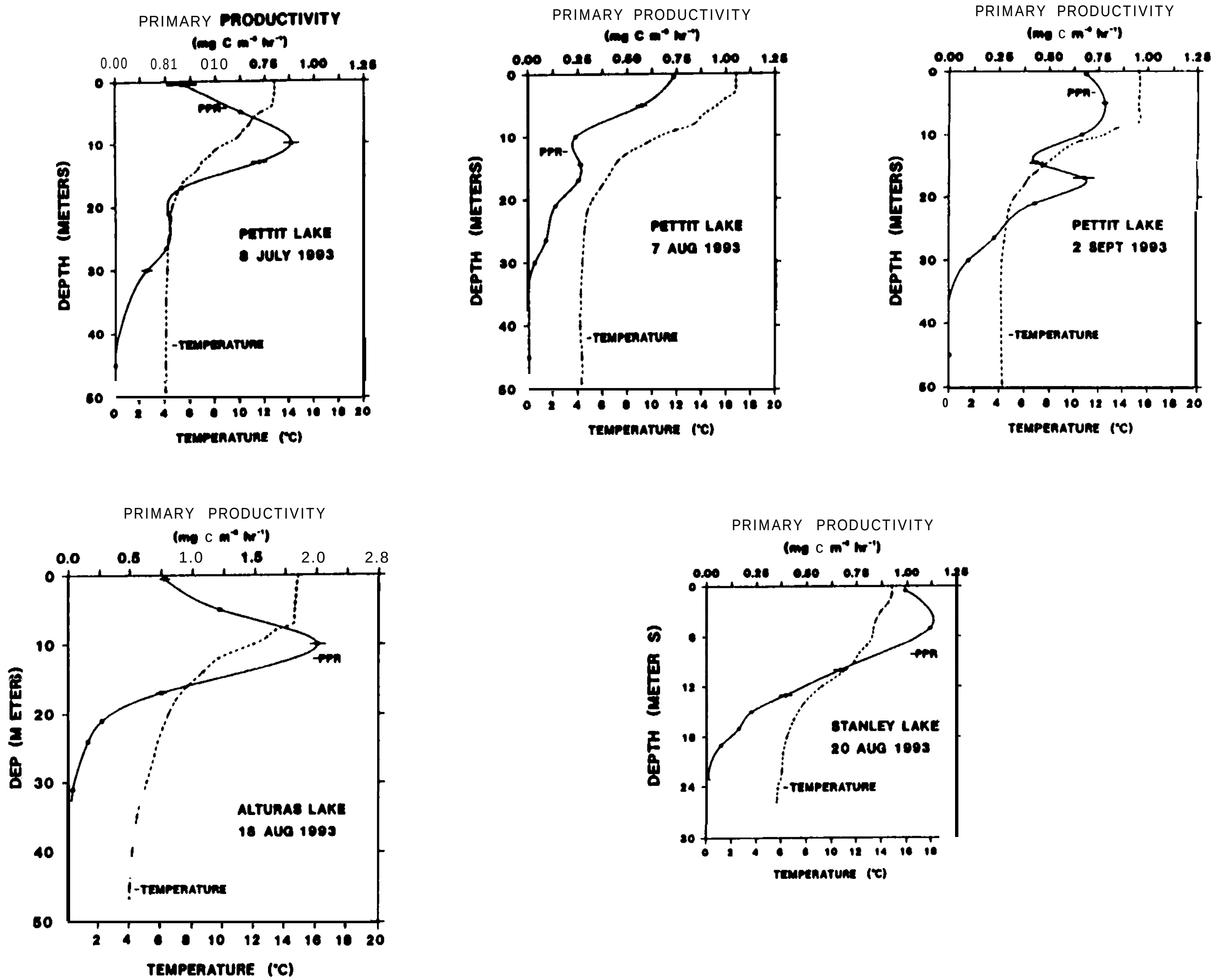


Figure 14. Depth profiles of primary production (PPR) and temperature in Pettit, Alturas, and Stanley lakes during 1993. Moan and ranges of duplicate measurements of primary produation are shown.

metalimnia and hypolimnia still contributed substantially to overall water column productivity.

Zooplankton

Although biomass, densities, and seasonal successional patterns varied considerably among lakes, the same species were found in all five lakes. The biomasses of zooplankton taxa in the epilimnia (0-10 m) of each lake during 1992 and 1993 are shown in Figure 15a-e. Both Redfish and Alturas Lakes had very low zooplankton biomasses dominated by Bosmina and *Holopedium*, with only a small contribution of Daphnia. In 1992 zooplankton biomass in Redfish Lake increased rapidly in May and reached peak levels by July. In 1993 our under-ice sample from a single station contained a large biomass of *Bosmina,* but by ice-out these had disappeared. Zooplankton biomass in Redfish Lake increased slower in 1993 than in 1992, reaching peak densities in July. In 1993 crustacean biomass in Alturas Lake was exceptionally low, remaining less than 1 µg/L until late August at which time a small peak in the carnivorous zooplankter *Polyphemus* occurred. Net efficiencies of zooplankton samples were similar among lakes in 1993, averaging 68%. The highest net effeciency was measured in Redfish Lake (mean of seasonal values 74.6%, SE 5.7) and the lowest in Pettit (mean 60.7%, SE 2.6). Alturas, Stanley and Yellow Belly Lakes were intermediate at 69.7% (SE 4.21, 69.7% (SE 5.4) and 68.7% (SE 4.1) respectively. The similarity of these values indicates that among lake differences in net efficiencies were not responsible for the variation in zooplankton observed.

Zooplankton biomass in Pettit and Yellow Belly epilimnion tows (0-10 m) reached maximum values of 50-100 µg/L in 1993, slightly less than highs (>120 µg/L) in 1992 (Fig. 15 c,d). Both lakes were dominated by Daphnia followed by *Holopedium* and *Bosmina* with sparse representations of both cyclopoids and *Polyphemus.* The later spring peak of all zooplankton species observed in Alturas Lake was also apparent in Pettit and Yellow Belly Lakes with no zooplankton biomass increase until early June in 1993.

Stanley Lake showed a similar pattern to the other four lakes with a late zooplankton peak in mid-June and overall less biomass in 1993 than in 1992 (Fig. 15 e). Stanley was dominated by *Daphnia* and Epischura in both years, but the third most prominent species shifted from *Holopedium* in 1992 to *Bosmina* in 1993. Stanley, Pettit, and Yellow Belly Lakes exhibited the highest and most consistent biomass of large cladoceran zooplankton likely preferred by zooplanktivorous fish.

Crustacean zooplankton biomass differed by 10 to 50-fold among the different Sawtooth lakes (Fig. 16a). In both years the highest total crustacean biomasses *were* observed in Yellow Belly and Pettit Lakes, followed by Stanley Lake. The lowest total zooplankton biomass was observed in Redfish and Alturas lakes in both years with Alturas Lake exhibiting almost no crustacean biomass in 1993. The most pronounced inter-annual difference between 1992 and 1993 was a pattern of generally less total zooplankton biomass in 1993 and a later spring bloom for most species in Pettit, Yellow Belly, and Stanley Lakes in 1993.

In contrast to total biomass, total zooplankton density (#/L) revealed a pattern of higher densities in 1993 than in 1992 especially in Pettit Lake (Fig. 16b). Alturas showed similar reductions in total densities throughout the summer in 1993 as was observed in biomass estimates. Both Redfish and Stanley Lakes demonstrated a spring peak of high densities in late March and early April, but this peak in density was not reflected in the biomass estimates in Stanley suggesting high numbers of very small zooplankton.

Daytime depth distributions of crustacean zooplankton varied among the lakes (Fig. 17). In Redfish, Alturas and Yellow Belly Lakes zooplankton biomass was greatest in the 0-10 m depth strata. In these lakes the seasonal biomass patterns presented for epilimnetic samples provides a good representation of overall zooplankton abundance and inter-annual variability (*Fig.* 16). In Pettit and Stanley Lakes, however, much of the zooplankton biomass was present during the day at depths below the thermocline. In

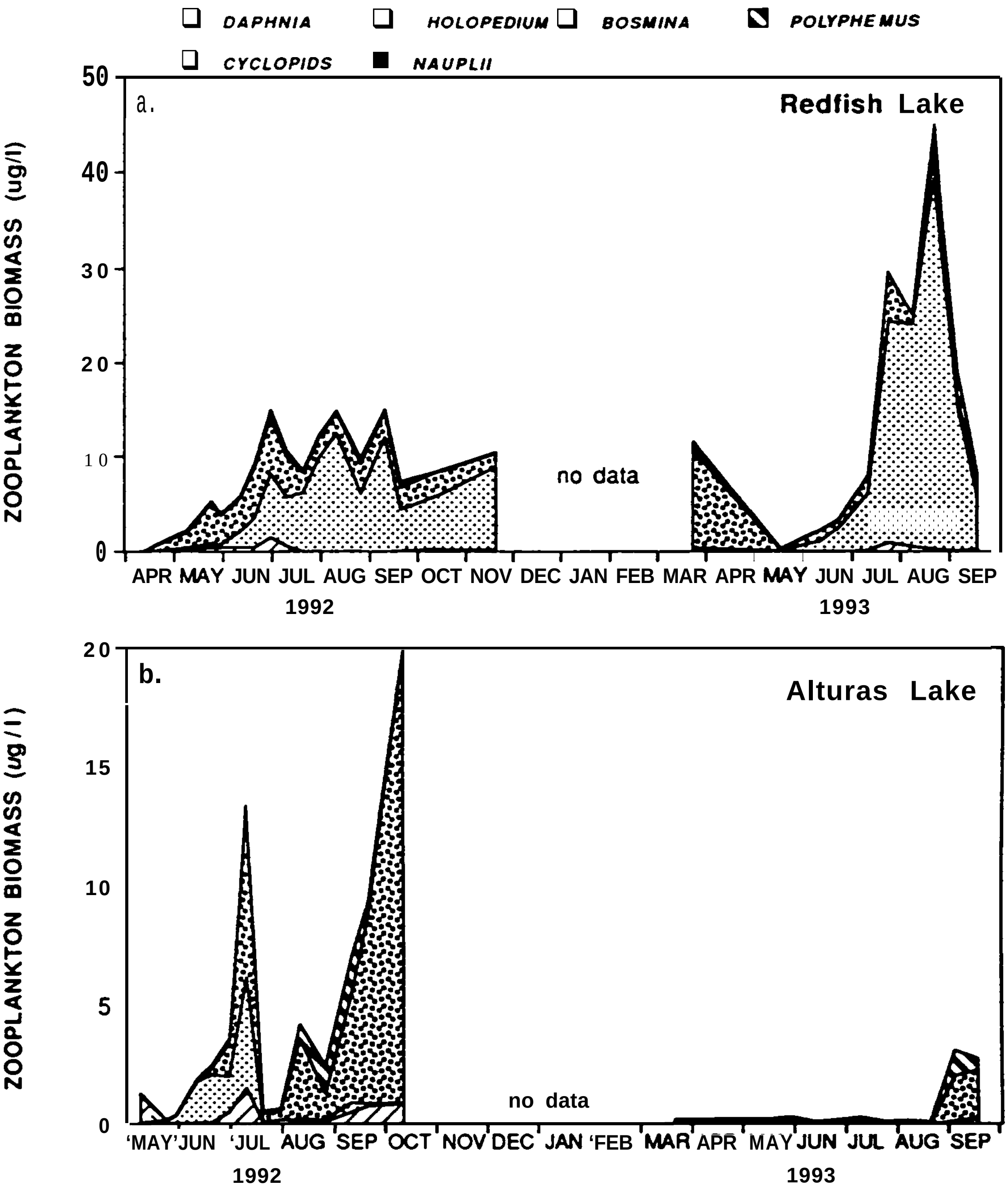


Figure 15. Seasonal changes in the biomass of zooplankton **taxa** in epilimnetic tows (10-0 **m)** for the Sawtooth Valley Lakes in 1992 and 1993. **a) Redfish, b)** Alturas, **c)** Pettit, d? **Stanley,** and e) Yellow Belly.

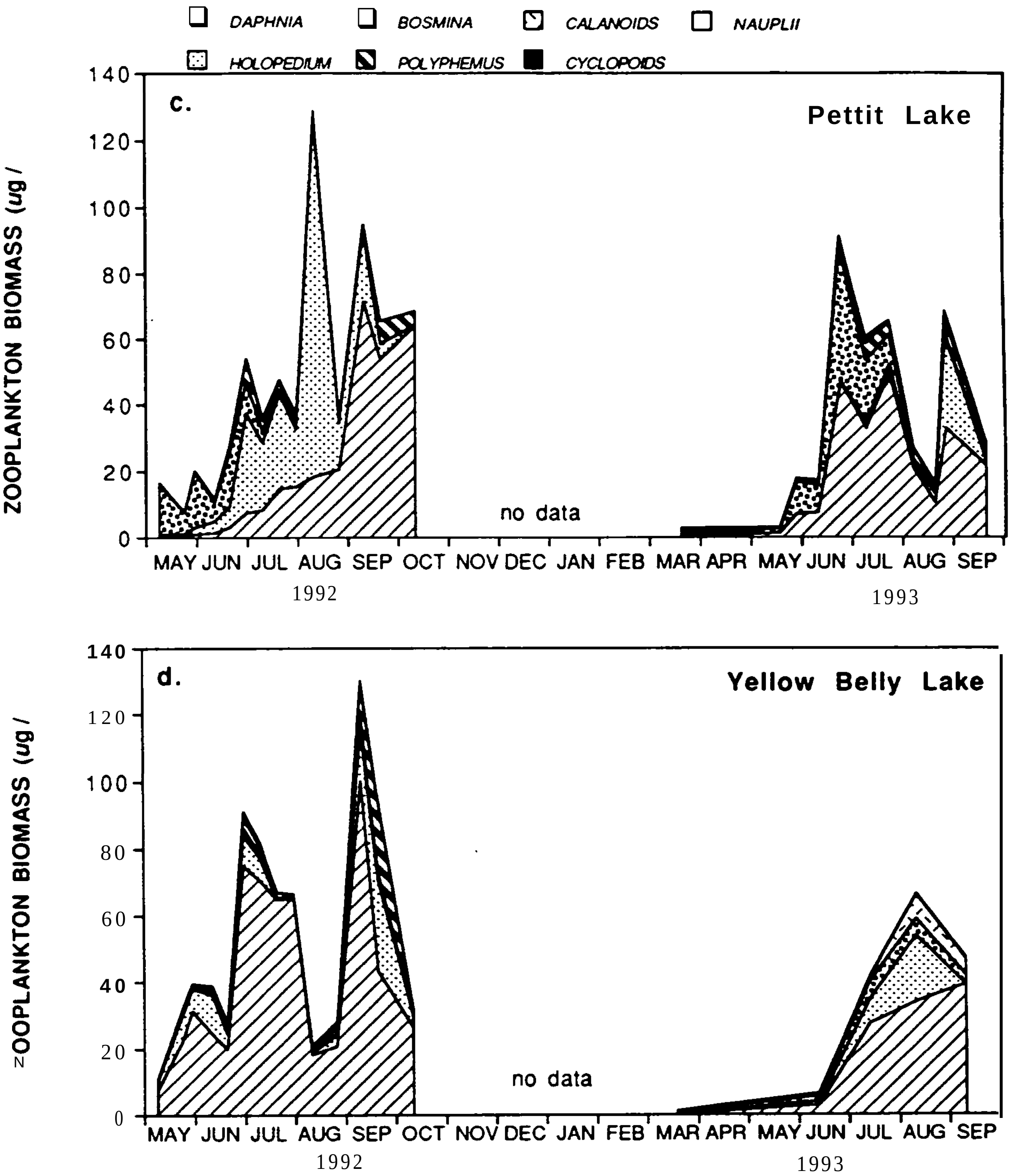
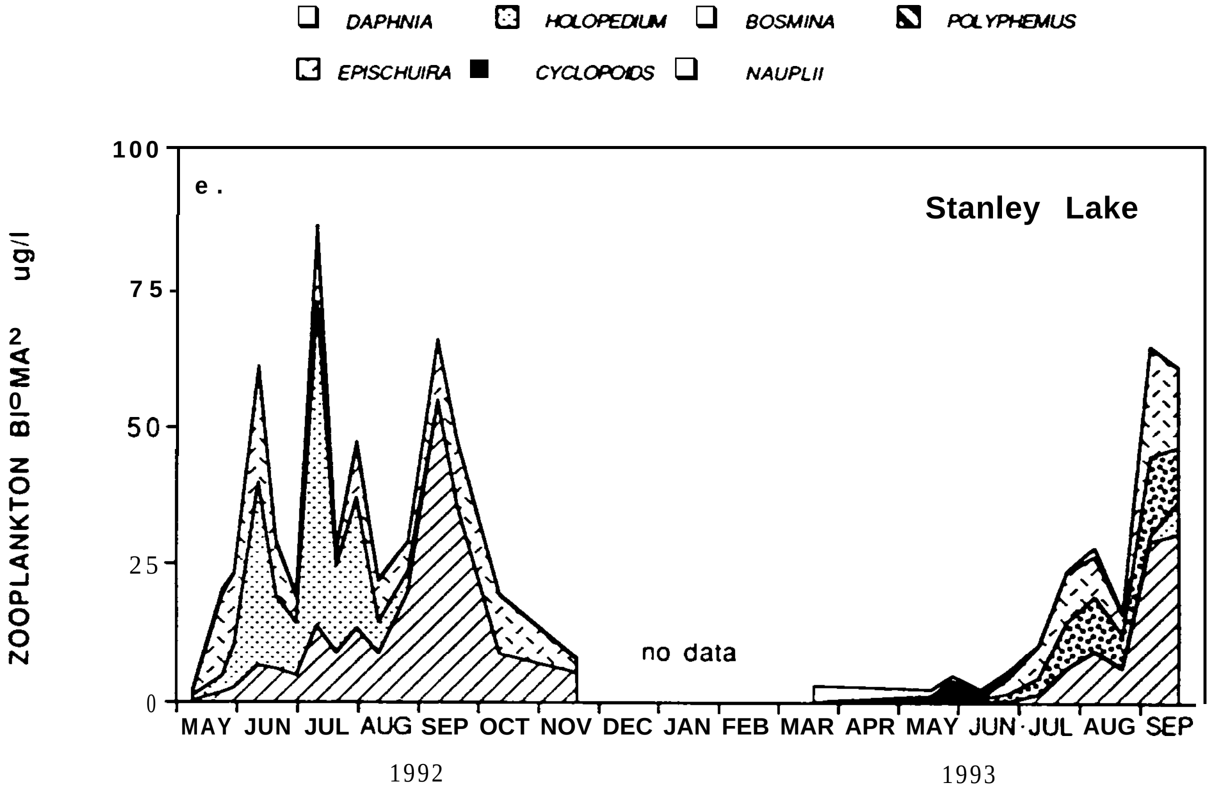


Figure 15. Seasonal changes in the biomass of zooplankton **taxa** in epilinmetic tows (10-0 **m) for** the Sawtooth Valley Lakes in 1992 and 1993. a) **Redfish, b)** Alturaa, **c)** Pettit, **d)** Stanley, and **e)** Yellow Belly.



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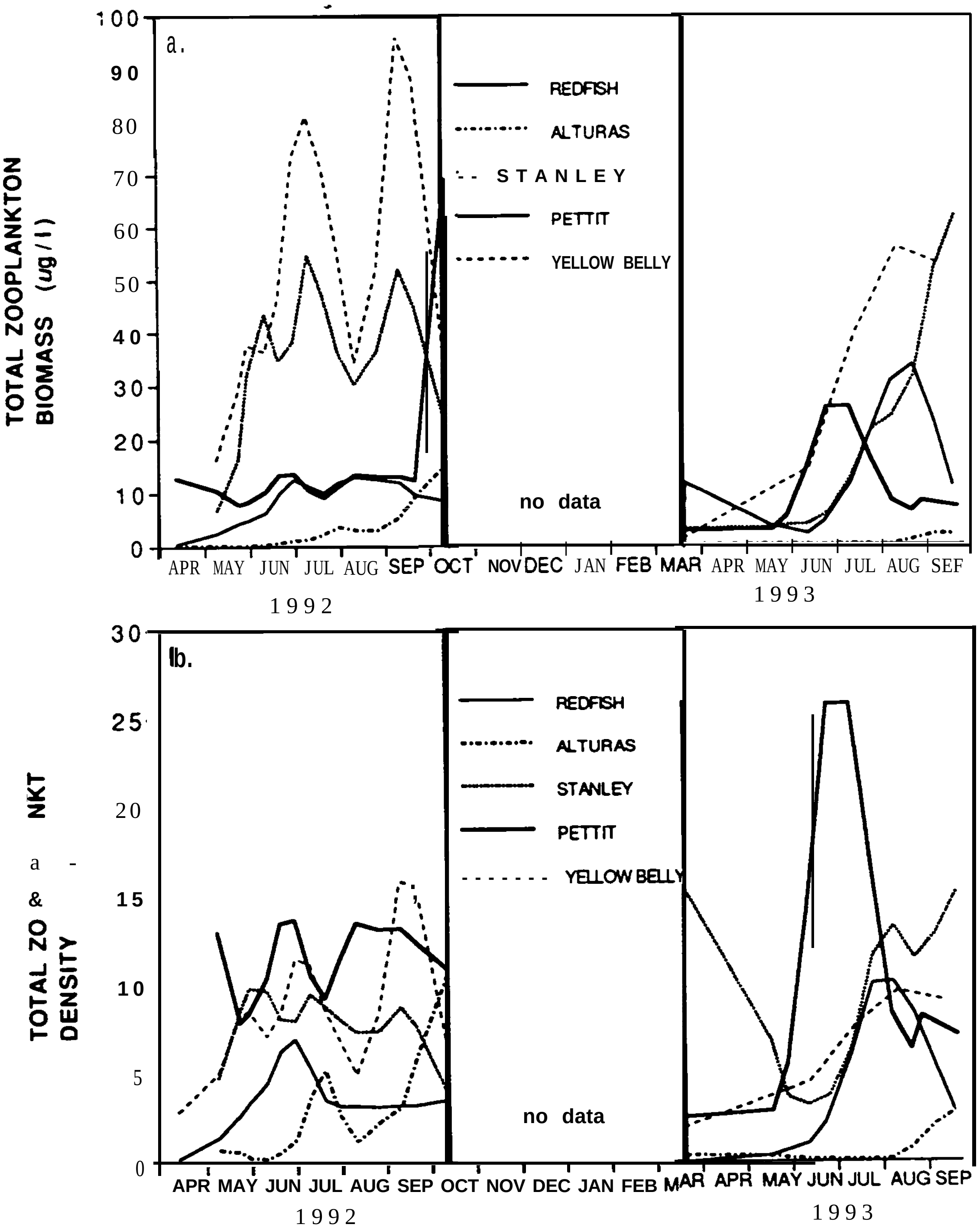


Figure 16. Seasonal distribution of total **crustacean** zooplankton from epilinmetic tows (10-0 m) for the Sawtooth Valley Lakes in 1992 and 1993. Three-point running means are graphed to smooth the **curves**. **a)** Biomass **b)** Density.

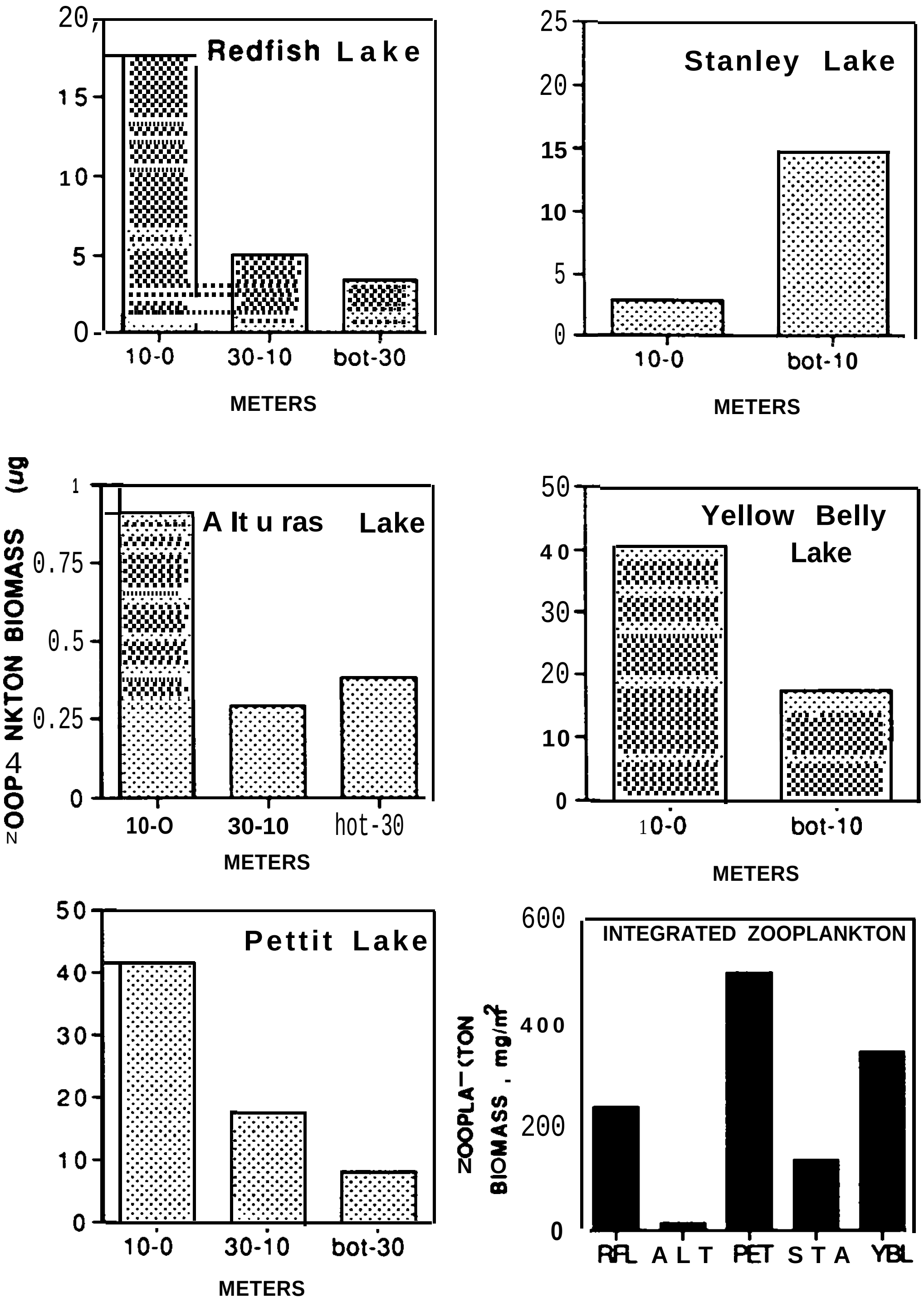


Figure 17. Depth distribution of crustacean zooplankton in Sawtooth Valley Lakes. Depth strata represent metered zooplankton tows from 10 m to the surface (10-0), 30 m to 10 m (30-10), and approximately 1 m off the bottom to 30 m (bot-30). Integrated zooplankton biomass from the bottom to the surface in each lake expresses as micrograms per square meter of lake surface area.

these lakes crustacean biomass should be integrated through the water column to provide an estimate of zooplankton food resources available for planktivorous fish (Fig. 17).

Daphnia exhibited a pattern of diel vertical migration in Stanley Lake (Fig. 18a). Daphnia were present in high numbers in the top 5 m during the night (0000 h) and just off the bottom during the day (1200 h). Daphnia appeared to be distributed somewhat evenly throughout the water collum at both 1600 h and 0400 h, but were observed in highest numbers in the middle of the water collum at 2000 h and 0800 h.

Epischura were present at high densities near the surface for most periods (Fig. 18a). *Bosmina* and *Holopedium* showed little vertical movement and were observed in relatively similar densities in different depth strata throughout the day and night.

Examination of the depth strata containing the highest concentrations of each species at each sampling period indicated that *Daphnia* migrated extensively from near 20 m deep at noon to approximately 5 m at midnight. The pattern of movement for the other species was less apparent *(Bosmina)* or non-existent *(Holopedium* and ***Epischuira***) (Fig. 19a).

Crustacean zooplankton showed little diel vertical migration in Redfish over the 24 h period sampled (Fig. 18b). *Holopediutn* were located in highest concentrations in the top 20 m of the water column, showing a slight upward migration at night. *Bosmina* were distributed almost evenly from the surface down to 45-50 m and did not migrate. The few *Daphnia* present were observed primarily in the top 15 m. Similarly, the few cyclopoids present were found just off the bottom from 1200 h to 1600 h. The modes of zooplankton density in different depth strata indicate that *Holopedium* moved from the metalimnion during the day to the epilimnion at night (Fig. 19b).

Hydroacoustic estimates **of *Fish*** Abundance

Density estimates of fish varied considerable among the lakes during the September 1993 sampling period. When expressed as

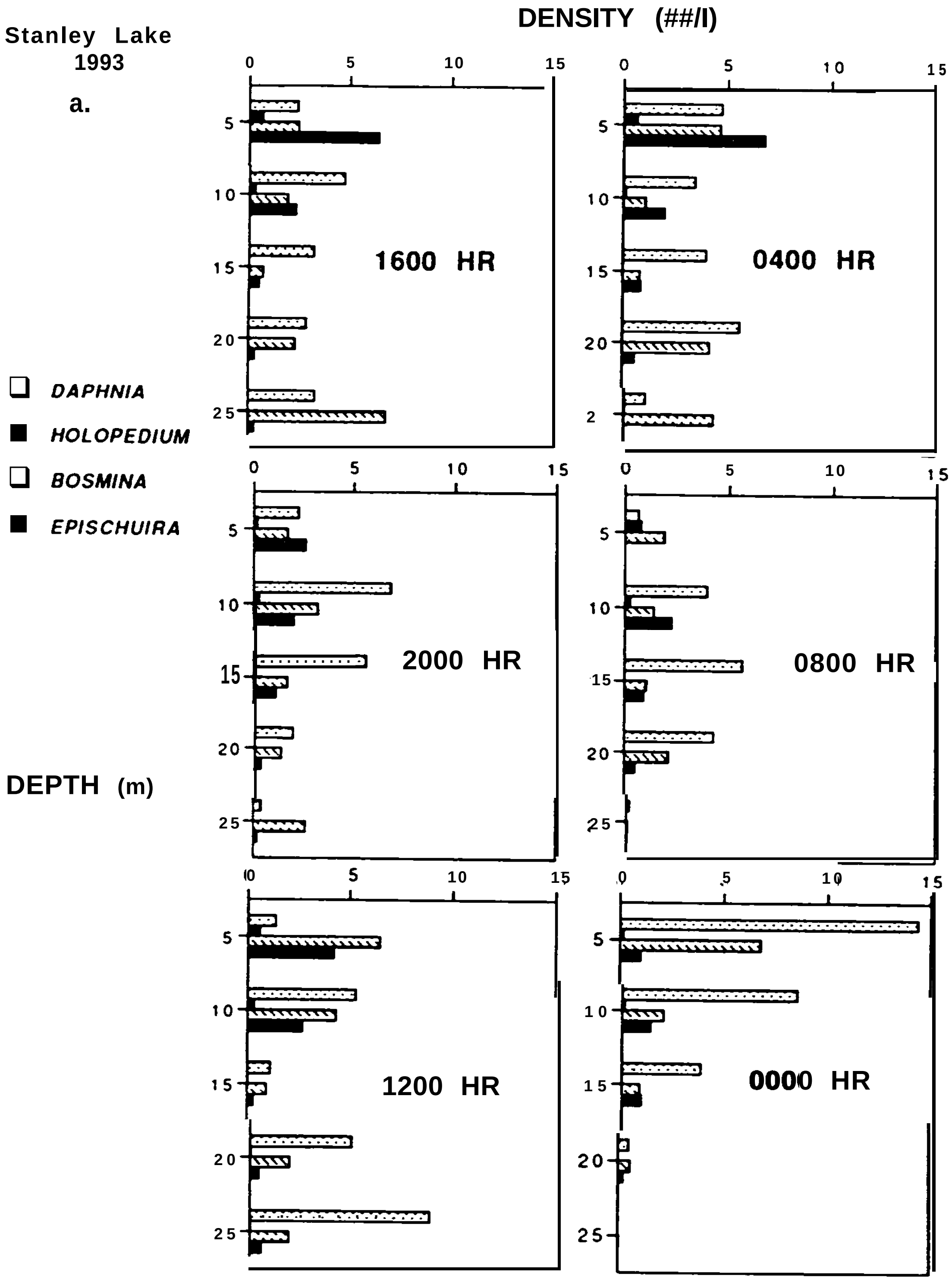


Figure 18. **Diel** vertical distribution *of* zooplankton densities from September, 1993. **a)** Stanley **b)Redfish**

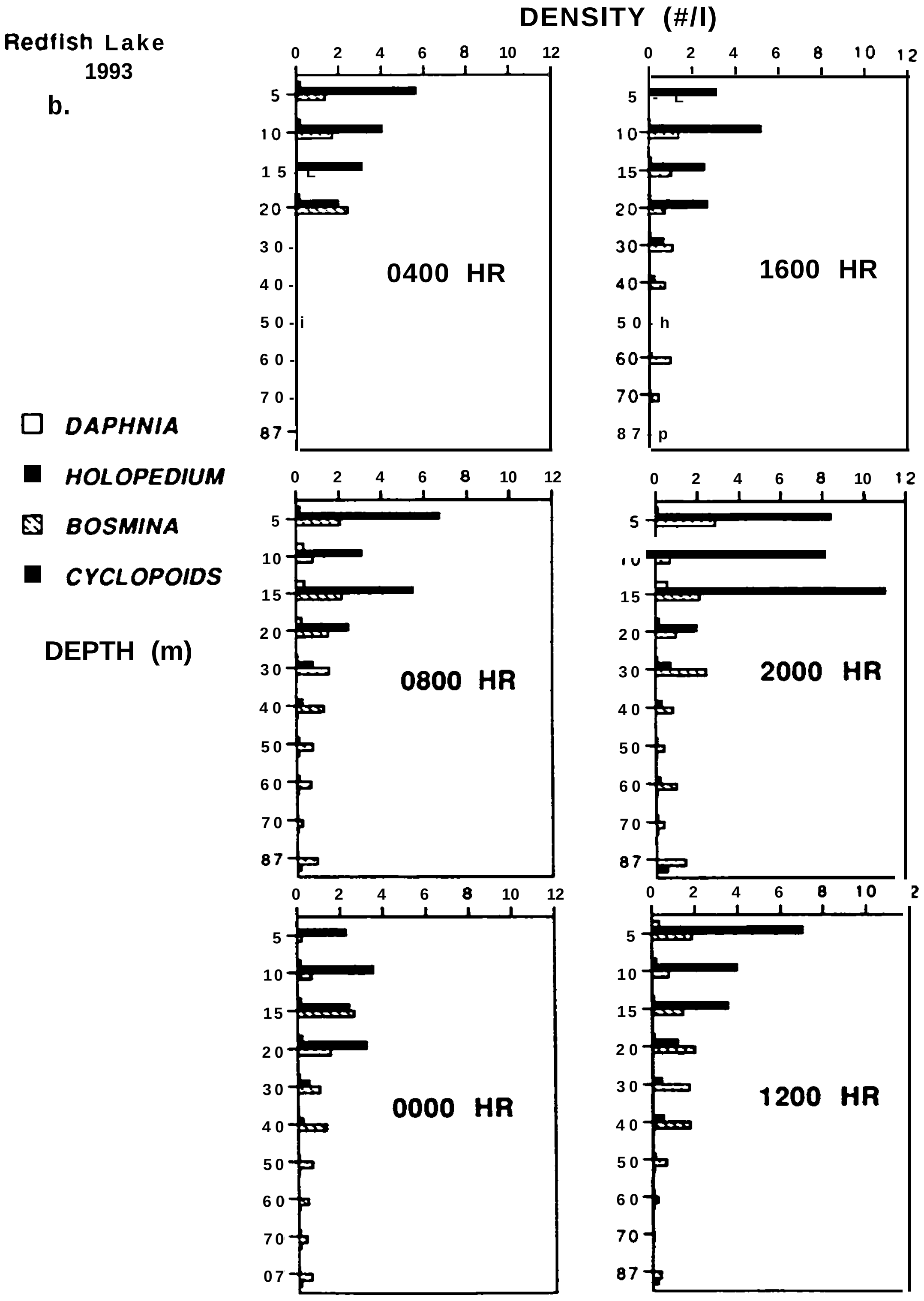


Figure 18. **Diel** vertical distribution of **zooplankton** densities from September, **1993.** a) Stanley **b)Redfish**

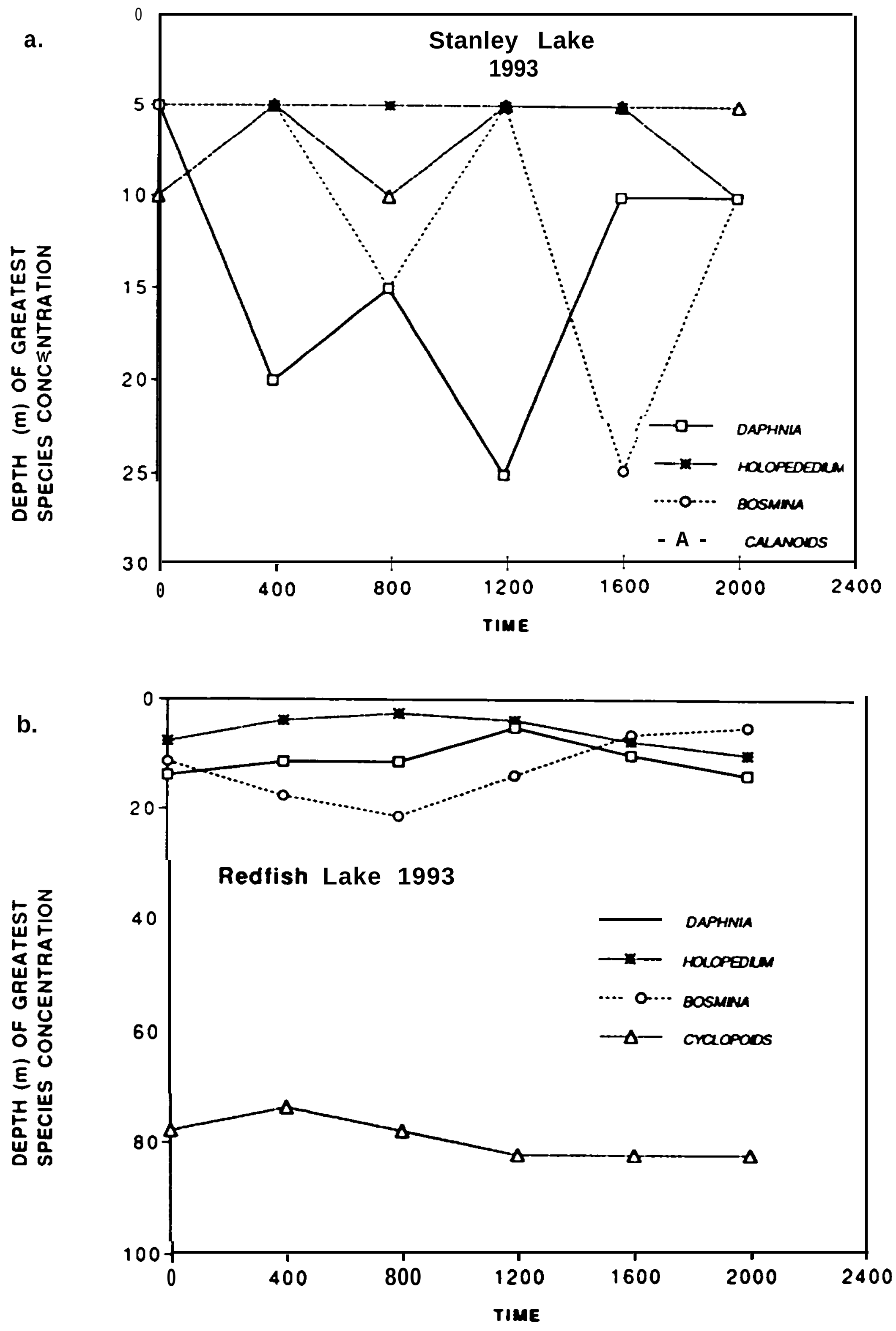


Figure 19. Depth strata containing the highest density of each crustacean zooplankton species over the 24 hour sampling period in September 1993. **a)** Stanley **b) Redfish.**

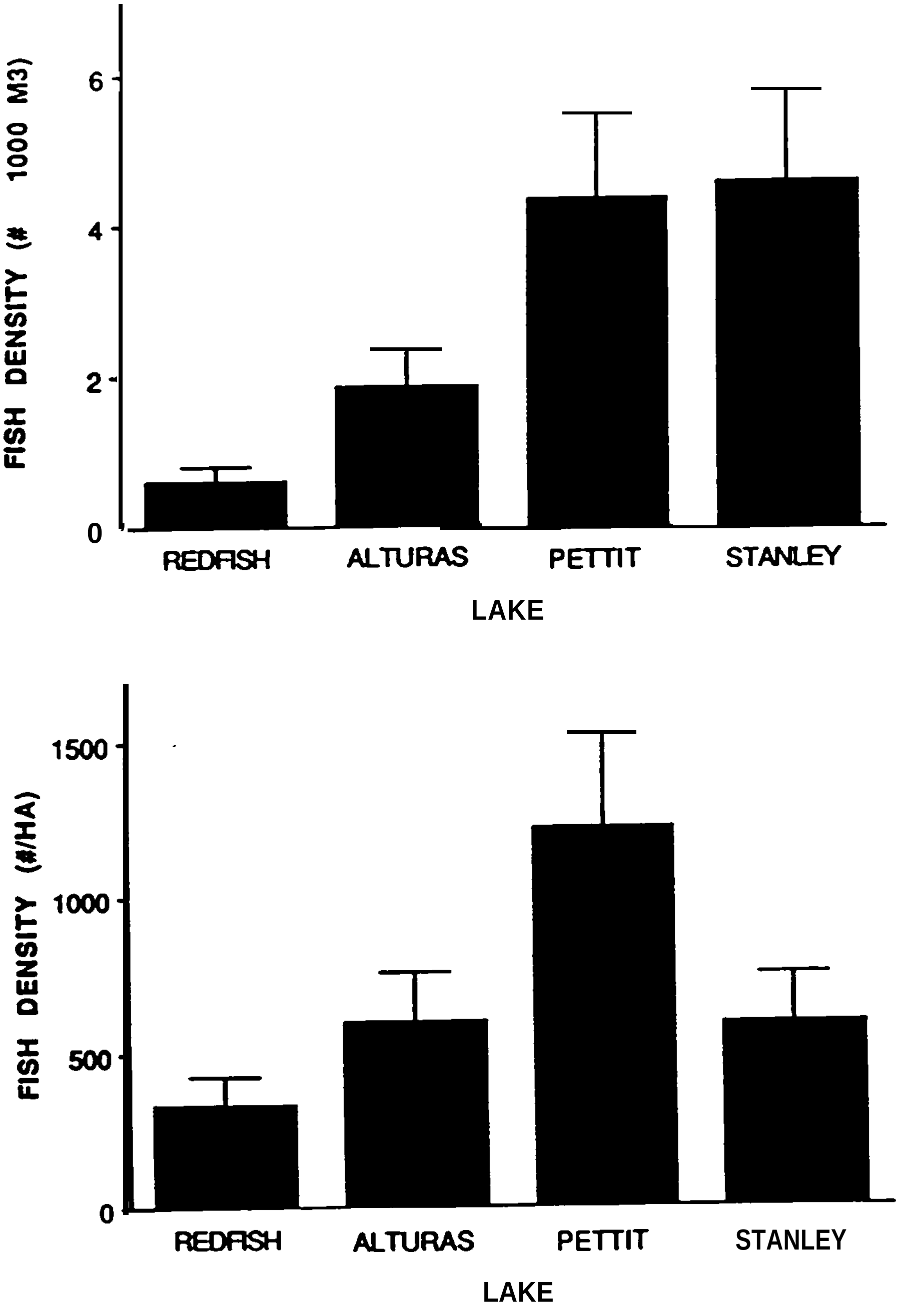


Figure 20. lean density of **fish expressed** in **volumetric** (top) and **areal (bottom)** terms as measured with hydroacoustic surveys conducted on each of four Sawtooth Valley Lakes. Error bars represent one standard error of 7-11 transects surveyed on each lake.

number of fish per water volume, Stanley and Pettit lakes contained the highest density (>4 fish 1000 m^{-3}), followed by Alturas (1.9 fish 1000 m^{-3}) and Redfish (0.4 fish 1000 m^{-3}) (Fig. 20, Appendix 3). When expressed as number of fish per hectare, Pettit had the highest densities, Redfish had lowest densities, and Stanley and Alturas were intermediate (Fig 20). These density estimates were similar to those of the 1992 surveys for Redfish, Pettit and Stanley lakes. The 1993 Alturas densities were approximately 60% greater than values reported for 1992 (Beauchamp et al. 1993).

Depth-stratified estimates of fish abundance also varied considerably among the lakes (Fig. 21). In Redfish and Alturas Lakes, highest fish densities were present in metalimnetic depth strata for all three size classes of fish. Fish densities were high between 10-20 m in Redfish and between 15-40 m in Alturas Lake. In Stanley and Pettit Lakes small and medium sized fish were most abundant in the 0-5 m strata, whereas larger fish were most abundant in deeper water.

Density of small fish in epilimnetic waters was particularly variable among the lakes (Fig. 21). In Pettit and Stanley lakes the abundance and spatial variation of small epilimnetic targets was great. Mean values of greater than 10 fish per 1000 m^3 were present in the 0-5 m depth strata of both lakes. The coefficient of variation for these samples was also high, exceeding 70% for both lakes. In Redfish, and particularly Alturas, the density of small epilimnetic targets was reduced. The coefficient of variation remained high in Redfish, but fell to less than 40% in Alturas.

The high variation associated with these small, epilimnetic targets resulted from two sources. First, the volume of water sampled in the 0-5 m depth strata was small due to the conical shape of the transmitted beam of sound. The inclusion of a few targets in this depth strata results in high density estimates in some transects and values of zero in other transects. This is particularly true for the Redfish survey where five of eleven transects indicated no small targets in the 0-5 m strata. In

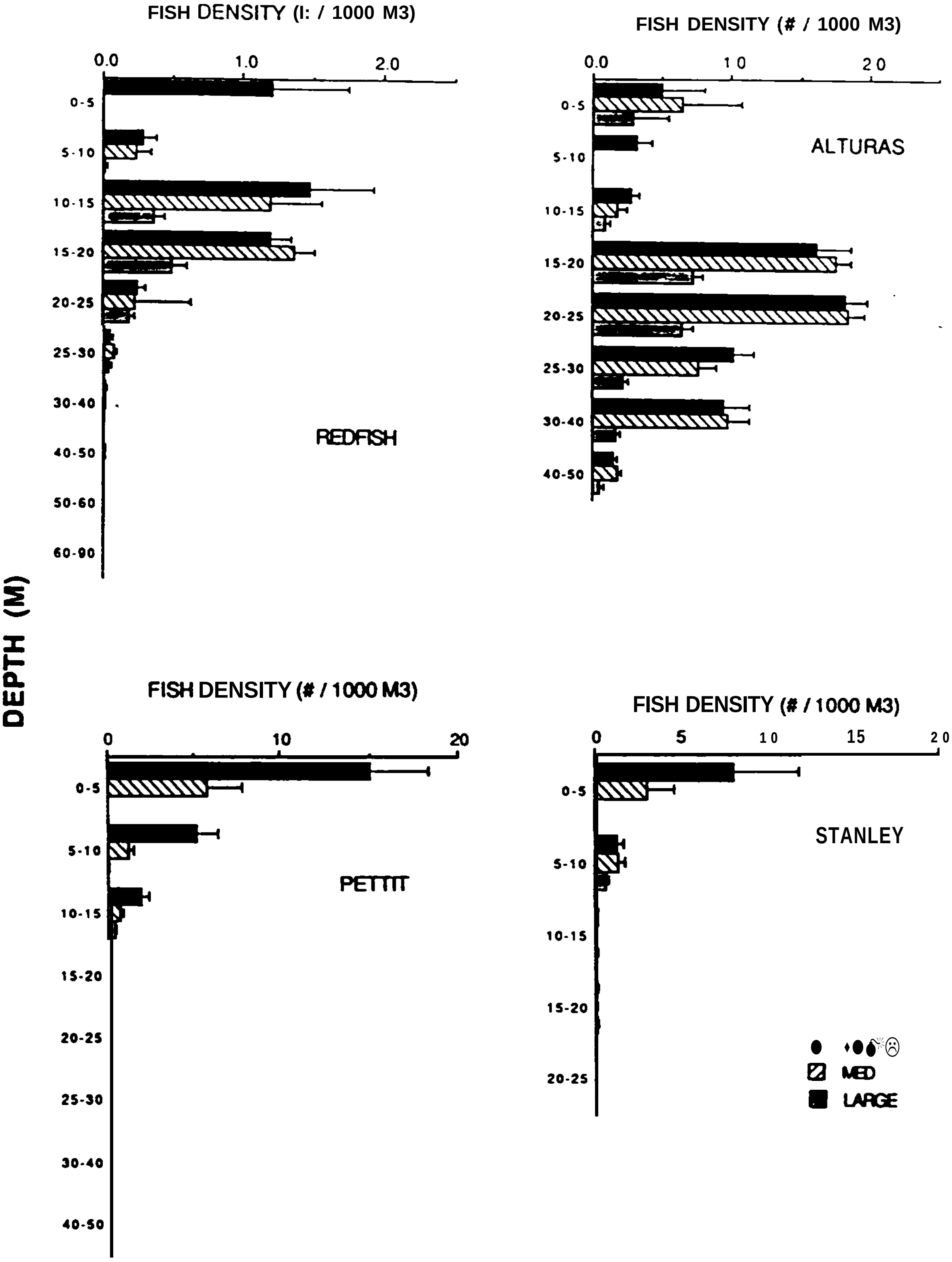


Figure 21. Mean density of fish targets in different depth strata of a) Redfish, b) Alturas, c) Pettit and d) Stanley Lakes. Beginning and end densities of the 205 m depth strata are not included. Small (-59 to -51 db), medium (-51 to -43 db) and large (-43 to -29 db) fish targets are shown. Error bars represent 1 S.E. of the mean.

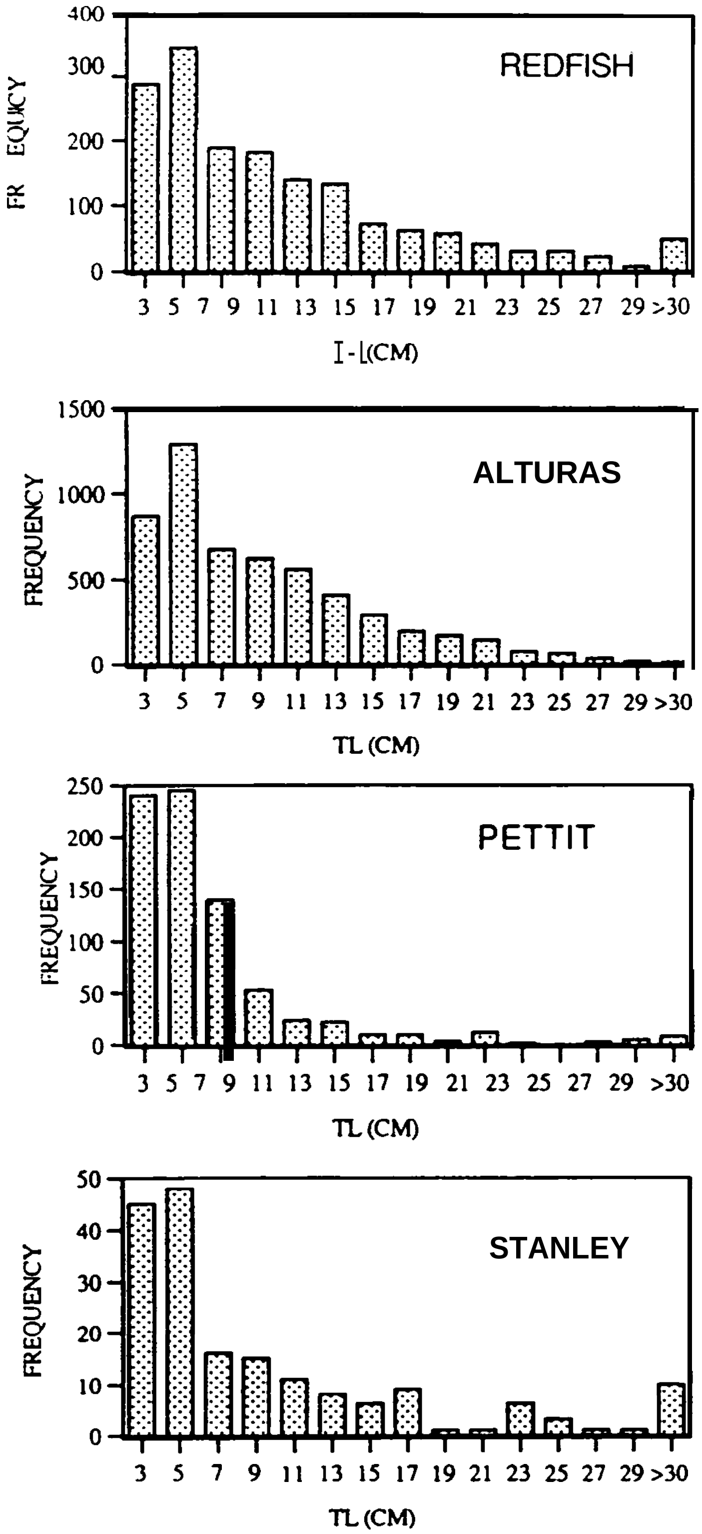


Figure 21b. Size distributions of fish targets from the hydroacoustic **survey** conducted September 1993. Targets strengths were converted to fish total lengths using equations in Love (1971). Note the differences in scale on the y-axis.

Pettit and Stanley lakes, however, all transects included some targets.

A spatial pattern in densities of small targets in the O-5 m strata was evident in Pettit, Stanley and to some extent Redfish surveys (Fig. 22). Densities of these small targets was generally greatest in transects at the beginning and end of the surveys. In Pettit Lake very high densities of small fish in the O-5 m strata were recorded on transects 1, 7 and 10. These same transects contained high epilimnetic fish densities during the 1992 survey (Beauchamp et al. 1993).

Results from trawl surveys conducted in 1992 indicated that most of these small epilimnetic targets were redside shiners (Beauchamp et al. 1993). During field sampling in 1993 many of these fish were observed in Stanley and Pettit Lakes, and were captured and released from seine hauls in June. In order to approximate abundances of kokanee in the 1993 survey and to compare the 1993 results with surveys conducted in 1991 and 1992, we partitioned acoustic targets from the 1993 survey into species based on the depth-stratified catch in mid-water trawls in 1992 (Beauchamp et al. 1993).

Size-frequency distributions of fish targets from the September 1993 acoustic survey varied among lakes. Although a dominant mode ocurred in all lakes in the 4-6 cm length class, the frequency of smaller and larger targets varied considerably (Fig. 21b). In Pettit and Stanley Lakes, a large proportion of the acoustic targets were present in the 2-4 cm length class. These targets were likely the small cyprinids present in 1992 trawl catches and 1993 snorkle observations (D. Teuscher, personal observations). In Redfish and Stanley Lakes, a relatively high frequency of targets was present in the largest size class (>30cm). These targets were likely piscivorous-sized salmonids or cyprinids.

Partitioning the acoustic targets into species produces a pattern of kokanee abundance that differs from measurements of total fish abundance (Fig. 23). Kokanee density was greatest in Stanley and Alturas lakes when density was expressed either in

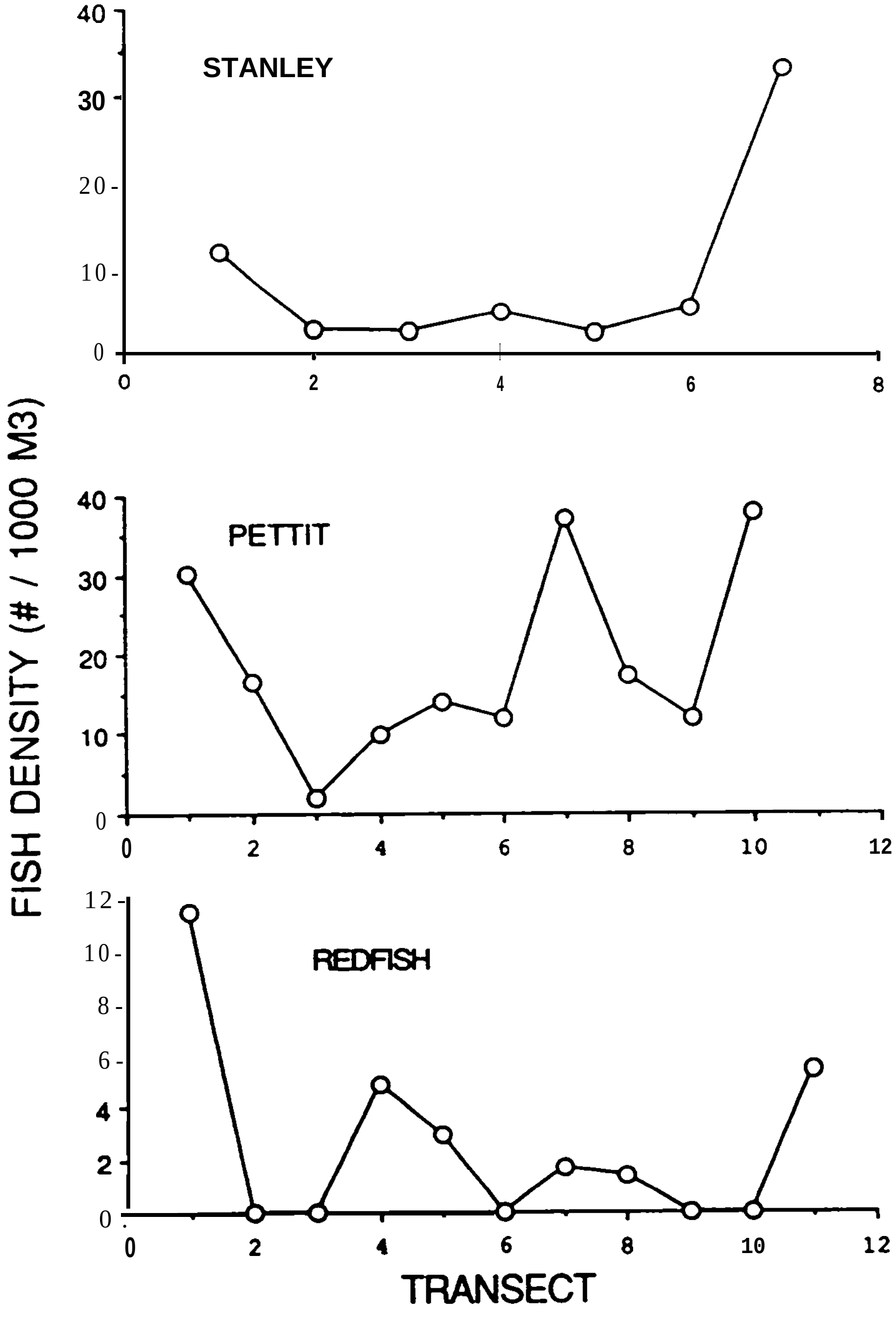


Figure 22. Fish density present in the 2-5 m depth strata along each transect of Stanley, Pettit and Redfish Lakes. Note peaks in fish density on the beginning and end transects for each lake.

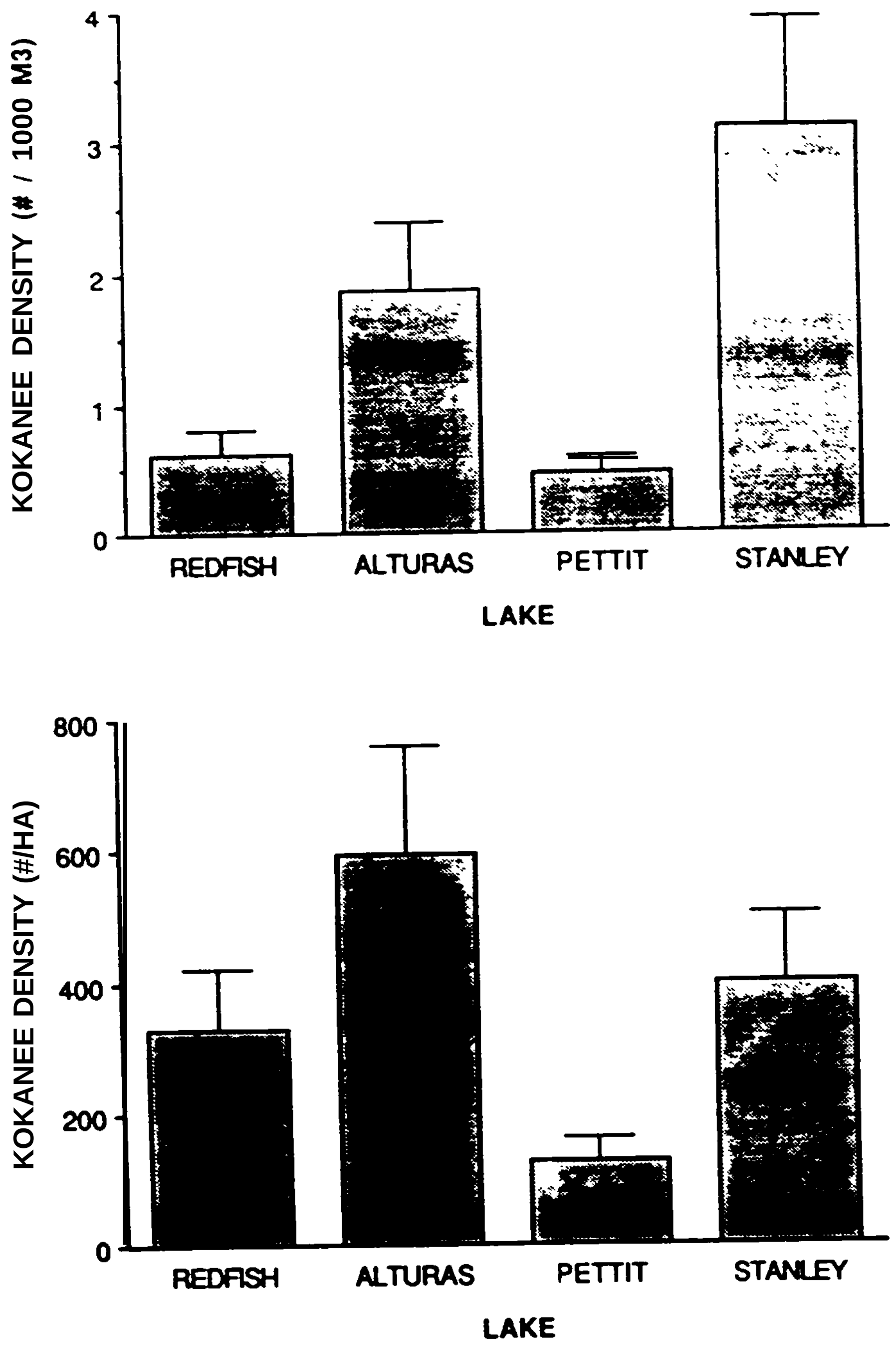


Figure 23. Volumetric (top) and areal (bottom) estimates of kokanee density from the hydroacoustic surveys. The proportion of acoustic targets within each depth strata that were kokanee were estimated based on relative catch of kokanee in mid-water trawls in 1992. Error bars represent ±1 S.E.

volumetric or areal terms. Kokanee density was lowest in Pettit Lake where only 0.5 fish/1000 m^3 were present.

Comparison of results from acoustic surveys conducted in the lakes from 1991-1993 indicated that kokanee abundance has been relatively constant in Pettit and Stanley lakes, but much more variable in Redfish and Alturas lakes (Fig. 24). Kokanee abundance in Redfish lake increased from 90,000 fish in 1991 to almost 200,000 fish in 1992 and 1993. Kokanee abundance in Alturas Lake declined approximately 60% from 1991 to 1992 and then rebounded to almost 1991 levels in 1993. Changes in the density of small fish targets were responsible for the increased abundance of kokanee in Redfish Lake in 1992. Increases in the abundance of small and medium targets were responsible for the larger estimate of kokanee in Alturas Lake in 1993.

The daytime survey of Redfish Lake indicated that most targets were present in the 0-10 m depth strata (Fig. 25). Relative abundance of targets declined with depth, such that very few targets were present below 30 m. Some of the targets in the 0-10 m strata could have been rainbow trout stocked in 1992. If so, these targets would not have been present during the nighttime survey in that rainbow trout usually remain on the bottom at night where they are not sampled with acoustic methods (Luecke et al., submitted; R. Tabor and W. Wurtsbaugh, unpublished data). The moderate abundance of targets observed in the 10-20 m strata, and the lack of targets present below 30 m indicates that kokanee were likely remaining in the metalimnion during daytime hours. This strata was where highest fish densities were present at night. Results from our daytime survey suggest that kokanee were not making extensive diel vertical migrations in the Redfish Lake during the late-summer of 1993. These conclusions are similar to those drawn from the 1992 survey (Beauchamp et al. 1993).

DISCUSSION

Inter-year fluctuations climatic and limnological conditions appeared to have a large effect on the Sawtooth Valley Lakes. 1993

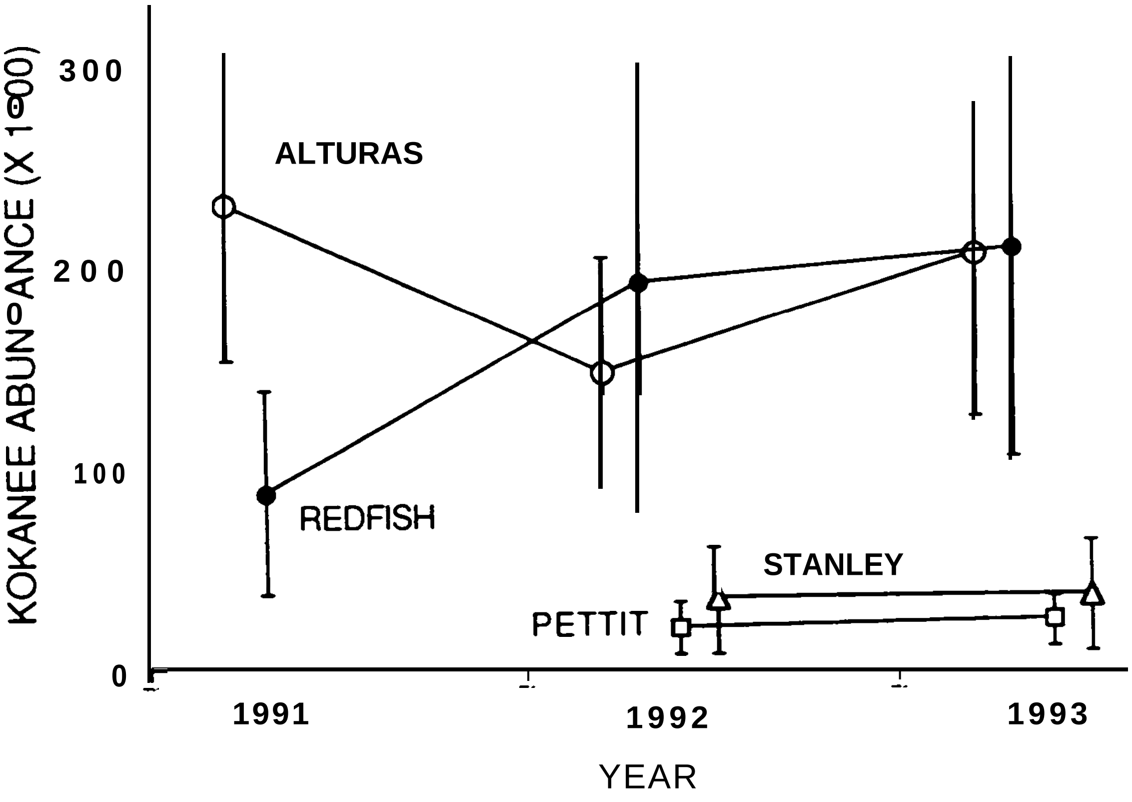


Figure 24. Total abundance of kokanee in **four** Sawtooth Valley Lakes estimated with acoustic **surveys** in 1991-1993. Abundance estimated from acoustic density, proportion of **kokanee** captured in depth-stratified mid-water trawls, and the volume of water present in each depth strata. Error bars represent 95% C.I. **of** the mean.

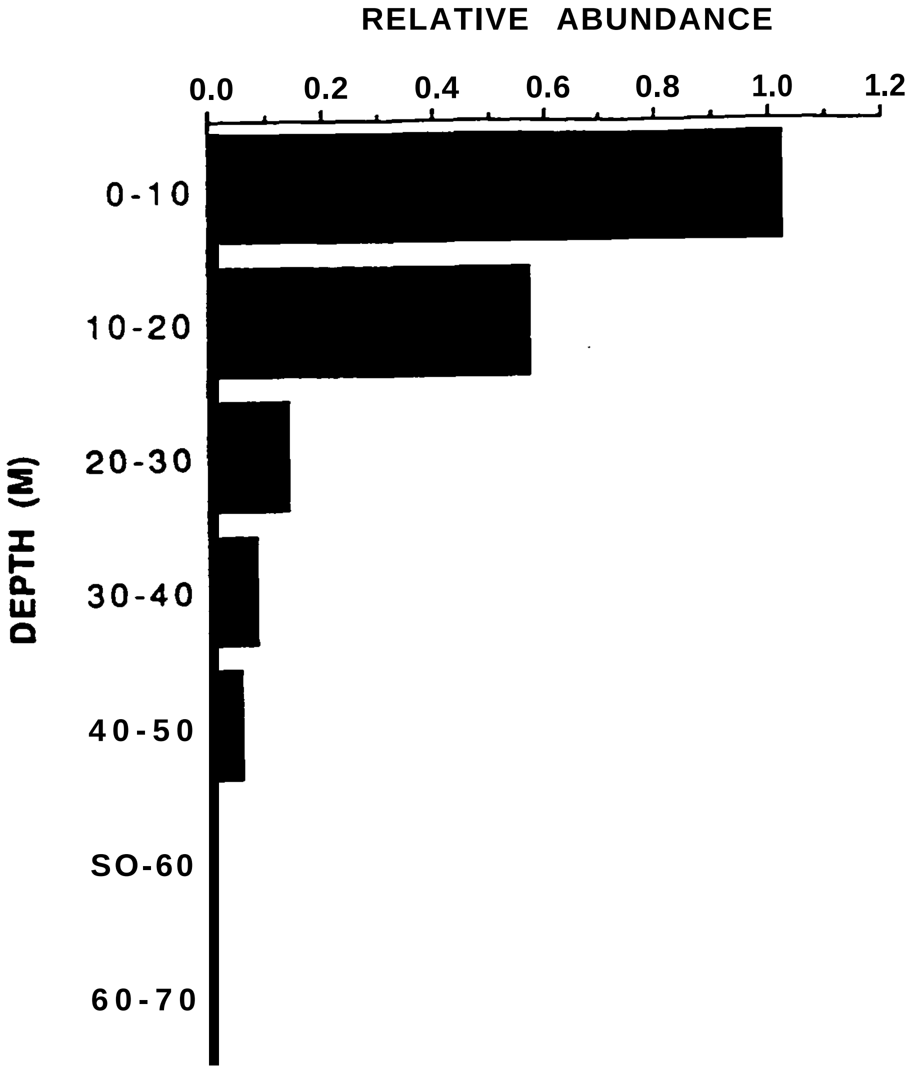


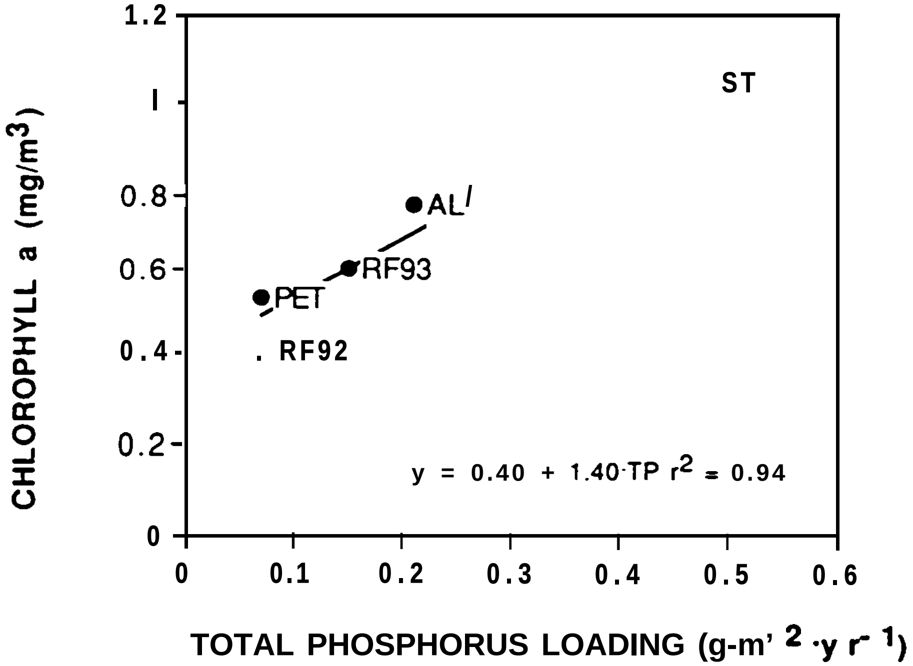
Figure **25.** Relative abundance of fish targets present in daytime acoustic survey of **Redfish Lake** collected 17-Sept-93. The number of targets sampled in each depth strata was corrected for sampling effort in each strata to calculate relative abundance.

was a much cooler and wetter year than 1992. While these differences did not effect the relative rankings of productivity between lakes, the increased run-off and/or cooler water temperatures led to a larger, but later spring algal bloom. Mean summer chlorophyll levels and/or algal biovolumes were higher in all lakes in 1993 than in 1992.

Chlorophyll concentrations in the lakes appear to be closely related to nutrient loading to each system (Fig. 26). Pettit and Redfish lakes had the lowest area1 loadings of nitrogen and phosphorus (Chapter 1), and they also had the lowest levels of chlorophyll, and the lowest levels on TN and TP (Chapter 2). Stanley Lake, with relatively high loading, had the highest levels of chlorophyll.

The effects of the weather and primary productivity dynamics on zooplankton are threefold. First, the cool temperatures and the later spring algal bloom in 1993 may be the cause for a zooplankton peak that was delayed compared to 1992. Second, the shortening of the zooplankton growing season may have resulted in the lower overall zooplankton biomass in the Sawtooth Valley Lakes. Finally, these dynamics may have effected species composition in the lakes. Higher densities of the small bodied *Bosmina* in 1993 may be due to their capability of quick reproduction during the truncated summer of 1993. Finally, in Stanley Lake, higher densities of *Bosmina* may have led to an increase in the population of the predacious calanoid *Epischura*.

The differing response of fish abundance in Redfish and Alturas Lakes between 1992 and 1993 acoustic surveys suggests that no general pattern of fish recruitment and volume of stream inflow water operates in these lakes. 1992 was the last year of a seven-year drought. A return to average snowfall in 1993 caused stream inflows to increase two to three-fold. In Alturas Lake abundance estimates of small fish increased by 50% whereas no such increase was observed in Redfish densities. Our uncertainty about the origin of small fish in Alturas may make this comparison moot. In Redfish almost all kokanee spawning occurs in Fishhook Creek



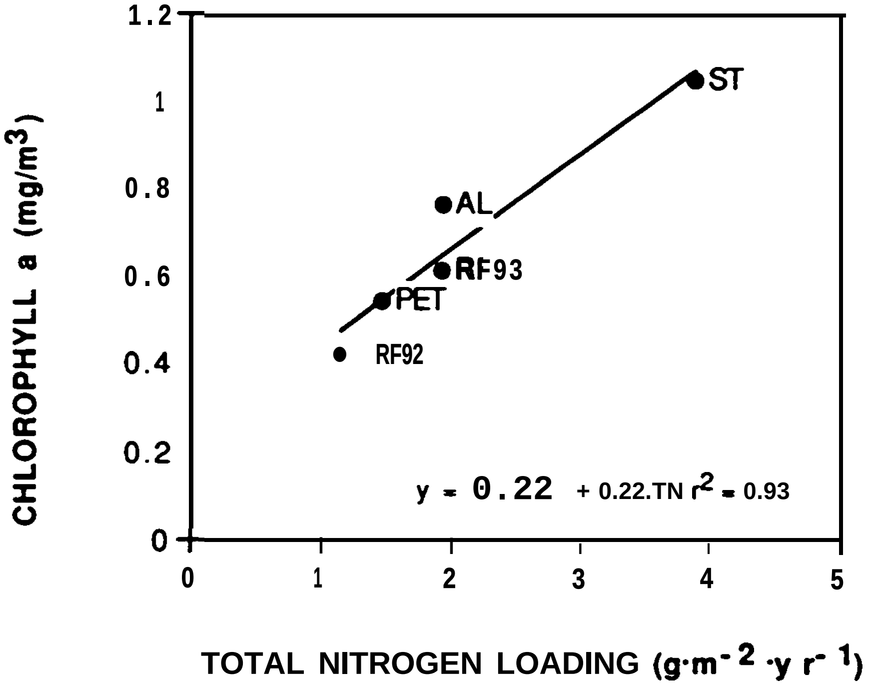


Figure 26. Relationship between areal loading of total phosphorus (above) and total nitrogen (below), and mean summer epilimnetic chlorophyll concentrations for Redfish (RP), Pettit (PET), Alturas (AL) and Stanley (ST) lakes during 1993. Data for 1992 are also shown for Redfish Lake. Loading values are from Chapter 1.

(Spalding 1993). In 1993 almost no recruitment of kokanee was attributed to the inflow stream to Alturas Lake. It is not clear where the small fish targets present in the Alturas survey derive, but the acoustic evidence suggests that shore spawning by Alturas kokanee may be substantial. Alternatively, these small fish targets present in the 15-40 m depth strata could be a different species altogether. Results of 1993 trawling surveys may shed some light on this possibility.

The high densities of small fish targets in the epilimnia of Pettit and Stanley Lakes were likely redside shiners. Although these fish would likely consume zooplankton, their omnivorous feeding habits and propensity to remain in littoral regions make it unlikely that these fish would compete intensively with juvenile sockeye salmon for zooplankton food resources. Kokanee, on the other hand, would likely compete strongly with juvenile sockeye owing to their similar planktivorous feeding habits. Results from 1992 and 1993 acoustic surveys and zooplankton biomass patterns suggest that juvenile sockeye salmon would experience less competition from kokanee in Pettit Lake than in Stanley, Alturas or Redfish Lakes.

The pattern of fish abundance likely has influenced species composition, size structure, vertical distribution, and density of zooplankton among the lakes. The highly planktivorous feeding ability of kokanee suggests that lakes with high areal densities of kokanee should have lower densities of small-bodied zooplankton. This appears to be the case for Alturas Lake, where the highest pelagic fish densities and lowest crustacean zooplankton biomass were present. The decline of zooplankton biomass from 1992 to 1993 may have resulted from increased densities of fish (Fig. 21). This suggestion also appears to hold in Pettit Lake where the lowest densities of kokanee were present with high abundances of large-bodied cladocerans. This pattern does not appear to hold, however, in Stanley Lake where relatively high kokanee densities (400 ha^{-1}) were present with large-bodied cladocerans. The diel vertical migration exhibited by Daphnia in Stanley Lake may allow this taxa to thrive in the presence of large numbers of kokanee.

Diel vertical migration shown by Daphnia in Stanley Lake appeared to follow a pattern of migrating down in the water column in the morning and then back up again in the evening. In Redfish Lake *Daphnia* remained in the metalimnion throughout the day where algal food resources were sufficient and temperatures were suitable for egg production (Stich and Lampert 1981). The lack of any clear pattern of migration in Redfish Lake in contrast to Stanley Lake may reflect the varying degrees of predation pressure among the two lakes (Clark and Levy 1988, Stich and Lampert 1981, Gabriel and Thomas 1988). The estimated abundance of *O.nerka* in 1993 in Stanley Lake was around 400 fish ha^{-1} versus about 300 fish ha^{-1} in Redfish Lake. In addition, Redfish Lake zooplankton may not migrate as strongly as in Stanley because deep light penetration (1% light levels >35 m) would require long migrations to escape potential predation by *O.nerka*, whereas refuge in the more productive Stanley could be as shallow as 10 m (Levy 1987, Clark and Levy 1988). Alternatively, the larger algal populations in Stanley compared to other lakes may have allowed Daphnia reproduction to exceed losses due to planktivorous fish.

The Sawtooth Valley Lakes are highly oligotrophic lakes with chlorophyll levels less than 1 mg m^{-3} for most of the year. Measurements of algal photosynthesis also indicated that the lakes are unproductive. They also had the lowest chlorophyll a levels when compared to eight other 0. nerka lakes in Idaho investigated by Rieman and Meyers (1992). However, ascertaining productivity of these lakes is somewhat complicated by the role of the deep chlorophyll layer (DCL).

Our work in 1993 gives some insight into the composition of the DCL and its significance to overall primary productivity of the Sawtooth Valley Lakes. Two- to more than ten-fold increases in chlorophyll a occurred between the epilimnion and the DCL. But using only chlorophyll a to examine phytoplankton standing crop does not fully describe the algal community because algae will often increase pigment production under low-light conditions (Steemann Nielsen and Jorgensen 1968). This seems to be the case in the Sawtooth Valley Lakes, where the chlorophyll:algal biovolume ratio was greater at the 1% light level than in the epilimnion (Fig. 27). The higher biovolume and chlorophyll levels in the DCL may contribute to the substantial primary productivity measured in the metalimnia and hypolimnia of most of the lakes (Figs. 13 and 14). The extremely clear epilimnia of most of the lakes permits adequate light to penetrate to considerable depth, thus allowing phytoplankton to grow. In all of the lakes except Stanley, over 50% of the primary production occurs below the epilimnia in the DCL.

Deep chlorophyll layers in lakes have varying characteristics which suggest multiple processes contributing to their formation and maintenance. In mictic lakes, the DCL generally forms and progresses along with summer thermal stratification (Shortreed and Stockner 1990). DCL can be controlled by light attenuation in the water column - several researchers have found the DCM to occur at or near the 1% light level (Fee 1976, Richardson et al. 1984). The DCL may also be controlled by differential sinking rates of algal cells and/or better nutrient climate at depth (Fee 1976, Shortreed and Stockner 1990, Gasol et al. 1992). The possibility of top-down trophic control contributing to the DCL may be particularly applicable in 0. nerka lakes because juvenile 0. *nerka* do not utilize the epilimnion for much of the growing season due to temperature limitations. Instead, they prey on zooplankton in the meta- and hypolimnion (LeBrausser et al. 1976, Shortreed and Stockner 1990).

While all of these factors seem applicable to the DCLs of the Sawtooth Valley Lakes, an additional reason may be the plunging inflows of these lakes (Chapter 1). Colder, thus denser, inflows are delivering nutrients to the metalimnion and hypolimnion of these lakes, contributing to the ultra-oligotrophic status of the epilimnion and the formation of the DCL.

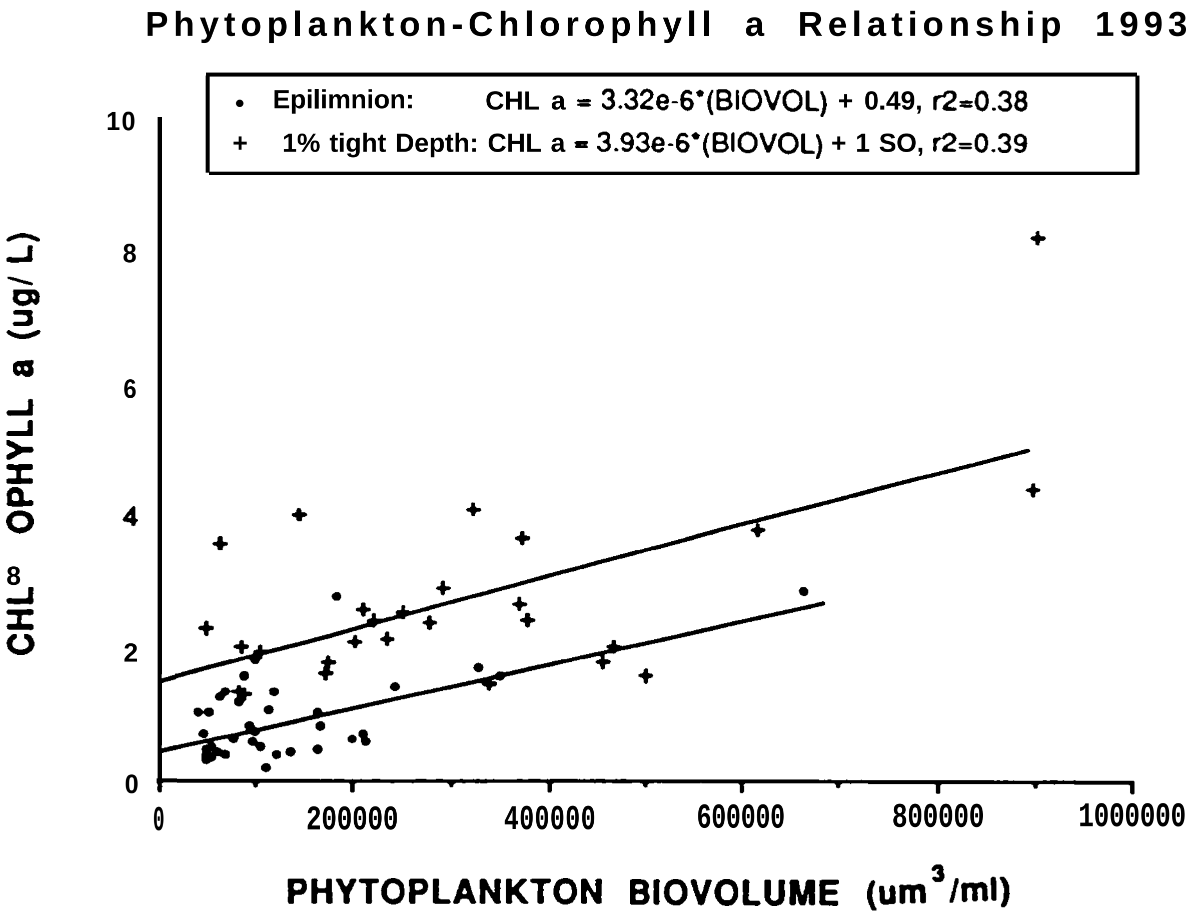


Figure 27. Relationship between chlorophyll a and phytoplankton biovolume in the Sawtooth Valley Lakes during 1993. Shown are points for samples collected from the epilimnion, and for samples collected at the 1% light level.

Lake fertilization would decrease the amount of light reaching the DCL. Much of the DCL occurs around the compensation point. Below this point, metabolic costs exceed gains from photosynthesis (the actual compensation point varies for individual taxa of phytoplankton, but the 1% light level is nominally given as the bottom of the photic zone). Shading from fertilization would decrease the depth of the 1% light level and the width of the photic zone. This could affect the distribution of lake primary productivity. In shading experiments conducted for two weeks during August of 1993 in the DCL of Redfish and Pettit Lakes, chlorophyll a levels decreased by 9-24% in shaded 10-L mesocosms when compared to control mesocosms'. Thus, gains in primary productivity from lake fertilization need to carefully be evaluated against productivity decreases in the DCL.

'Results of the shading experiments and the plunging inflows will be reported in a Master's thesis of H. Gross, to be completed in the summer of 1994.

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Table 1. Mean summer Secchi depths and chlorophyll a levels for the Sawtooth Lakes in 1992 and 1993.

	Mean Summer Secchi Depth (m)		Mean Summer Epilimnetic Chlorophyll a Concentration (µg/L)	
Lake	1992	1993	1992	1993
Redfish	13.5	11.9	.42	0.61
Alturas	14.0	10.4	.50	0.76
Pettit	15.57	13.2	.40	0.54
Stanley	8.7	7.3	.60	1.05
Yellow Belly	13.0	9.2	.44	0.63

Table 2. Spring overturn and mean seasonal (May-October) epilimnetic nutrient concentrations for the Sawtooth Valley Lakes, 1992 and 1993. All values are in µg/L, except for TN:TP ratio, which is by weight. "1993 means for Y. Belly Lake are for June-Sept. [b]Mean contains one outlier which will be re-tested; actual mean may be lower.

1992	Spring Overturn		May-October Mean						
Lake	TP	TN	TP	SRP	NO,-N	NH,-N	TKN	TN	TN:TP
Redfish	6	66	8.3	1.9	7.1	---	44	51	6.1
Alturas	10	74	7.9	1.1	4.0	---	74	78	9.94
Pettit	9	87	5.9	2.0	4.6	---	83	87	14.7
Stanley	11	102	8.0	1.9	4.7	---	87	92	11.5
Y. Belly	9	74	8.2	2.3	9.4	---	101	110	13.4

1993	Spring Overturn		'May-October Mean						
Lake	TP	TN	TP	SRP	NO,-N	NH,-N	TKN	TN	TN:TP
Redfish	10	63	6.8	1.0	2.4	3.2	63	65	9.7
Alturas	15	87	8.6	1.2	3.0	2.6	92	95	11.0
Pettit	9	109	6.5	2.2	2.1	3.0	73	76	11.7
Stanley	16	167	7.9	1.2	4.7	[b]11.6	89	94	11.9
Y. Belly	--	--	4.3	1.7	5.0	[b]12.5	74	79	18.4

Table 3. Chlorophyll a levels in the deep chlorophyll maxima (DCL) in the Sawtooth Valley Lakes in 1993.

Lake	Mean DCL Peak in Chlorophyll a (µg/L)	1993 DCL Peak Range (June-October)	% Increase in Mean DCL Peak Over Mean Epilimnetic Chlorophyll
Redfish	3.09	2.21-4.75	407%
Alturas	2.70	2.00-3.37	255%
Pettit	3.31	2.01-4.54	513%
Stanley	2.40	1.58-4.49	119%
Yellow Belly	6.57	2.64-8.17	943%

Table 4. Common phytoplankton genera of the Sawtooth Valley Lakes.

Taxa	Mean Equivalent Sperical Diameter (µm)
Bacillariophyta	
Asterionella	7.6
Cyclotella	9.1
Fragellaria	11.3
Melosira	8.4
Navicula	10.3
Synedra	8.6
Tabellaria	14.9
Chlorophyta	
Chlorococcales	
*Chl*amydomonas	5.7
Chlorella	10.6
Oocystis	24.7
unidentified Chlorococcales	4.0
Desmidiaceae	16.1
Arthrodesmus	
Cosmariuzn	
Spondylosium	
Starastrum	
Chrysophyta	
Dinobryon	12.6
Cyanophyta	
Anabaena	6.1
Microcystis	20.9
Oscillatoria	**6.2**
Dinophyta	21.7
Peridinium	
Glenodinium	

Chapter 3

Effects of Nutrient Enhancement on Plankton and the Growth of Juvenile Salmon in Redfish Lake

by

Phaedra Budy
Chris Luecke
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and
Howard P. Gross

INTRODUCTION

The Snake River strain of sockeye was declared endangered in 1991 in response to the decline in the number of anadromous adults returning to nursery lakes in the Snake River watershed. These fish historically migrated over 900 miles from the Pacific ocean to five lakes in the Sawtooth Valley of central Idaho (Redfish, Pettit, Alturas, Stanley, and Yellow Belly lakes) which served as both the spawning grounds for the returning adults and as nursery lakes for juveniles (Bjornn et al. 1968). The decline in the number of adults returning from the ocean has likely led to decreased productivity of nursery lakes in that fewer decomposing carcasses added marine nutrients to the freshwater lakes (Krohkin 1967, and Donaldson 1967 *in* Kyle et al. 1993). Because whole-lake fertilization has successfully increased sockeye production in coastal Alaskan and British Columbia Lakes (Kyle et al. 1988; Stockner 1987, 1992), nutrient enhancement has been proposed as one possible management strategy for aiding in the recovery of this endangered population. However, this approach to sockeye management could decrease water clarity and thus the aesthetic value of the lakes to recreationists. The potential conflict in management goals suggests that for whole lake fertilization to be implemented, strong evidence that nutrient enhancement improves sockeye growth or survival should be demonstrated and weighed against the expected loss of water clarity.

We designed and performed a series of fertilization experiments in large limnocorral enclosures in one of the five lakes (Redfish Lake). The objectives of the experiments were: (1) to determine whether juvenile sockeye growth could be enhanced through nutrient additions, and: (2) to evaluate the effect of increased nutrient loading on water clarity.

METHODS

Fertilization experiments were performed in large limnocorral enclosures. The limnocorrals were 5 m in diameter by 18 m deep and made of impermeable, fiber-reinforced polthyethlene, floated at the surface and weighted at the bottom. Each bag was slowly dropped (12 hours) through the water column with the bottom open so that the initial conditions were similar to the lake. There were two replicates for each treatment:

2 controls with fish
2 low nutrient additions with fish
2 high nutrient additions with fis

The experiment was designed to determine whether fish growth could be enhanced by nutrient additions, and to assess how the amount of nitrogen and phosphorous added would affect water transparency, phytoplankton and zooplankton biomass.

At the start of the experiment, on June 29, mean TP and TN concentrations in the limnocorrals were 5.6 (range of 5.1-5.9) μg/L and 67 (range of 63-76) μg/L, respectively. In the low fertilization treatment we increased phosphorus by 75% and nitrogen by 150% over the experiment (i.e. 4.2 μg-P and 84 μg-N/L were added). In the "high" nutrient treatment phosphorus and nitrogen concentrations were increased by 150% and 250% respectively (8.4 μg-P/L and 168 μg-N/L). Nutrients were added in the form of $(NH_4)_2HPO_4$ and NH_4NO_3, at a 20:1 mass TN:TP ratio. The high TN:TP ratio was used to reduce the likelihood of stimulating nitrogen-fixing cyanobacteria (Schindler 1977). Weekly, nutrient solutions were stirred into each limnocorral at the surface. In order to promote rapid initial growth of the plankton community, 40% of the nutrients were added during the first week of the experiment; the remaining 60% was added in equal parts over the following 9 weeks.

In lieu of endangered sockeye salmon, we used kokanee salmon (0. nerka) in our experiments. The juvenile kokanee used in the

Redfish experiment were caught shortly after they emerged from the gravel of Fishhook Creek, an inflow to Redfish Lake. The fish were temporarily reared in a hatchery until the experiment was begun. Initial total lengths and wet and dry weights were measured on a sub-sample of fish prior to placing 25 juvenile kokanee in each corral. Approximately half way through the experiment (31 July 1993), 12-14 fish were removed from each enclosure with dip and lift nets, measured (total length), weighed (wet), and preserved for stomach analysis. At the end of the experiment, the fish were anesthetized by adding approximatly 70 kg of dry ice to each corral, and they were then removed with dip nets and SCUBA divers. Between 80 and 92% of the fish were recovered from the individual corrals. The fish were then measured (total length), weighed (wet), and pumped for stomach contents. All fish were then frozen and subsequently dried to constant weight. Dry weights were used because they provide a more accurate estimate of changes in tissue weight if water content of the fish varies during a sampling interval (Brett et al. 1969).

The six limnocorrals were sampled weekly or biweekly for different limnological parameters. Temperature, oxygen, and light profiles were sampled bi-weekly using a YSI and/or Hydrolab meter to measure temperature and oxygen, and a LiCor radiometer to measure light. Measurements were taken every meter from the surface to 17 m, or just off the bottom of the limnocorral. The temperature and oxygen profiles were compared among treatments and to profiles observed in Redfish Lake.

Secchi disk measurements were taken weekly to estimate water clarity. Repeated measure ANOVAs were used to analyze the Secchi readings for statistical significance accounting for time and initial differences among treatments (Wilkinson 1990).

Water for chlorophyll a analysis was also collected weekly from the epilimnion (O-6 m) and from 0 to 17 m in each limnocorral with a depth-integrating Tygon tube. Biweekly, we collected additional chlorophyll samples from the metalimnion and near the bottom of the limnocorral with a 4 L Van Dorn bottle. Initially, and at 4 and 8 weeks into the experiment, ^{14}C primary productivity measurements were made at 4-5 depths. Additional depths were sampled for chlorophyll a analysis concurrent with the ^{14}C measurements.

On three dates (initially, and at 4 and 8 weeks into the experiment) we saved epilimnetic, metalimnetic, and near-bottom water for nutrient analyses. Samples collected for nutrient analyses were placed in polyethylene bottles which were first rinsed with 0.1 N HCl and then 3 times with aliquots of the actual sample. Nutrient samples were stored in an ice cooler and then frozen upon return to our field laboratory. We collected samples for nutrient analyses to measure the effects of nutrient additions on total nitrogen (TN) and total phosphorus (TP) concentrations. TN was calculated from the sum of Total Kjeldahl Nitrogen (TKN) and nitrate+nitrite nitrogen (NO_3-N). Unfiltered water was used for TKN and TP analyses; samples analyzed for (NO_3-N) were filtered through 0.45-µm membrane filters. TP samples underwent a persulfate digestion and were then analyzed calorimetrically in our lab (Utah State University Limnology Laboratory) using the molybdate - absorbic acid method. Nitrogen analyses were conducted by University of California (Davis) Limnology Laboratory calorimetrically using a Kjeldahl digestion for TKN and the hydrazine method for NO,. Replicates, spikes, and standard solutions were used for quality control/quality assurance.

On the dates and depths that nutrient samples were collected, we also preserved samples for phytoplankton enumeration with Lugol's iodine solution. A 100-ml aliquot from each phytoplankton sample was filtered through a 0.45-µm cellulose filter. The filters were cleared and permanently mounted, according to the method of 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 Likens 1990). Phytoplankton were taxonomically classified as follows: Cyanophyta (blue-green algae), Chlorophyta (green algae), Chrysophyta (*Dinobryon* sp.), Bacillariophyta (Diatoms), and Dinophyta (*Peridinium* sp.) .

Chlorophyll a measurements were used as an alternative measure of phytoplankton standing crop. Two 50-ml aliquots per sample were filtered through 0.45-μm cellulose acetate membrane filters. Filters were either temporarily frozen or placed directly into 6 ml of 100% methanol for chlorophyll a pigment extraction in the dark for 24-48 hours. The extracts were then analyzed before and after acidification (Holm-Hansen and Riemann 1978) using a Turner model 111 fluorometer. Corrections were made for phaopigments. The fluorometer was calibrated using commercial chlorophyll a standards which were verified spectrophotometrically.

Rates of *in* situ primary production were measured with the ^{14}C-technique (Wetzel and Likens 1990). Water samples from each depth analyzed was placed into three 25-ml glass scintillation vials, taking care not to expose the plankton to direct sunlight. Each vial was inoculated with 80 μL of 25 μCi/ml of $^{14}CHO_3$. To measure non-photosynthetic ^{14}C uptake, we inoculated one vial from each depth with 150 μL with Diuron (dichloro-phenyl-dimethylurea; DCMU), a photosynthetic inhibitor. The vials were resuspended in the water column in clear acrylic plastic tubes hung from an incubation line. Incubations were normally conducted from 1000 to 1400 hrs (Mountain Standard time). Within 2 h of the end of the incubation the entire contents of each vial was filtered through 0.45 μm cellulose nitrate filters (Micro Filtration Systems) and rinsed with 0.1N HCl. They were then air dried, and subsequently counted by liquid scintillation spectrometry using Readysafe' cocktail. Counting efficiency varied between 91 and 93% in all of the samples. Production rates were calculated by subtracting carbon uptake in the DCMU treatments from the light treatments. Dissolved inorganic carbon was estimated from pH, and alkalinity measurements determined with the Gran procedure (Wetzel and Likens 1990). Productivity in the water column was partitioned into that occurring in the epilimnion and in the lower strata. Because stratification was not clearly defined for much of the summer, we used a nominal depth strata of 0-7.5 m for calculating the production that occurred in the epilimnion.

To evaluate periphyton accumulation in the limnocorrals, we suspended a weighted 17m x 10-cm wide strip of polyethylene limnocorral material in the center of each mesocosm. Three times during the season (at approximately 4, 8, and 11 weeks into the experiment), we measured periphyton accumulation on each strip. Two, 14-mm diameter disks were bored from each periphyton strip at depths of 0.5, 5, 10, 13, and 17 m. The disks were held in an ice chest, returned to the field laboratory and extracted into 6-ml aliquots of 100% methanol. We then analyzed the samples for chlorophyll *a* fluoremetrically as described above.

Zooplankton were sampled from 17-10 m and from 10-0 m using a closing 80-pm mesh net, 35 cm in diameter and 150-cm long equipped with an anti-reverse General Oceanics flow meter. Samples were immediately preserved in formalin-sucrose and subsequently counted and measured. Biomass was calculated using length-mass regressions (McCauley 1984; Koenings et al. 1987). Densities were corrected for the volume of water sampled. Cladoceran egg-ratios were calculated as an estimate of zooplankton production (Paloheimo 1974). Repeated measure ANOVAs were used to analyze zooplankton data for statistical significance accounting for time and initial differences among treatments (Wilkinson 1990).

RESULTS

Temperature and oxygen profiles in the limnocorrals were very similar and closely paralleled those observed in the lake (Figure 1). Oxygen conditions were above 7.5 mg/l throughout all limnocorrals for the duration of the experiment and thus were suitable for fish growth. Surface temperatures ranged from 11.3°C to 16°C but at the bottom of the limnocorral remained below 7.5 °C.

Secchi depth transparency was deepest in control treatments, second deepest in low nutrient treatments, and shallowest in high treatments (Figure 2), and these differences were statistically significant (ANOVA, repeated measures, df= 24, F= 3.306, $P < 0.001$).

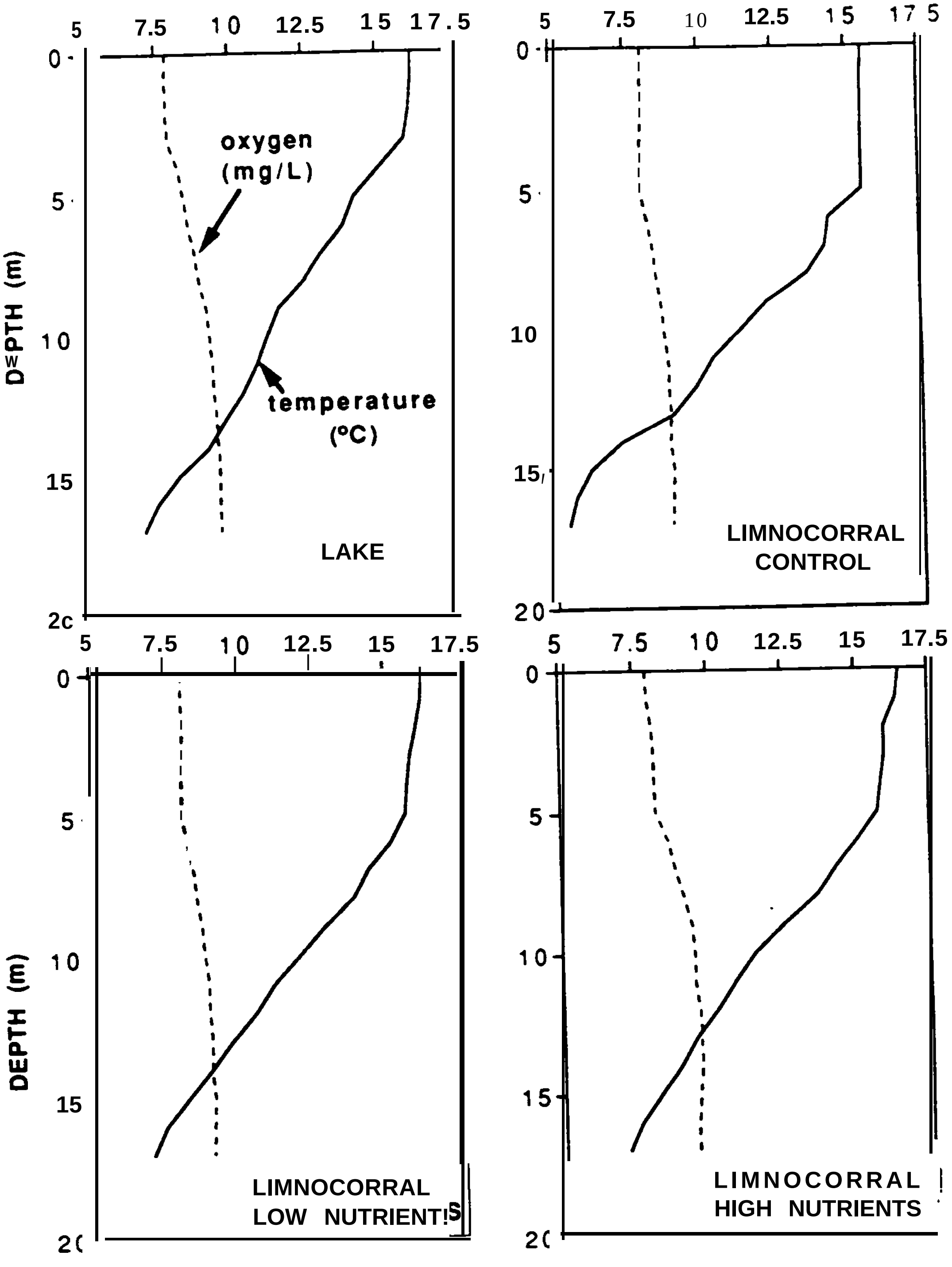

Figure 1. Temperature and oxygen profiles from one of each treatment limnocorrals in Redfish Lake on 8 August 1993 and of the ambient lake conditions on 6 August 1993.

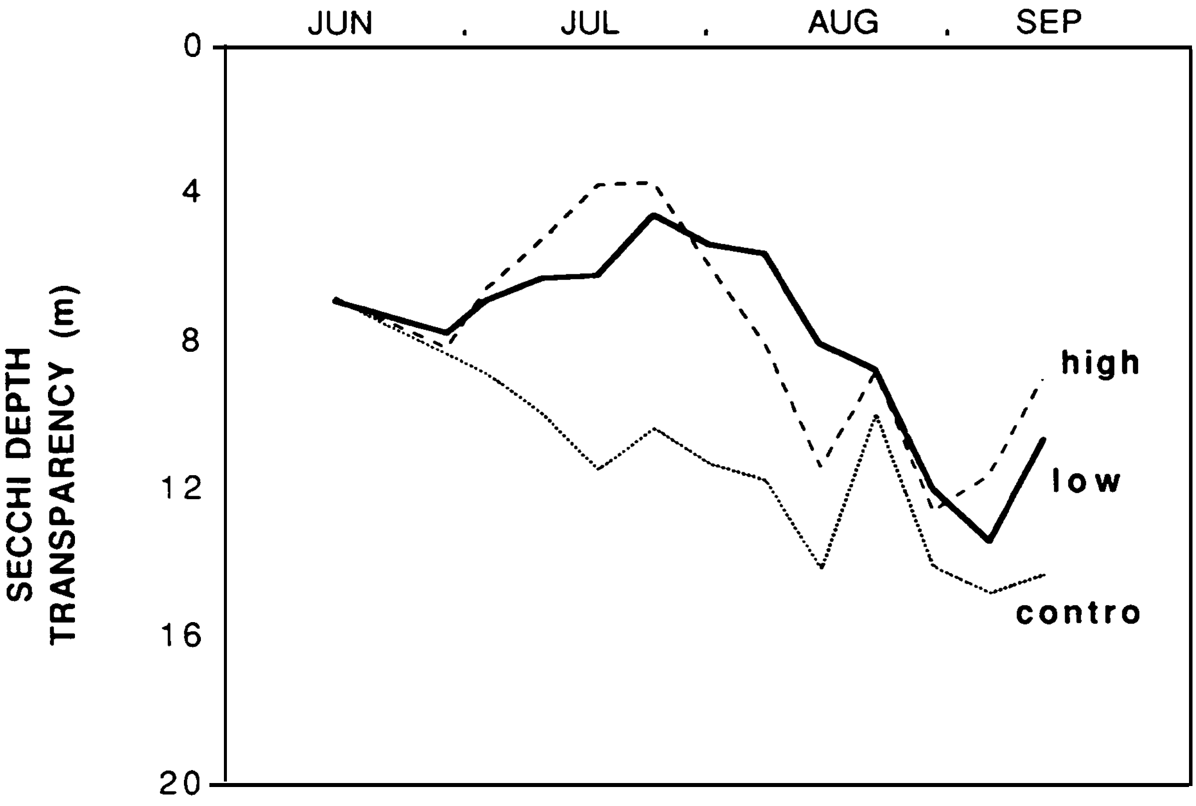

Figure 2. Secchi depths as a mean of two replicates for each treatment in Redfish Lake limnocorrals.

In the control treatments Secchi depths ranged from 7-15 m, whereas water transparency in the low and high nutrient treatments were 4-6 m less during July and most of August.

Mean chlorophyll levels in the low and high nutrient treatments were greater than the controls in all samples analyzed (Figures 3 and 4). The epilimnetic chlorophyll levels in the control treatments followed a trend similar to the lake during the experiment, although the limnocorral values were slightly higher (Figure 5). The mean weekly concentrations of chlorophyll in the 0-17m integrated samples were 0.83, 1.80, and 2.60 µg/L for the control, low, and high treatments, respectively. Values ranged from 0.57-1.07 µg/L for the controls, 1.22-2.70 µg/L for the low treatments, and 1.26-4.84 µg/L for the high treatments over the 11-week experiment (Figure la). The mean O-17 m chlorophyll level for the high treatments were greater than that of the low treatments on 9 of 11 weekly samplings after nutrient additions began.

Chlorophyll concentrations in the epilimnions of the corrals were lower than the integrated whole water column estimates. The means of the weekly epilimnetic chlorophyll levels were 0.65, 1.94, and 2.19 µg/L for the control, low, and high treatments, respectively. These values ranged from 0.36-l-16 µg/L for the controls, 1.49-3.41 µg/L for the low treatments, and 1.20-4.32 µg/L for the high treatments (Figure 3 b). The mean epilimnetic chlorophyll values for the low treatment exceeded that of the high treatment on 8 of the 11 weekly samplings after fertilization began.

The phytoplankton community structure was similar in all 3 treatments at the start of the experiment (Fig. 4, see 29 June 1993), with a mixture of diatoms, chlorophyta, and *Dinobryon* at all depths. The dominant diatoms were *Synedra* and *Tabellaria* spp., while *Cyclotella* and *Melosira* spp. were subdominant. The dominant chlorophyta were *Chlorella* and small *Chlorococcales* spp.

After 4 weeks (Fig. 5, see 25 July 1993), the phytoplankton community structure in the control treatments were similar to that of 29 June 1993. Mean biovolumes declined by 35% in the epilimnia,

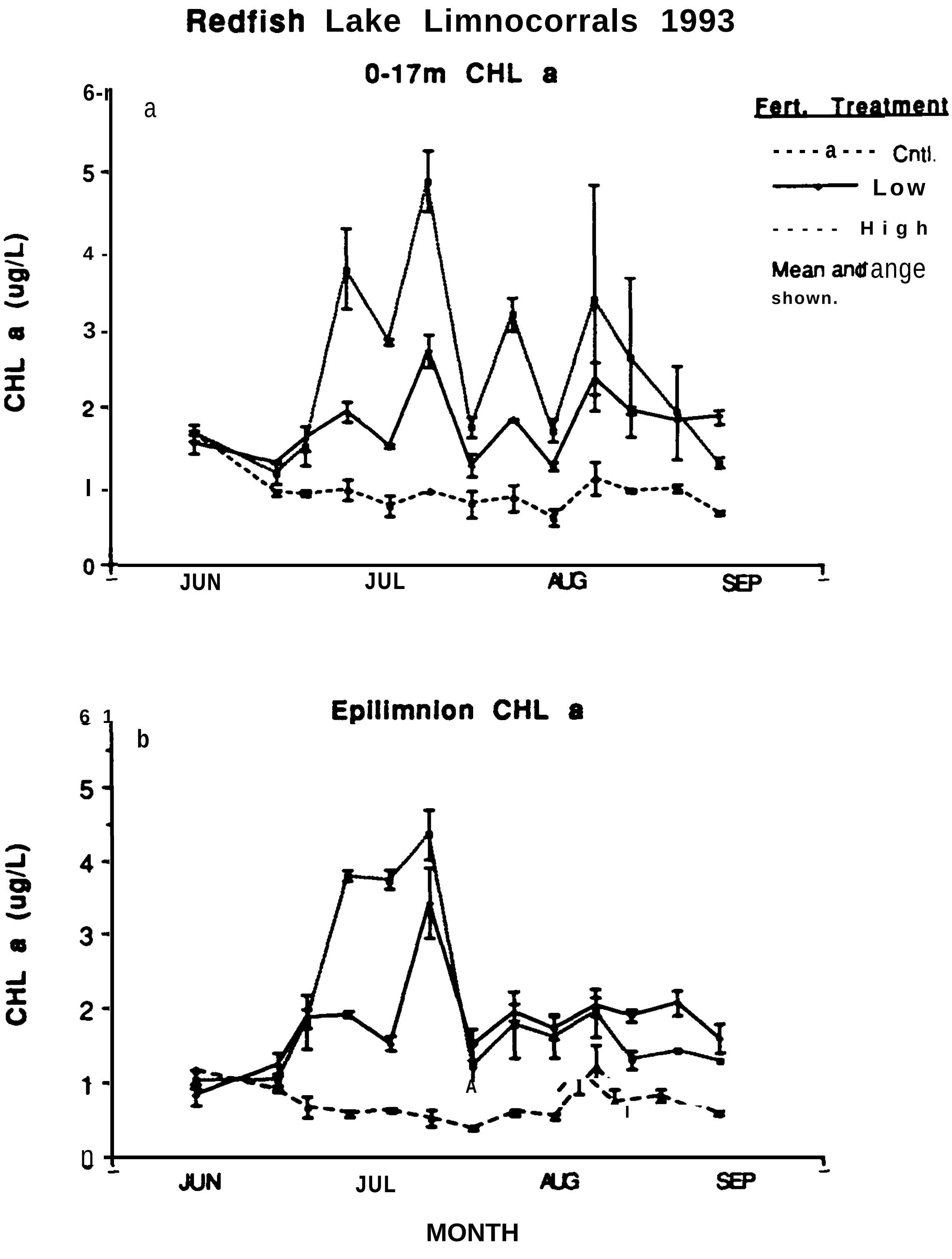

Figure 3. Mean chlorophyll levels ($\mu g/L$) from (a) 0-17 m and (b) epilimnetic tube samples collected from Redfish Lake limnocorrals, 1993. Control, low, and high treatments shown; error bars show range (n=2). Nutrient additions began on June 29, 1993.

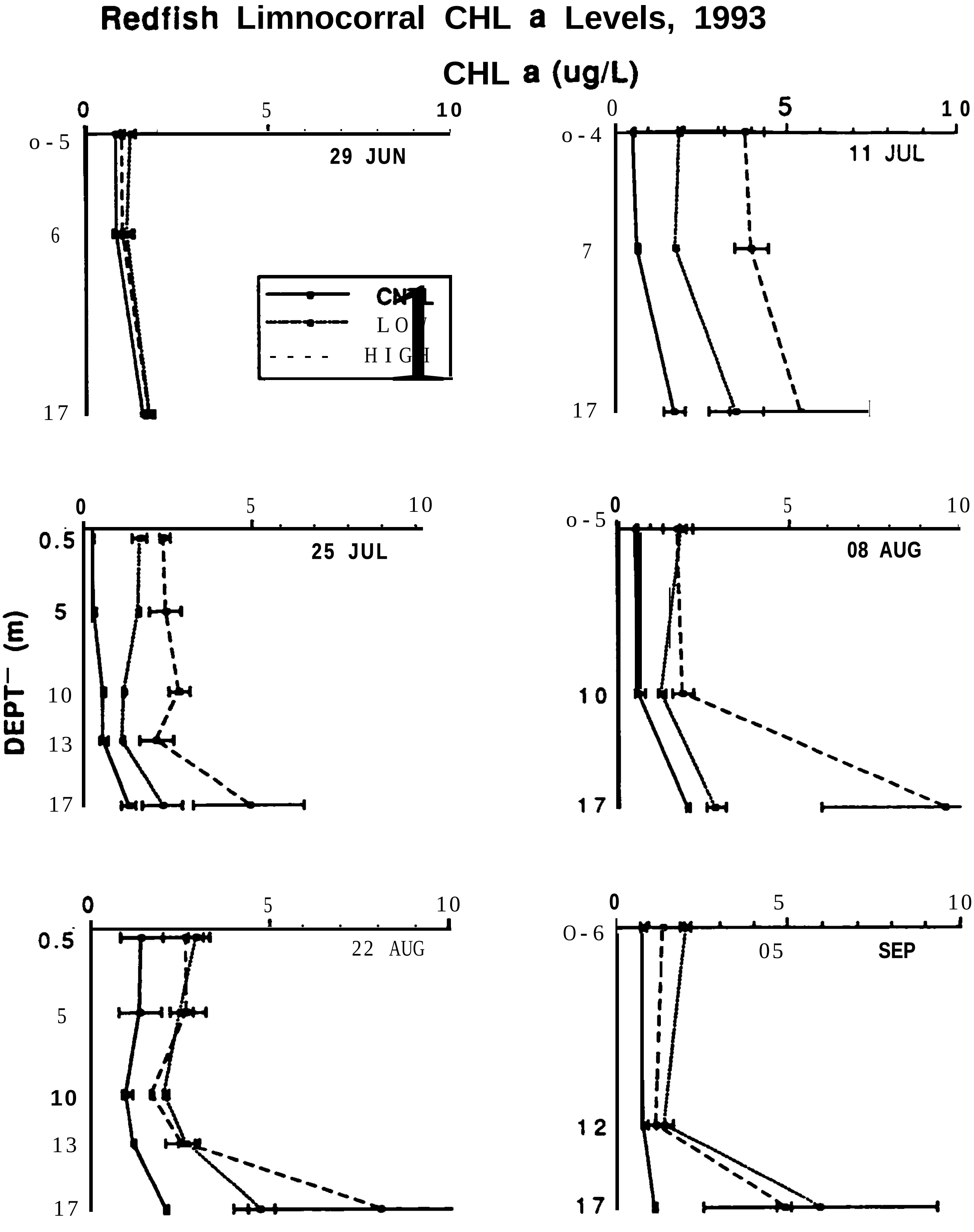

Figure 4. Depth profiles of mean chlorophyll levels ($\mu g/L$) in Redfish Lake limnocorrals, 1993. Control, low, and high treatments shown; error bars show range (n=2). Epilimnetic samples on 29 JUN, 11 JUL, 08 AUG, and 05 SEP were collected with a depth-integrating Tygon tube; all other samples were collected with a Van Dorn bottle.

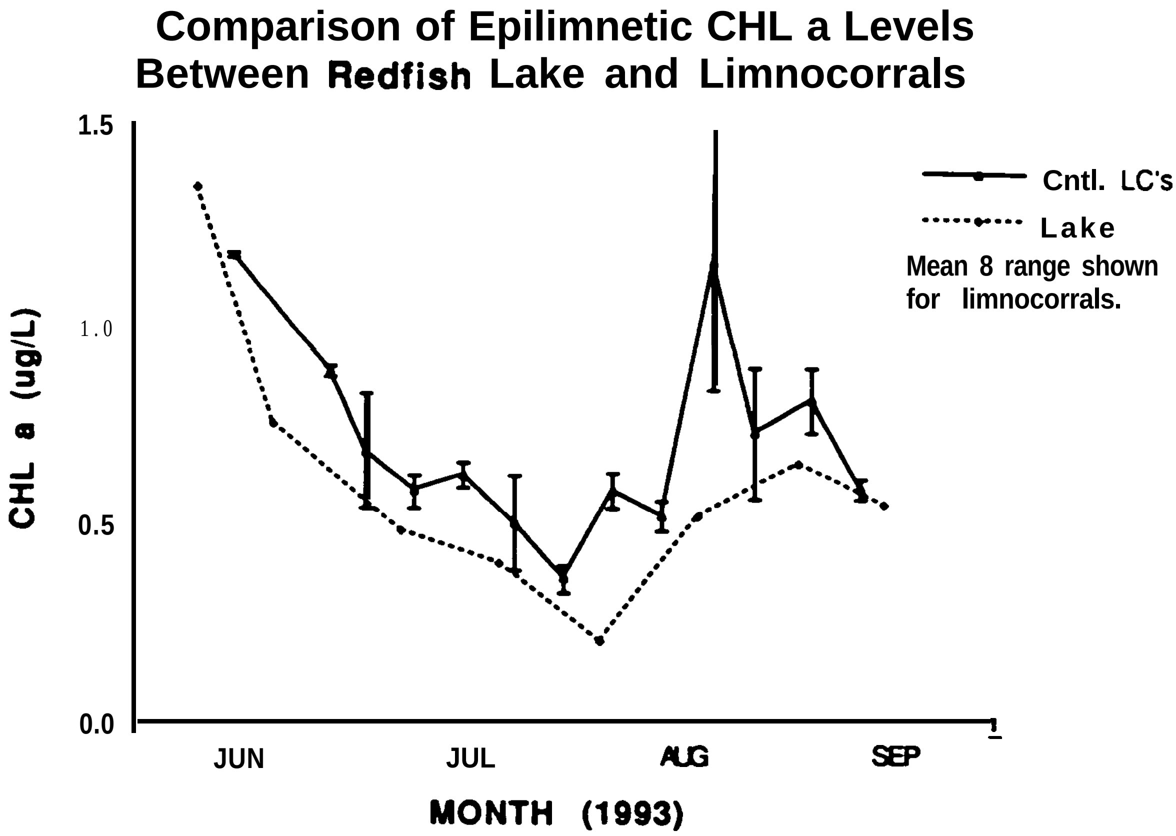

Figure 5. **Comparison** of **lake** and control **limnocorral epilimnetic** chlorophyll a levels for Rcdfish Lake, 1993. Error bars show range **for** limnocorrals **(n=2)**.

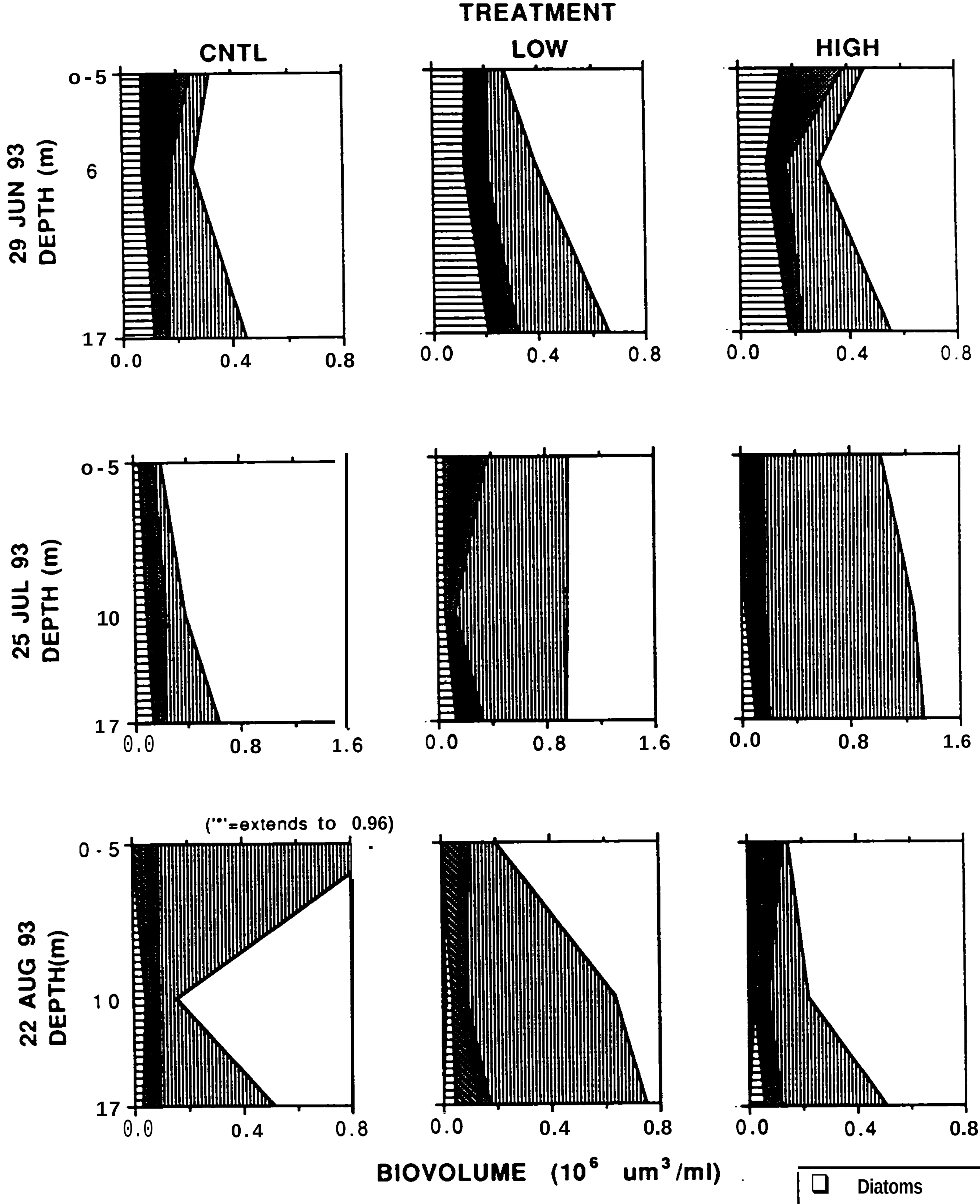

Figure 6. Mean algal biovolumes for **Redfish** Lake limnocorrals at epilimnetic, metalimnetic, and near-bottom depths, 1993. Graphs from same date shown horizontally; graphs from same treatment shown vertically. Note that biovolume axes for 25 JUL 93 are double that of the other two dates.

while increasing by -50% in the deeper waters. In both the LOW and HIGH treatments, the make-up of the communities was similar to that of 29 June 1993, but the mean algal biovolumes had increased at all depths (43-340%), with the highest biovolumes achieved in the HIGH treatment. The greatest increase was in the diatoms, by 70-860% in the LOW and 250-1000% in the HIGH treatments. In both treatments, *Synedra* and *Tabellaria* continued to be the dominant diatoms, while *Dinobryon* biovolumes dropped by 45-100%. The Chlorophyta generally increased in the LOW and HIGH treatments, with the greatest increases in the epilimnion of the LOWs (260%).

After 8 weeks (Fig. 5, see 22 Aug. 1993), biovolumes declined from 25 July 1993, with the exception of a tremendous bloom of *Tabellaria* in the epilimnia of the control treatments. This bloom generated a 380% increase in total epilimnetic biovolume of the control treatments, reaching absolute biovolumes greater than the LOW or HIGH treatments. However, biovolumes declined by 60% at 10 m and 18% at 17 m in the control treatments. In all treatments, *Tabellaria* clearly was the dominant diatom. *Dinobryon* and the Chlorophytes declined in all treatments, while *Peridinium* were observed for the first time during the experiment. Total biovolume declined in the LOW treatments by 21-79% and in the HIGH treatments by 61-85%, with the greatest declines in the epilimnia. A bloom of *Anabaena* made up 37% of the HIGH epilimnia biovolume.

Periphyton chlorophyll on limnocorral walls varied by three orders of magnitude over the course of the experiment (Figure 7a-f) and was markedly stimulated by nutrient additions in both the low and high treatments (Figure 7a). Periphyton was barely detectable by eye in the control treatments, whereas in the low, and particularly the high treatment, 2-5 mm thick aufwuch communities developed patchily on the walls of the limnocorrals. These accumulations were particularly noticeable in the epilimnion, but periphyton chlorophyll increased at all depths in all treatments as the season progressed (Figure 7 a-c).

In order to compare the total amount of algae stimulated by the nutrient additions, we estimated the combined amount of

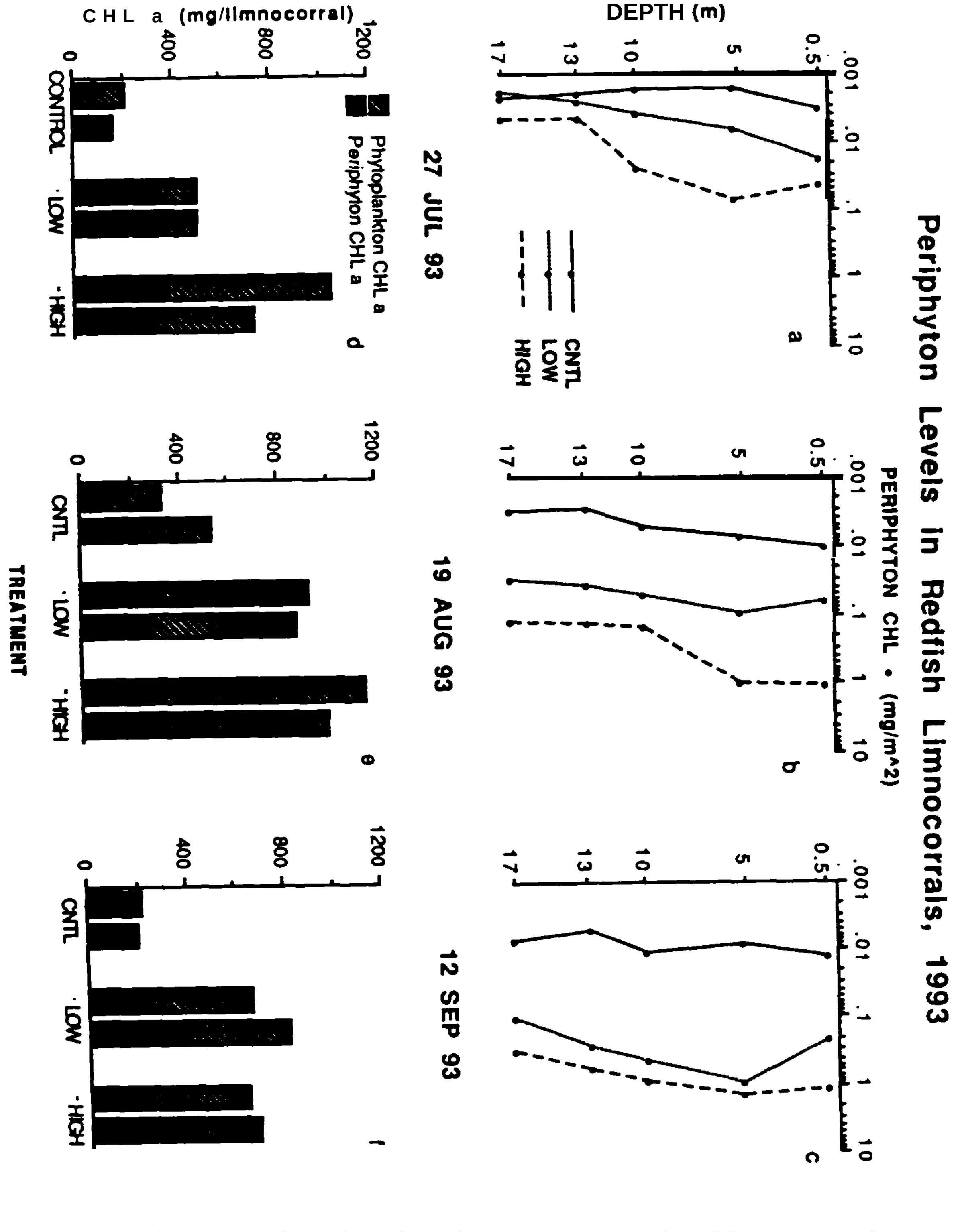

Figure 7. Periphyton levels in the **Redfish** Lake limnocorrals, 1993, as measured from periphyton strip sampling. Mean periphyton density at 5 depths for control, low, and high treatments (n=2), expressed as mg/m^2 of chlorophyll a, for three dates in 1993 (a, b & c). Whole limnocorral chlorophyll a, expressed as mg/corral, in phytoplankton and periphyton assemblages in each limnocorral for three dates in 1993 (d, e & f).

Redfish Limnocorral Nutrient Levels, 1993

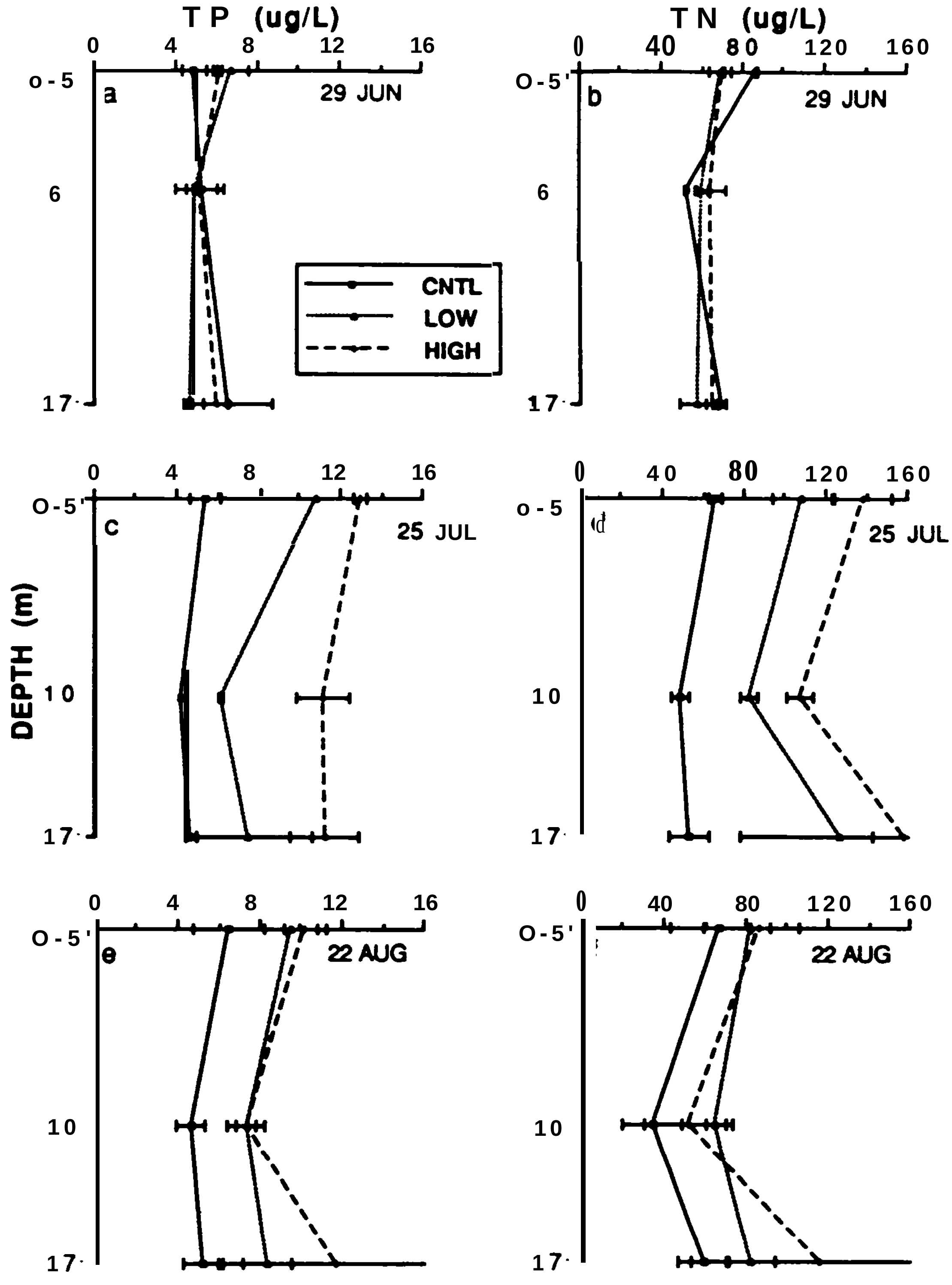

Figure 8. Depth profiles of total phosphorus (TP) and total nitrogen (TN) in Redfiah Lake limnocorrala, 1993. Control, low, and high treatments shown; error bars show range (n-2).

chlorophyll present in the periphyton and phytoplankton. Total chlorophyll levels in the fertilized limnocorrals always exceeded those in the controls (Figure 7 d-f). Total chlorophyll levels in the high treatments were greater than those of the low treatments on two dates (July 27 and Aug 19; Figure 7 d-e). However, on Sept. 12, combined chlorophyll in the low treatments was slightly greater than or equal to that of the high treatments (Figure 7 f).

As expected, nutrient additions increased TN and TP concentrations over initial and control concentrations (Figure 8). On 25 July, highest nutrient concentrations were in the high treatments, followed by the low and control treatments (Figure 8c and d). However, on 22 August, differences in nutrient levels between the high and low treatments were less clear, as the variability within treatments increased (Figure 8 e-f).

The nutrient additions not only stimulated algal biomass, but primary production as well. Fertilization particularly increased epilimnetic production (Figure 9a). Five days after the experiment began (4 July) primary production was exceedingly low in the control treatments, but fertilization had significantly increased production in the epilimnia of both the low and high treatments. Metalimnetic production rates, however, were similar in the controls and fertilized treatments. Three weeks latter (25 July) epilimnetic production was stimulated even more by both the low and high fertilizations, and the effects extended into the metalimnia. After four additional weeks (22 August) primary production was still enhanced by the nutrient additions, but the differences among treatments were less distinct.

The overall effect of fertilization on phytoplankton productivity was summarized by integrating production over the water column (Figure 9b). This analysis demonstrated that nutrient additions increased primary production 110-290% in the low fertilization treatment, and 90-490% in the high fertilization. The figure also shows that most of the increased production occurred in the epilimnia of the limnocorrals.

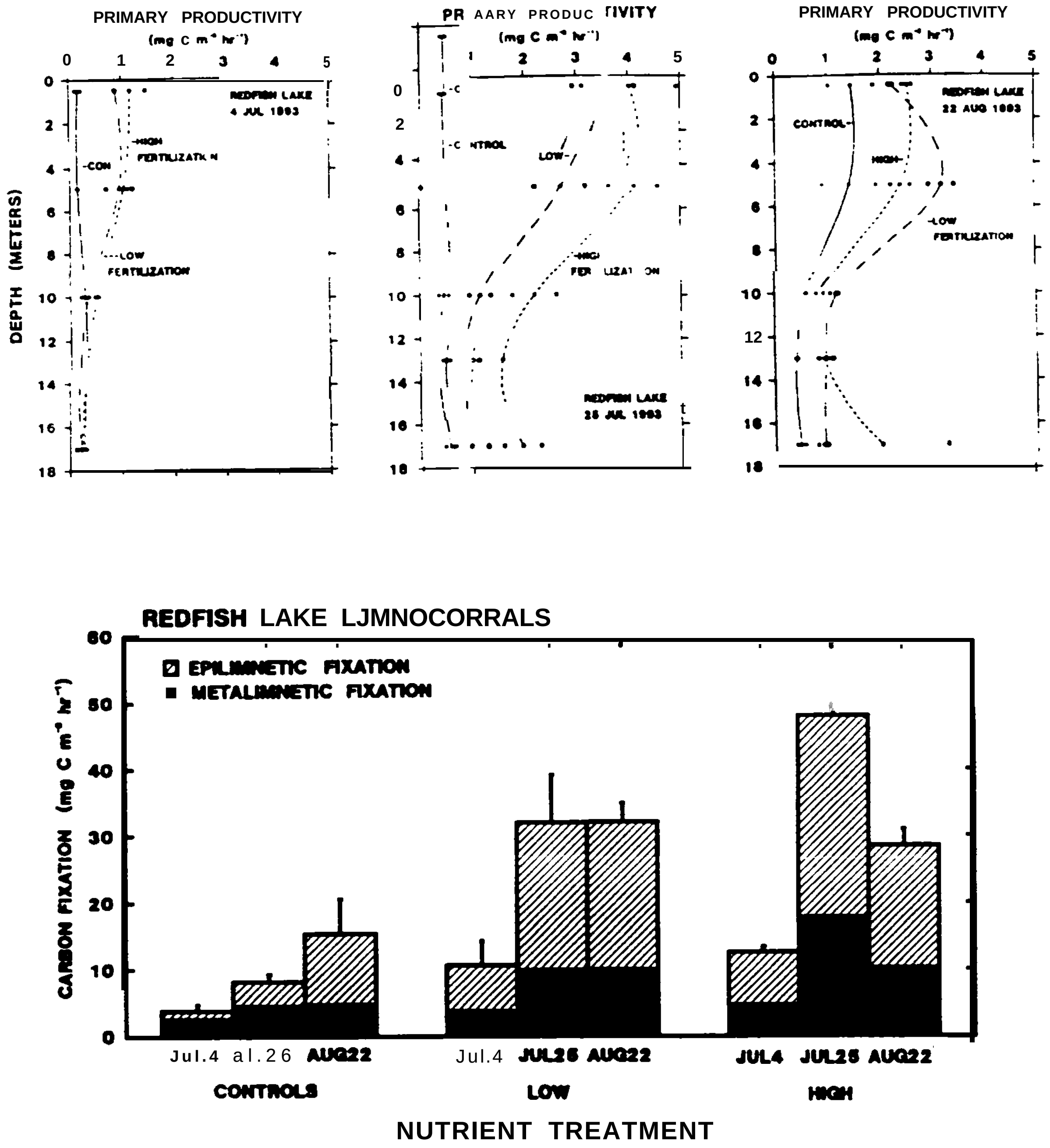

Figure 9. Rates of primary production measured in the **Redfish limnocorrals** on three dates during 1993. A) The upper figures show depth profiles **measured** at four (4 July) or five depths (25 July and 22 August) in the control (COW), low **fertilization (LOW)**, and high fertilization (HIGH) treatments.. **Two** replicate measurements and the mean value for each treatment are plotted.

B) The lower figure indicates total productivity in the three treatments integrated over the 17-m water column. Fixation rates are **separated** into that occurring in the **epilimnion (<7.5 m)**, and that in the **metalimnion.** The combined height of **each** histogram indicates total productivity.

Zooplankton biomass showed temporal trends during the experiment, but no clear response to nutrient additions was observed (Figure 10). Total zooplankton biomass was initially relatively high and then dropped to extremely low levels. A slight increase in zooplankton biomass was observed in mid-August after fish were permanently removed during the intermediate fish sampling. Twevle fish were removed from limnocorral #1, 13 from corrals 2 and 3, and 14 fish removed from the remaining corrals. Each limnocorral started with a different initial zooplankton biomass, but all corrals exhibited extremely low levels of zooplankton throughout the experiment.

Nutrient addition significantly increased (df= 24, F= 3.786, P=.001) egg production of adult female *Bosmina* in the 10-17 m depth strata (Figure 11), but no significant treatment effect was observed in the 0-10 m strata (p= 0.56). The highest *Bosmina egg* production was observed in the high nutrient treatment and the lowest in the controls with the low treatment intermediate. Zooplankton species composition throughout the experiment is shown in Figure 12. Both control limnocorrals (#1, 3) were dominated by *Bosmina* with high abundances of *Holopedium* for the first month of the experiment. A small peak in both Daphnia and *Holopedium* was observed in early August corresponding with the intermediate fish sampling where fish were removed from each corral. Low densities of cyclopoids and nauplii were present for the duration of the experiment.

The low nutrient treatment limnocorrals showed a similar pattern to the controls with the exception of generally higher and more consistent abundances of Daphnia and *Holopedium* (Figure 12). Both high treatments also showed a dramatic decline in all species other than *Bosmina* in late July.

No increase in sockeye growth in relation to nutrient enhancement was observed (Figure 13). The mean dry weight of kokanee at the end of the experiment was highest in the control treatment and lower but almost identical in the low and high nutrient additions (Figure 14). Fish in all of the limnocorrals

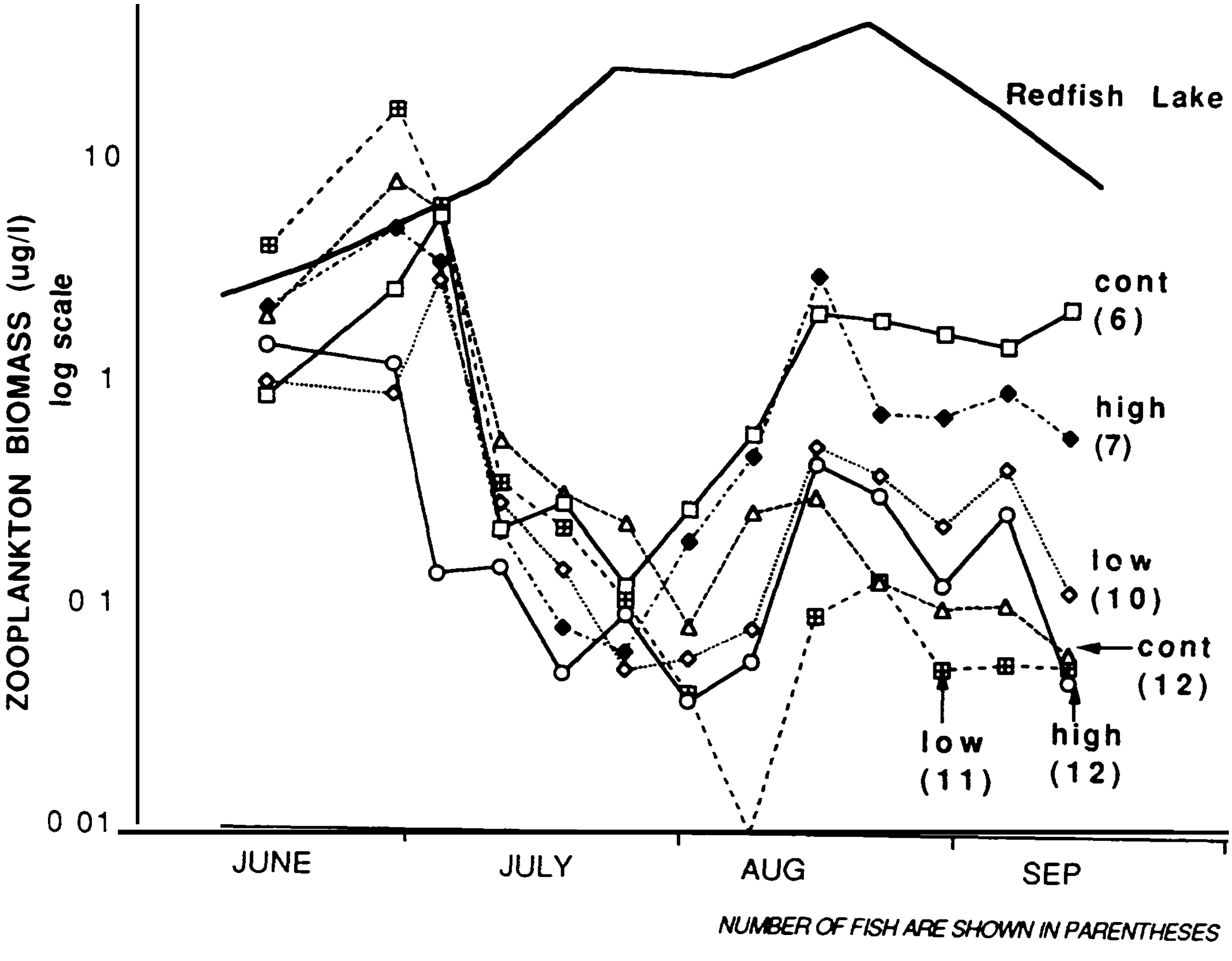

Figure 10. Total zooplankton biomass (μg/l) in Redfish Lake limnocorrals from the beginning of the experiment (15 June 1993) until the end (17 September 1993), and sampled in 10-0 m tows during regular lake limnology in Redfish Lake (thick solid line). Final number of juvenile kokanee collected in each limnocorral are shown in parentheses.

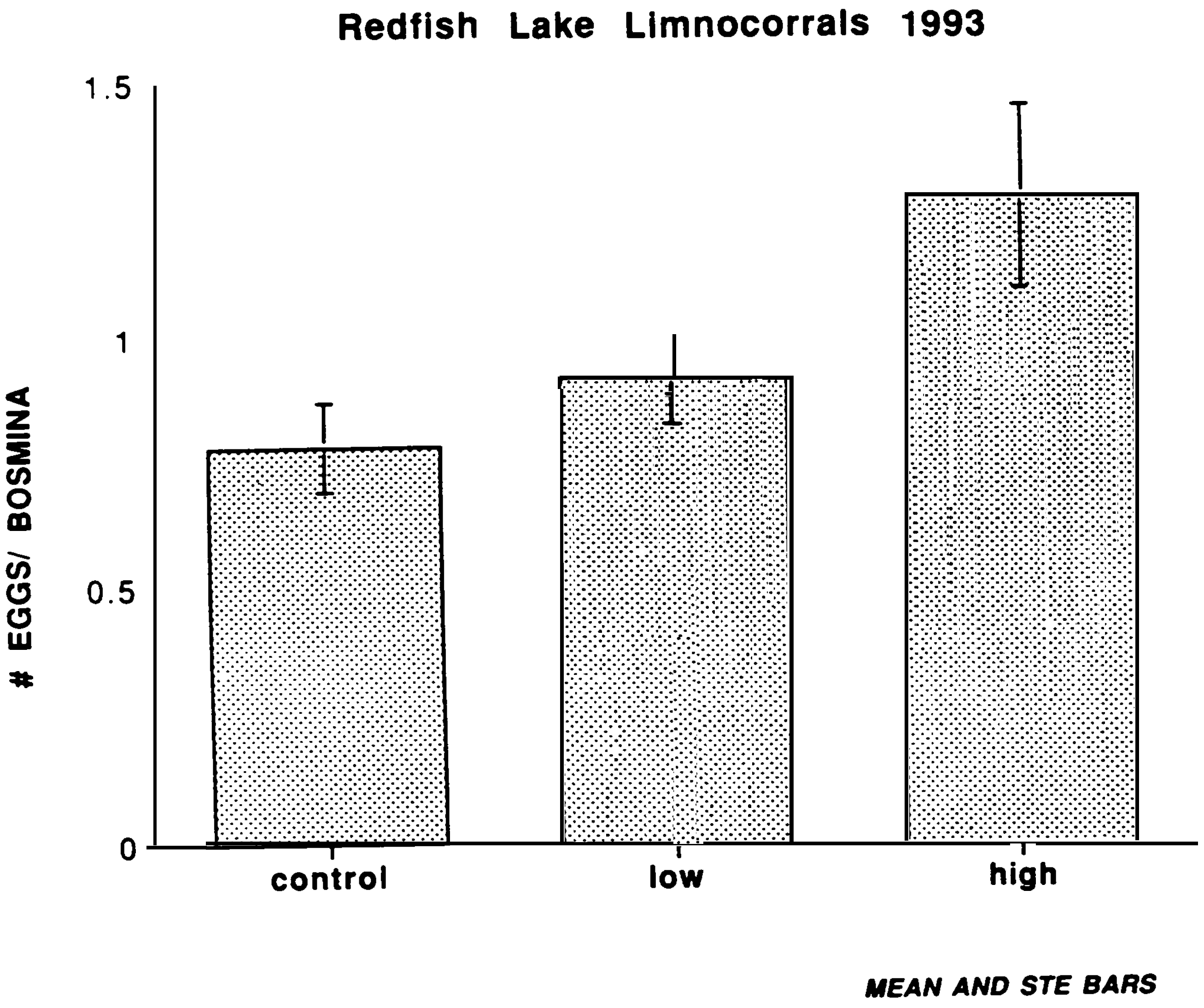

Figure 11. Mean number of eggs per adult *Bosmina* in Redfish Lake limnocorrals collected between 29 June and 11 September 1993 in 17-10 meter tows with standard error bars given.

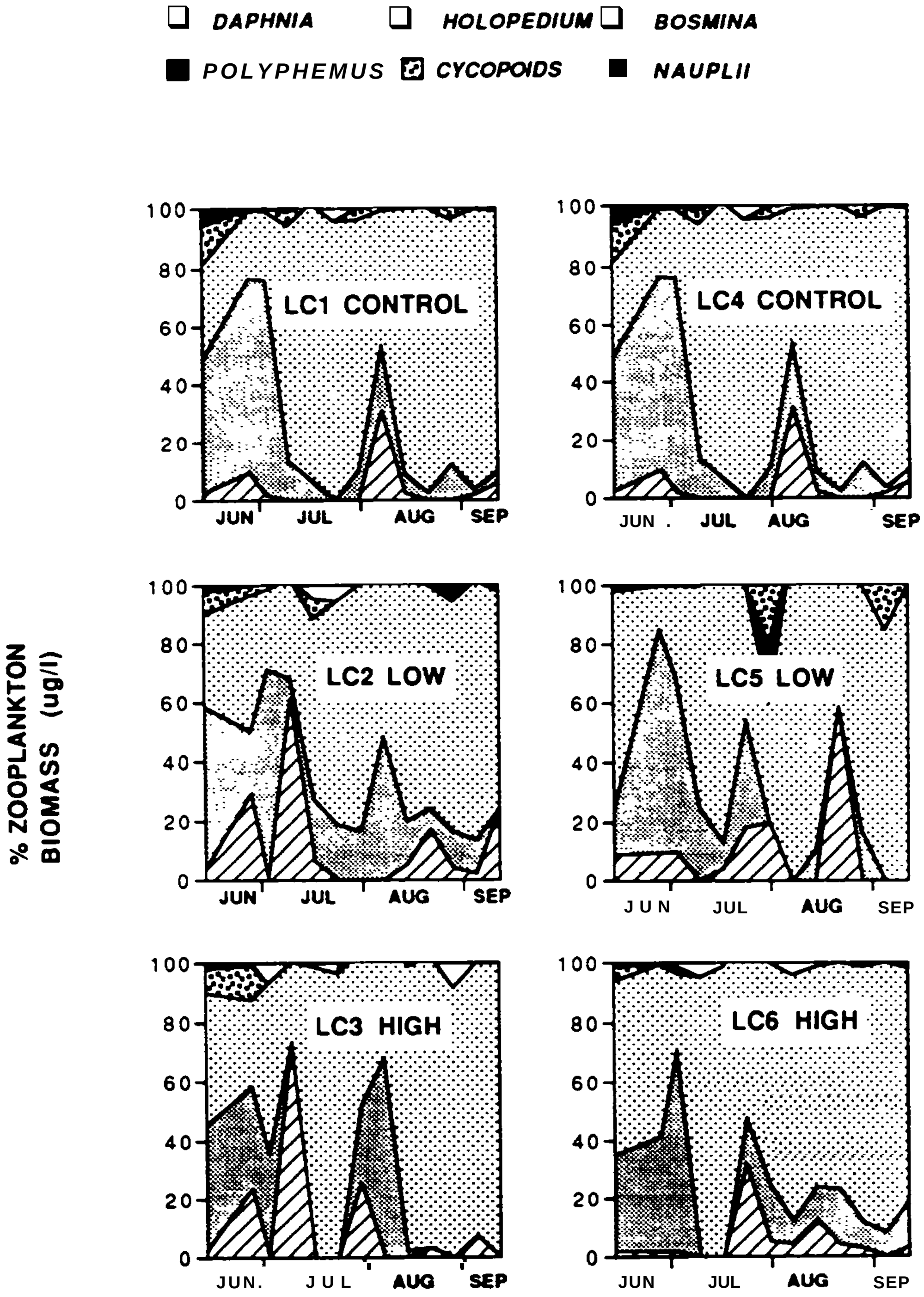

Figure 12. Zooplankton species composition as a percentage of the total zooplankton biomass throughout the experiment for each limnocorral.

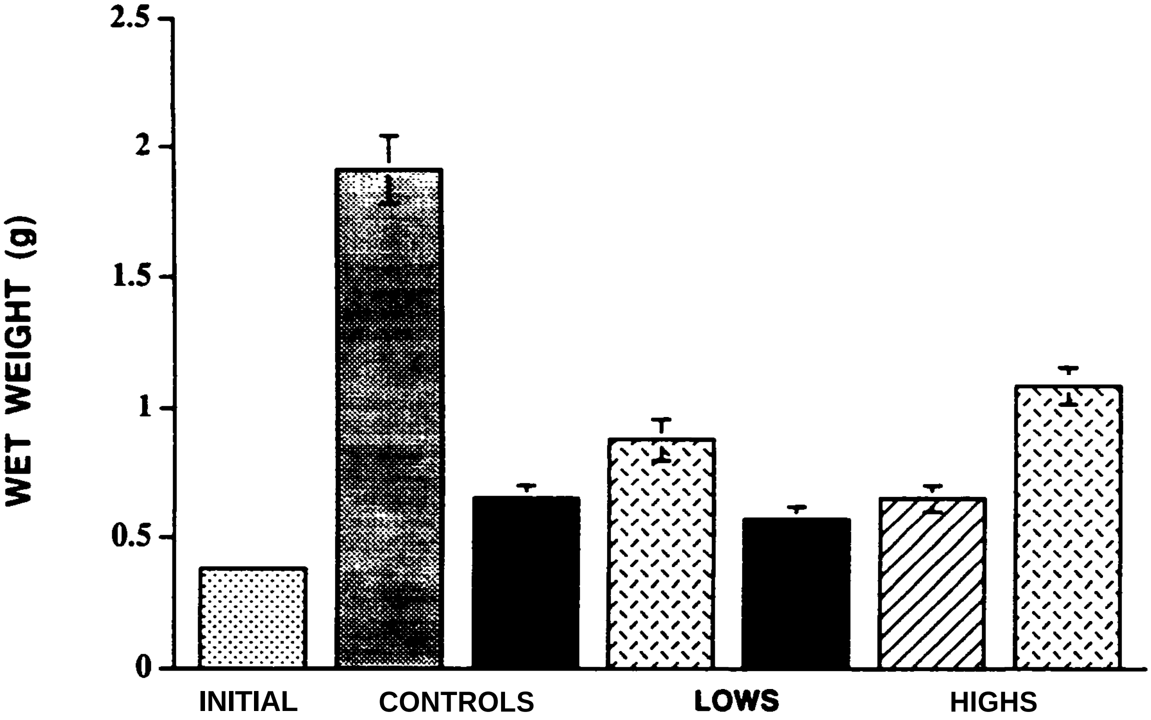

Figure 13. Final fish weight, measured as wet weight, in grams for each limnocorral by treatment with standard error bars given, 13 September 1993.

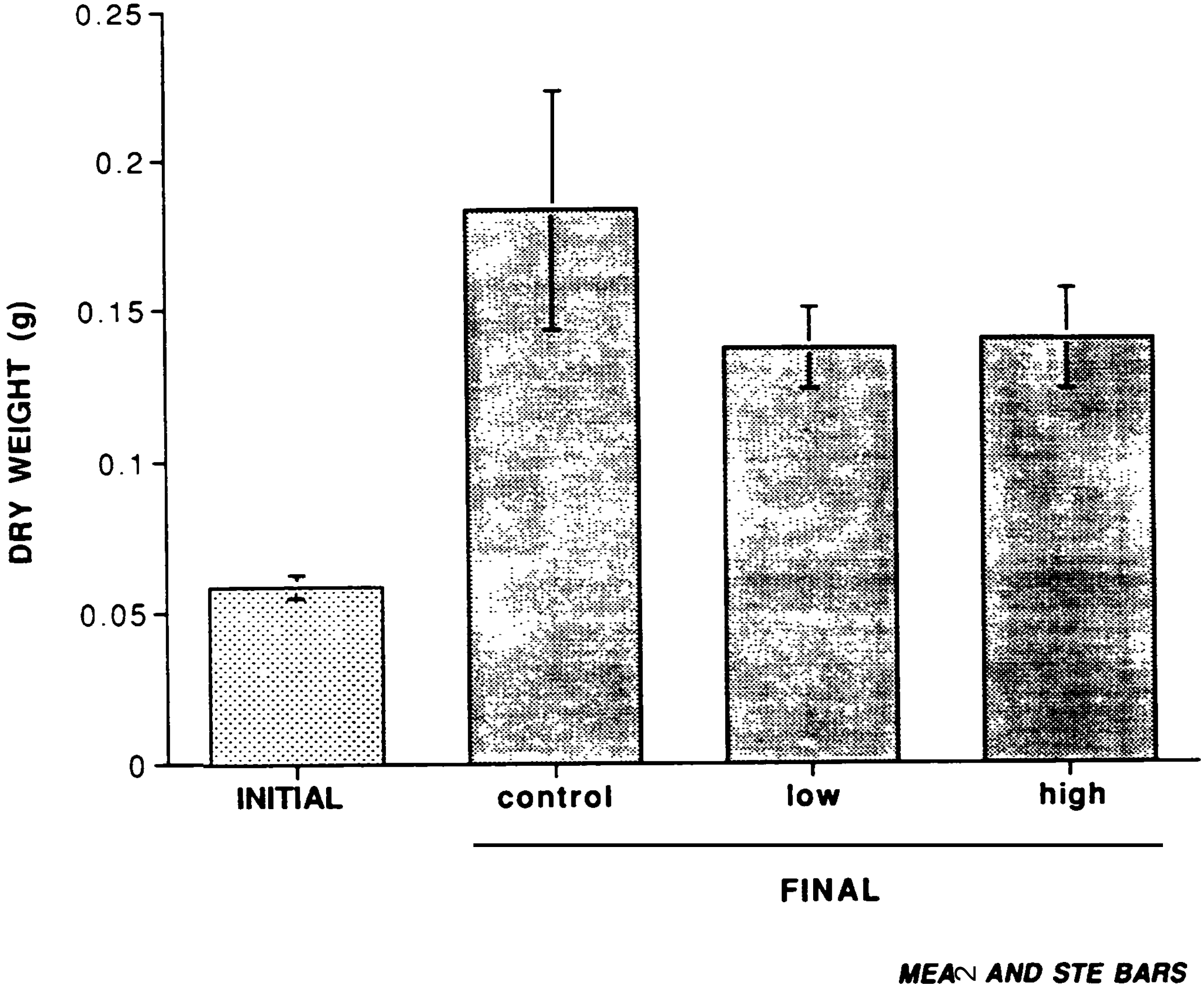

Figure 14. Final dry weight of limnocorral fish compared to initial dry weight (grams). The mean of two replicates and standard error bars are given.

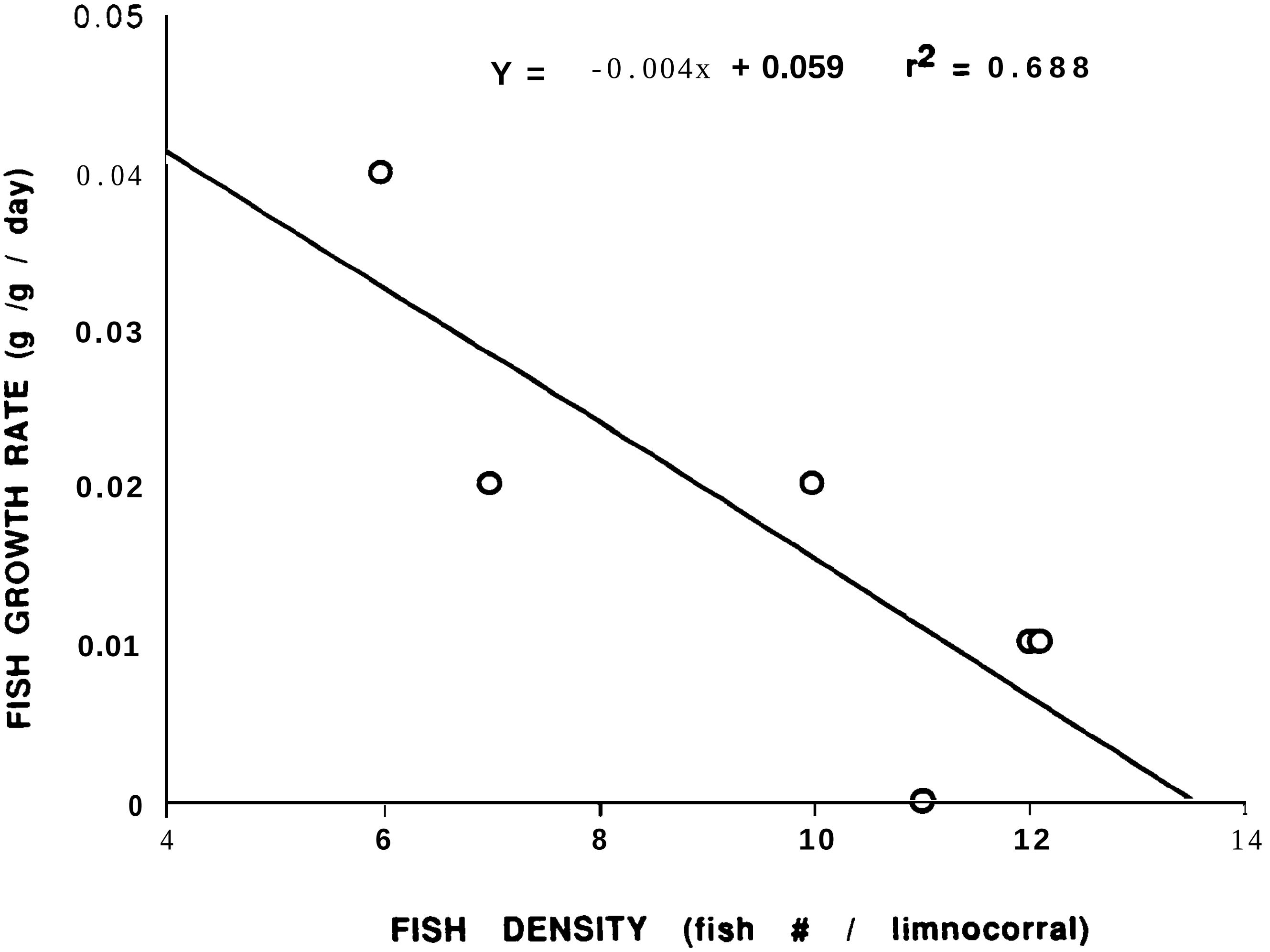

Figure 15. Fish growth in g/g/day as a function of final fish density (#/limnocorral). G/g/day was calculated as the grams of growth per intermediate weight per day.

increased in mass during the experiment, but there was no significant effect of nutrient treatment. Fish growth was, however, inversely correlated with final fish density ($r^2 = 0.688$; $0.10<p<0.05$) (Figure 15). Examination of zooplankton results indicates that the corrals with the lowest fish density had the highest zooplankton biomass during the last month of the experiment (Figure 10).

DISCUSSION

The stimulation of the phytoplankton communities in the Redfish limnocorrals was strongly influenced by the timing of the nutrient additions. Sixty percent of the nutrients were added during the first four weeks of the experiment and the TP and TN concentrations were higher at the 4-week sampling than at the 8-week sampling. This suggests that a large portion of the added nutrients had sedimented out, or had become attached to the walls of the limnocorrals by the second week of sampling. The chlorophyll levels, however, were consistent with the timing of nutrient additions and the measured concentrations, being 3-5 times higher than controls in the fertilized corrals early in the experiment, and then dropping to approximately two-times the control levels. Primary production in July was also enhanced more by the heavy nutrient additions early in the study, than it was by the lower additions in August (Figure 9).

The epilimnetic chlorophyll levels in the low treatments were similar to or exceeded those of the high treatments for the last 7 weeks of the experiment. For the last 2 weeks of the experiment, the O-17 m chlorophyll levels in the low treatments also were similar to or exceeded that of the high treatments. This result may be partially explained by the greater success of periphyton in the high treatments, thus sequestering nutrients from the phytoplankton.

In order to evaluate the taxa-specific results of the nutrient additions to the limnocorrals, we must determine whether the increased phytoplankton biovolume is of the size and quality that will increase zooplankton growth. Caveats to this sort of analysis are mentioned in the discussion of Chapter 4.

The major increases in biovolumes observed in all three treatments were caused by the diatoms *Tabellaria* and *Synedra*, although it is not clear why the bloom of *Tabellaria* was observed in the control limnocorrals on 22 Aug. 1993. As reported in detail in Chapter 4, these species are of intermediate to high food value to *Daphnia* spp., but not used by Bosmina. Cyclopoids use the colonial diatoms at high levels (Schindler 1971, Tóth and Zánkai 1985, Knisely and Geller 1986). The *Anabaena* bloom observed in the epilimnia of the HIGH treatments could be problematic for Daphnia (Hein et al. 1993).

The analysis of zooplankton biomass through time showed no clear patterns of treatment response; however, egg production by adult Bosmina was significantly enhanced by nutrient additions in the metalimnetic (17-10 m) tows. These results indicated that nutrient additions enhanced phytoplankton growth and increased zooplankton egg production, but that the flow of energy to fish was interrupted. Further, because zooplankton exhibited an increase in both total biomass and larger cladocerans when some kokanee were removed midway through the experiment, it appears that the level of fish planktivory was too high to allow zooplankton populations to respond to increased abundance of phytoplankton.

Analysis of fish growth indicated that differential survival of juvenile fish in experimental treatments confounded interpretation of the nutrient effects on growth. Also, high initial fish densities, and the corresponding level of planktivory, may have kept the zooplankton from responding to the increased algae. Fish growth was, however, correlated with the fish density in the different limnocorrals. Kokanee in corrals with low densities of fish remaining at the end of the experiment grew at 2-3 times the rate of kokanee in corrals with higher fish densities. Although kokanee in all limnocorrals increased wet and dry mass over the course of the experiment, the growth of these fish was likely less than kokanee present in the Redfish Lake. The higher densities of crustacean zooplankton available to kokanee in net pens in Redfish Lake grew faster compared to kokanee in the limnocorrals (Chapter 5). The greater abundance of zooplankton in Redfish Lake compared to all of the limnocorrals suggests that kokanee in the lakes would grow at rates faster than those kept in limnocorrals. The inverse relationship between final zooplankton biomass and fish density indicated that fish strongly regulated zooplankton abundance in the limnocorrals, and likely had a greater effect than did the nutrient additions.

In summary, our results suggest that added nutrients could potentially enhance zooplankton biomass by increasing birth rates, but in our experiments variation in planktivory among limnocorrals overwhelmed the effect of nutrient additions. In the Sawtooth Valley Lakes, kokanee salmon would likely compete with juvenile sockeye as they exhibit a high degrees of overlap in diet and habitat use. Our results suggest that a reduction in the abundance of competitors could also be considered as a potential management strategy for increasing growth of stocked sockeye salmon, and could have a stimulating effect on zooplankton similar to or even greater than that expected from a whole lake fertilization.

The low level of nutrient additions to Redfish Lake limnocorrals significantly increased algal biomass and rates of primary production, and reduced water clarity. In part, this was due to the initially large nutrient additions made. Subsequently, when the rate of addition was lowered, transparency in the low treatment was reduced 2-3 m from controls, and remained greater than 8 m. This change would probably not elicit a strong response from the public concerning decreased aesthetic value of the lakes. The high initial rates of nutrient additions would, however, reduce water clarity sufficiently so that the change would likely be obvious to the public and perhaps less acceptable for aesthetic reasons.

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Chapter 4

Comparison of **Epilimnetic** and **Metalimnetic** Fertilizations on the Phytoplankton and Zooplankton of Pettit Lake, Idaho

by

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INTRODUCTION

Our research on the fertility of the Sawtooth Valley Lakes in 1992 showed that these systems were highly oligotrophic, and that algal production was strongly co-limited by nitrogen and phosphorus (Gross et al. 1993). The oligotrophic nature of the lakes may be due, in part, to the loss of sockeye salmon *Oncorhyncus nerka* that once returned, died, and released their marine-derived nutrients into the lakes (Chapter 1). Researchers in other lakes have shown that decreases in lake productivity can result from declines in anadromous salmon populations (Koenings and Burkett 1987). Consequently, the Sawtooth Valley Lakes may now be more oligotrophic than they were under pristine conditions.

Regardless of the magnitude of nutrients brought to the lake by returning salmon, it is likely that lake fertilization would increase overall trophic state of the lakes and increase plankton and fish production, as has been done in many other lakes (Nelson 1958, Hyatt and Stockner 1985; Kyle et al. 1988; Stockner 1987, 1992). Along with increasing primary and secondary productivity, another goal of these fertilizations has been to avoid negatively changing the plankton community structure or the lakes' oligotrophic status (Kyle et al. In press, Jackson et al. 1990).

The Sawtooth Valley Lakes are in a pristine setting and have high water clarity; they are prized natural resources. Lake fertilization could potentially stimulate plantkon, decrease water clarity, and thus a lake's aesthetic qualities. Consequently, a fertilization program to help recover the endangered Snake River sockeye salmon should ideally have a minimal impact on water clarity. Salmon enhancement projects in Alaskan and Canadian coastal lakes using surface-applied nutrients normally stimulate phytoplankton production markedly, but water quality remains characteristic of oligotrophic systems (Shortreed and Stockner 1990).

Another approach that may produce even less effect on surface water quality is to inject nutrients into the metalimnia of the lakes. LeBrasseur et al. (1978) were able to increase primary productivity at 10 m depth with a deep injection of nutrients in Great Central Lake, British Columbia. This is an appealing strategy for the Sawtooth Valley Lakes because it may allow for preservation of water clarity and an increase in sockeye production at the same time. Consequently, in 1993 we used large mesocosms in Pettit Lake to test whether epilimnetic and metalimnetic nutrient additions have different effects on water transparency and primary and secondary production.

METHODS

The nutrient addition experiments took place in six large mesocosms, or limnocorrals, over a ten-week period. Each limnocorral was 5 m in diameter and approximately 18 m deep with tops that floated above the lake surface. They were constructed with weighted curtains of impermeable, fiber-reinforced polyethylene. The bottoms were slowly dropped (12 hours) through the water column with the bottom open so that the initial conditions were similar to the lake. Once filled, SCUBA divers tied the bottoms closed. Each of the six limnocorrals were randomly assigned one of three treatments (n=2):

(1) controls

(2) nitrogen (N) and phosphorus (P) added to the epilimnion (EPI)

(3) N and P injected into the metalimnion (14.5 m) (META).

At the start of the experiment on 3 July, mean total P (TP) and total N (TN) concentrations in the limnocorrals were 4.4 (range of 3.9-5.2) $\mu g/L$ and 84 (range of 67-95) $\mu g/L$, respectively. Over the course of the summer we added 6 and 120 $\mu g/L$, respectively, of phosphorus and nitrogen to the EPI and META treatments. Nutrients were added in the form of $(NH_4)_2HPO_4$ and NH_4NO_3, at a 20:1 mass TN:TP ratio. This high ratio was used to reduce the likelihood of stimulating cyanophytes (Schindler 1977). Weekly, EPI nutrient solutions were stirred in at the surface, while META nutrient solutions were injected at 14.5 m with cooled metalimnetic water and a hose. Fifty percent of the nutrients were added during the first week of the experiment; the remaining 50% was added in equal parts over the following 9 weeks. We would have preferred a more even distribution of nutrient additions starting at an ~~early~~ earlier date than 3 July, but logistical difficulties caused the fertilization to start more than two weeks later than planned. In an attempt to stimulate zooplankton and fish production, we made the early, large addition of nutrients so that these organisms would have sufficient time to respond to increased phytoplankton production.

We measured temperature, oxygen and water transparency biweekly as described in Chapter 4. At weekly intervals we also collected water for chlorophyll a analysis from the epilimnion, and from 0 to 17 m in each limnocorral with depth-integrating Tygon tubes. Biweekly, we collected additional chlorophyll samples from the metalimnion and near the bottom of the limnocorral with a 4-L Van Dorn bottle. Initially, and at 4 and 8 weeks into the experiment, ^{14}C primary productivity measurements were made at 5 depths. Additional depths were sampled for chlorophyll a analysis concurrent with the ^{14}C primary productivity measurements.

On three dates (initially, and at 4 and 8 weeks into the experiment) we saved epilimnetic, metalimnetic, and near-bottom water for nutrient analyses. Samples collected for nutrient analyses were placed in polyethylene bottles which were first rinsed with 0.1 N HCl and then 3 times with aliquots of the actual sample. Nutrient samples were stored in an ice cooler and then frozen upon return to our field laboratory. Total nitrogen was calculated from the sum of Total Kjeldahl Nitrogen (TKN) and nitrate+nitrite nitrogen (NO,-N). Unfiltered water was used for TKN and TP analyses; samples analyzed for nitrate were filtered through a 0.45-µm filter prior to freezing. TP samples underwent a persulfate digestion and were then analyzed colorimetrically in our lab (Utah State University Limnology Laboratory) using the molybdate-absobic acid method. Nitrogen analyses were conducted by University of California at Davis Limnology Laboratory colorimetrically using a Kjeldahl digestion for TKN and the hydrazine method for NO,. Replicates, spikes, and standard solutions were used for quality control/quality assurance.

When nutrient samples were collected we also preserved samples for phytoplankton enumeration with Lugol's iodine solution, and took water for chlorophyll a analysis. Two 50-ml aliquots per sample were filtered through 0.45-μm cellulose acetate membrane filters, extracted into 6 ml of 100% methanol and analyzed fluorometrically using a Turner Model 111 fluorometer (Holm-Hansen and Riemann 1978). Corrections were made for phaopigments. The fluorometer was calibrated using commercial chlorophyll a standards which were verified spectrophotometrically.

A 100-ml aliquot from each phytoplankton sample was filtered through a 0.45 μm cellulose filter. Each filter was cleared and permanently mounted, according to the method of 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 Likens 1991). Phytoplankton were taxonomically classified as follows: Cyanobacteria (blue-green algae), Chlorophyta (green algae), Chrysophyta (*Dinobryon* sp.), Bacillariophyta (Diatoms), and Dinophyta (*Peridinium* sp.).

Primary productivity was measured at five depths in each limnocorral using the ^{14}C method as described in Chapter 3. Duplicate water samples were innoculated with "CHO, and incubated at their respective depths during midday for 4-5 h. To measure non-photosynthetic carbon uptake, DCMU was added to a single sample from each depth and incubated with the other replicates. The samples were filtered, dried and subsequently counted using liquid scintillation techniques. Available dissolved inorganic carbon was estimated by measuring alkalinity (Gran procedure), pH, temperature, and applying factors given in Wetzel and Likens (1991).

To evaluate if phytoplankton in the'limnocorrals were nutrient limited we conducted an *in vitro* bioassay similar to those done in 1992 (Gross et al. 1992). In the seventh week of the experiment (20 August) we collected water from 3 m from one limnocorral in each field treatment (CNTL-Corral #2; EPI-Corral #3; META-Corral #1). In Logan, 750-ml aliquots of water from each field treatment were distrubted into eight, 1-L polycarbonate flasks. Two flasks each received nitrogen, phosphorus, and micronutrients to determine if these were limiting the phytoplankton in the limnocorrals. The flasks were incubated at 15°C with an 18:6 hr photoperiod of 150 μE $m^{-2}s^{-1}$ light intensity. After five days the flasks were sampled for chlorophyll a which was analyzed as described previous. We then added phosphorus to one of the flasks from each N^+ treatment, and nitrogen to one of the P^+ flasks, to determine if N and P were co-limiting phytoplankton growth. After five additional days the flasks were again sampled for chlorophyll. The final experimental design for each of the three field treatments was thus:

Nominal Treatment	**Replic.**	Days of Sampling	Added nutrients
Control	2	5, 10	-No nutrients added
+N	2,1	5, 10	-0.17 M NH_4NO_3
+P	2,1	5, 10	-0.02 M NaH_2PO_4
+M	2	5, 10	-Mix of HEDTA, Fe, Mn, Zn, Cu, Mo, Si, and Bo (Gross et al. 1992)
+NP	2	10	-0.17 M NH_4NO_3; 0.02 M NaH_2PO_4

To quantify periphyton growth in each limnocorral we sampled three times during the season (at approximately 4, 8, and 11 weeks into the experiment). A weighted 10-cm wide strip of polyethylene limnocorral material was suspended from the center of each corral and extended to the bottom. On each sampling date, two 14-mm diameter disks were carefully bored from each periphyton strip with a cork borer at 5-6 depths. Disks were kept on ice until return to our field laboratory, where they were placed directly into 6-ml of 100% methanol for chlorophyll a pigment extraction in the dark for 24-48 hours. We then analyzed the samples for chlorophyll a fluoremetrically as described above.

We sampled zooplankton weekly using a Wisconsin-style, closing, 80-μm mesh net equipped with an anti-reverse General Oceanics flow meter. Two tows were taken in each limnocorral, one from 10 m to the surface, and the other from 17 m to 10 m. Samples were immediately preserved in a formalin-sucrose solution and subsequently counted and measured. In addition, biomass was calculated using length-mass regressions (McCauley 1984; Koenings et al. 1987) and corrected for the volume of water sampled. Egg-ratios of cladocerans were calculated as an estimate zooplankton production (Paloheimo, 1974). Repeated measure ANOVAs were used to analyze zooplankton data for statistical significance accounting for time and initial differences among treatments (Wilkinson 1991).

Because resident 0. *nerka* were unavailable in Pettit Lake, we attempted to use age-0 redside shiners (*Richardsonius*) in the limnocorrals to evaluate effects of nutrient additions on fish growth. The fish used were seined along the lake shores. Fifteen minnows (O-3-0.5 g) were placed in each corral. A sub-sample of 60 fish were sacrificed and used as an initial measure of wet weight. At an intermediate date, 31 July 1993, we attempted to remove some fish using a lift-net. Dry ice was added to each limnocorral at the end of the experiment in order to force fish to the surface where they could more easily be caught.

RESULTS

Water Clarity, Temperature and Oxygen

Fertilization significantly decreased water transparency in both the epilimnetic and metalimnetic treatments ($P<0.002$), but the reduction was not large. On most dates, transparencies were greater in the META treatments than in the EPI treatments. Water transparency remained above 8 m in all of the treatments throughout the experiment (Fig. 1).

Temperature and oxygen profiles closely paralleled the conditions observed in the lake, and there was little difference among treatments on each date (Fig. 2). Surface temperatures ranged from 11.0°C at the beginning of the experiment in July to 16.0°C at the end of the experiment in September. The temperature near the bottom of the limnocorrals (-17 m) ranged from 5.5°C in July to 6.0°C in September.

Dissolved oxygen conditions were suitable for fish growth (>5 mg/l) throughout the limnocorral for the duration of the experiment (Brett et al. 1969). Both the temperature and oxygen profiles showed no clear response to nutrient additions.

Nutrients

Both the EPI and META nutrient additions increased TN and TP concentrations over initial and control concentrations (Fig. 3). The nutrient profiles indicated, however, that there was considerable movement of the added nutrients between strata. The highest nutrient concentrations in the META treatments were at 17 m, but epilimnetic nutrient levels also increased in this treatment (Figs. 3c-f). In the EPI treatments, the highest nutrient concentrations were in the epilimnion on 31 July (Figs. 3c-d). However, on 29 August, they were highest at 17 m (Figs. 3e-f).

The nutrient sampling results indicate that considerable portions of the added nutrient also sedimented to the bottom of the corrals. The was especially evident on 29 August in the EPI treatments, where TN and TP concentrations were highest at 17 m (Fig. 3e-f). Part of this is probably due to a boundary condition imposed by the limnocorrals, allowing the accumulation of sinking phytoplankton, detritus, and other particulates near the bottoms of the mesocosms.

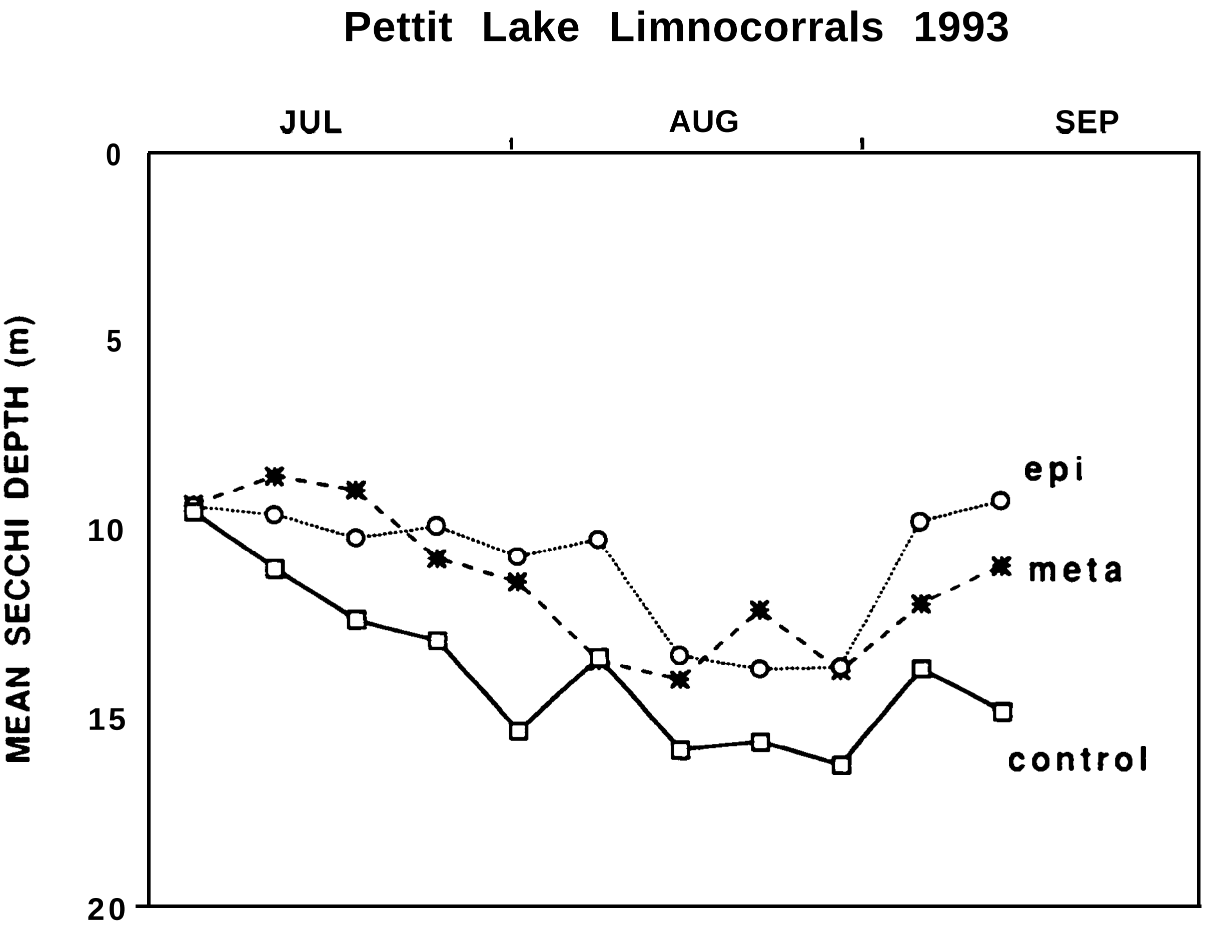

Figure 1. Secchi depths as the mean of two replicates from the start of the experiment in July until the end of the experiment in **September.**

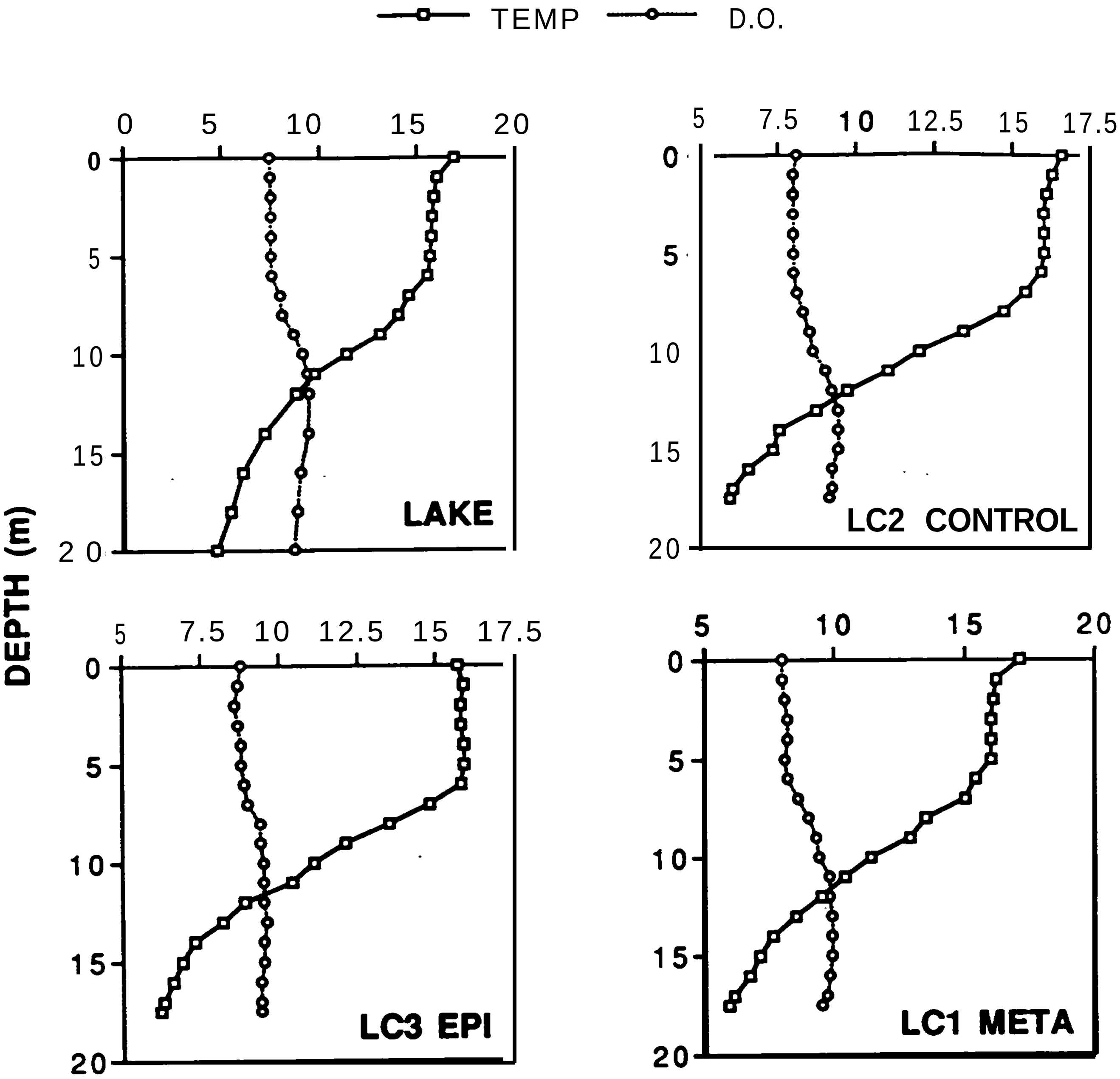

Figure 2. Temperature and oxygen profiles for Pottit Lake on 18 August 1993 and in one of each replicated treatment in the Pottit Lake limnocorrals on 4 August 1993.

Pettit Limnocorral Nutrient Levels, 1993

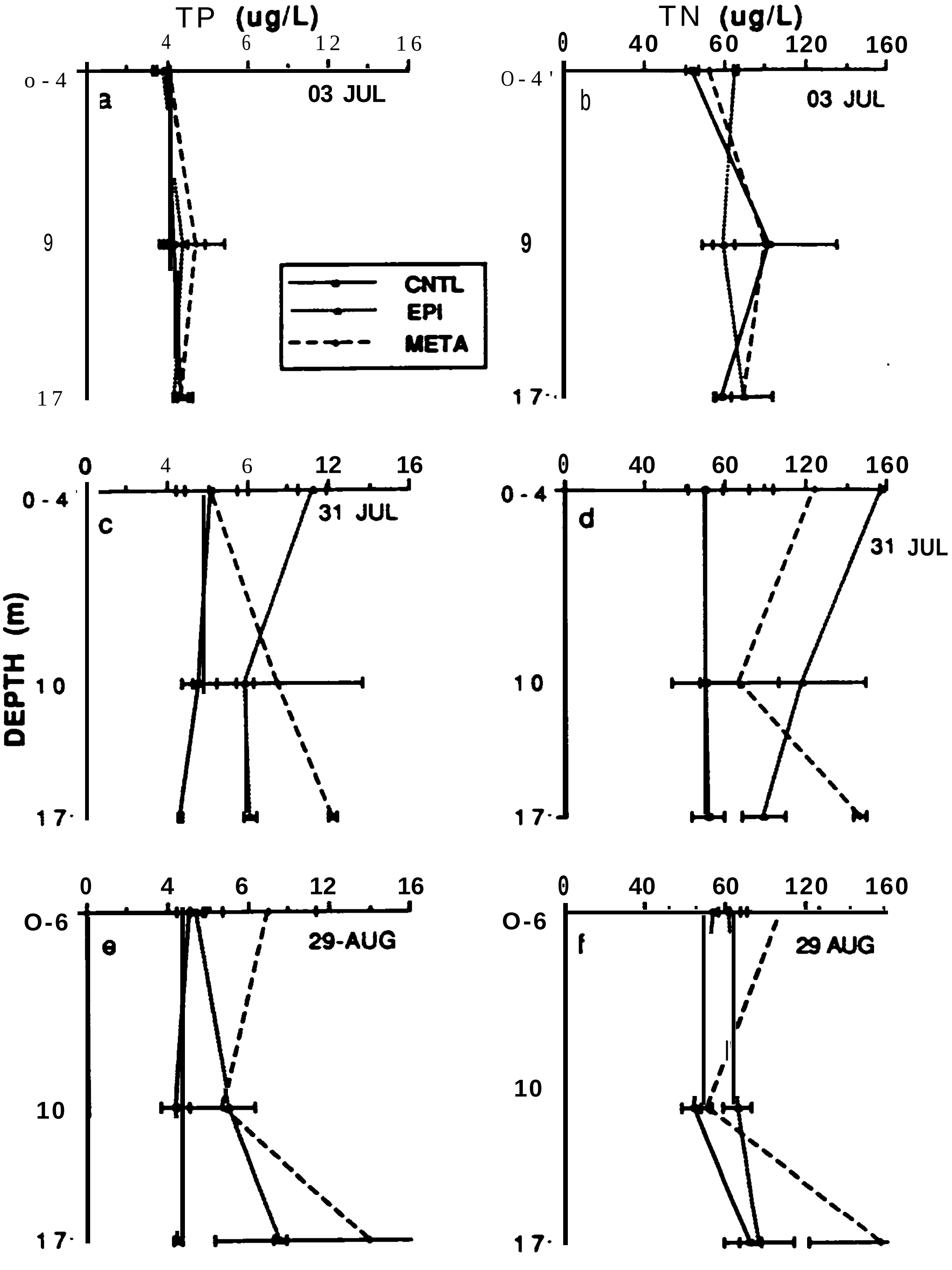

Figure 3 . **Depth profiles** of total phosphorus **(TP) and** total **nitrogen (TN)** in **Pettit Lake limnocorrals,** 1993. Control, **EPI, and META treatments shown; error bars** show **range (n=2). Epilimnetic samples** (0-4 **&** 0-6 **m) were** collected with 8 **Tygon** tub.; **metalimnetic** (9 **&** 10 **m) and** 17 **m samples were collected** with **a** Van Dorn **bottle.**

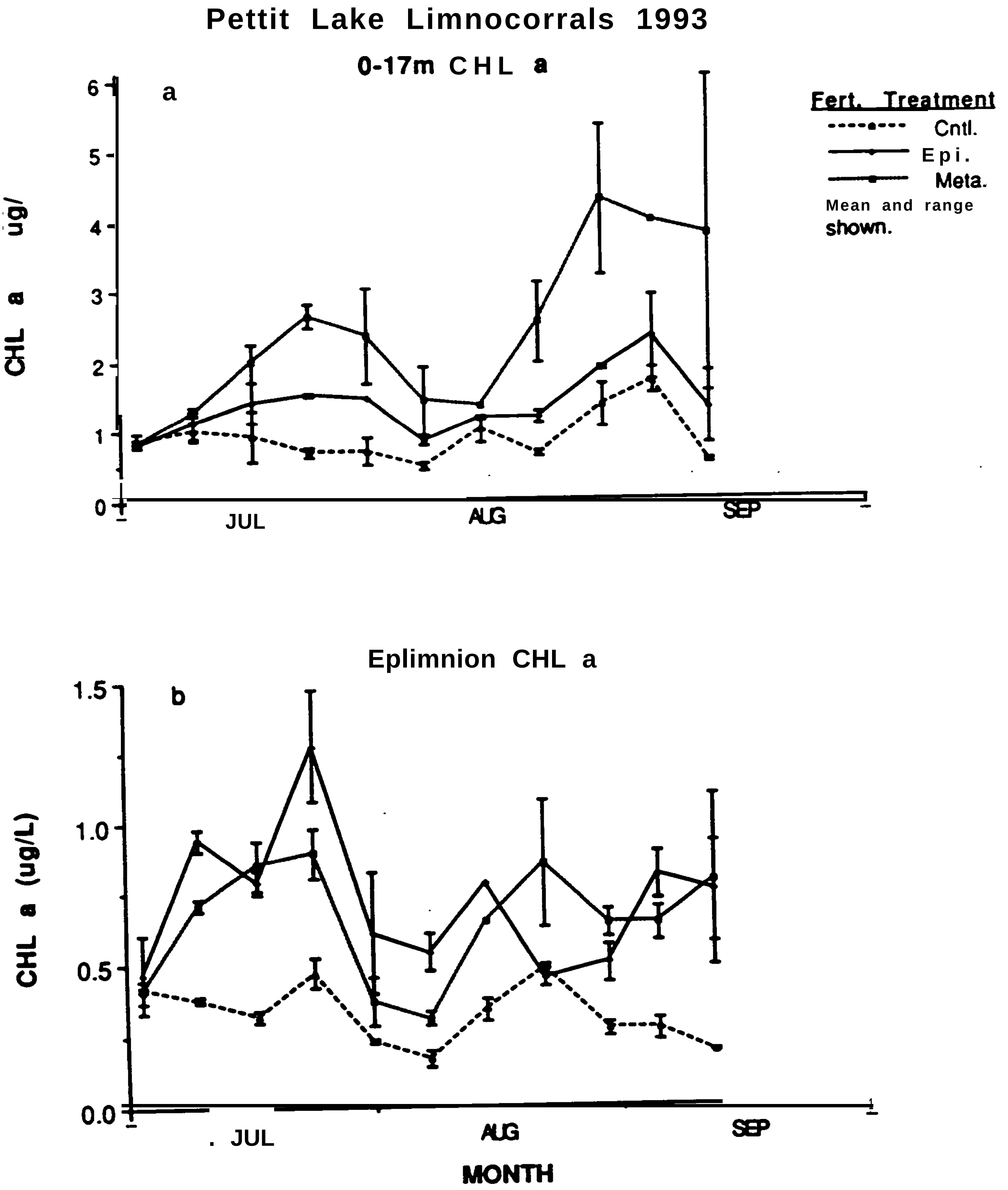

Figure 4. Mean chlorophyll levels (µg/L) from (a) 0-17 m and (b) epilimnetic tube samples collected from Pettit Lake limnocorrals, 1993. Control, EPI, and META treatments shown; error bars show range (n=2). Nutrient additions began on July 3, 1993.

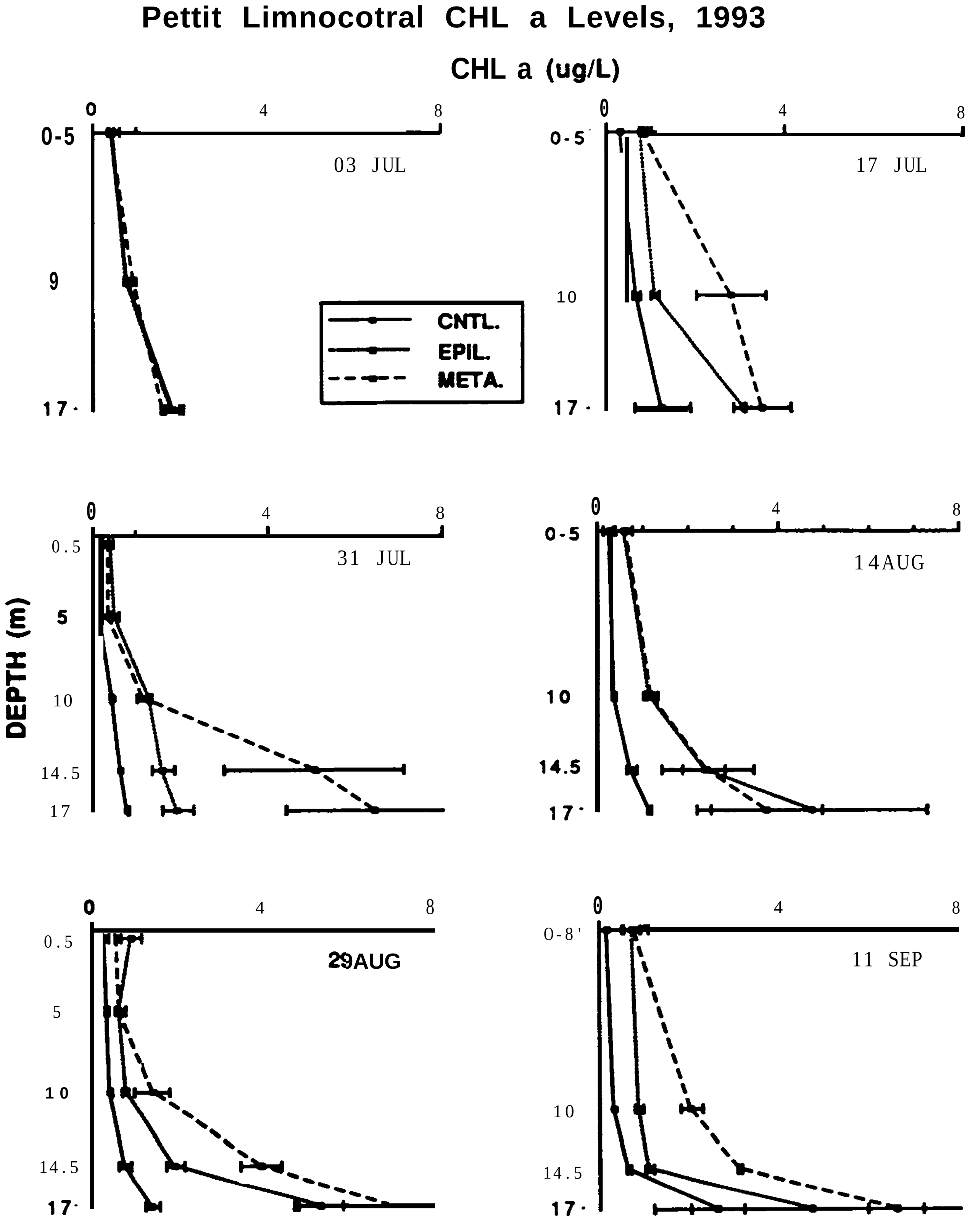

Figure 5. Depth profiles of mean chlorophyll levels (j&g/L) in Pettit Lake limnocorrals, 1993. Control, EPI, and META treatments shown; error bars show range (n=2). Epilimnetic samples on 03 JUL, 17 JUL, 14 AUG, and 11 SEP were collected with a depth-integrating Tygon tube; all other the samples were collected with a Van Dorn bottle.

Chlorophyll, Primary Production, & Algal Biomass

Mean chlorophyll levels in the EPI and META treatments were greater or equal to controls throughout the experiment (Fig. 4 and 5). The weekly mean epilimnetic chlorophyll levels were 0.31, 0.74, and 0.67 μg/L for the control, EPI, and META treatments, respectively. These values ranged from 0.17-0.48 μg/L for the controls, 0.45-1.27 μg/L for the EPI treatments, and 0.31-0.89 μg/L for the META treatments (Fig. 4b). The mean epilimnetic chlorophyll values for the EPI treatment exceeded that of the META treatment on 6 of the 10 weekly samplings after fertilization began.

The mean chlorophyll concentrations in the weekly O-17 m tube samples were 0.91, 1.42, and 2.56 μg/L for the control, EPI, and META treatments, respectively (Fig. 4a). The O-17 m chlorophyll concentrations for the META treatments were greater than those in the EPI treatments on all 10 weekly samplings after nutrient additions began.

Epilimentic chlorophyll levels in the control treatments followed a trend similar to concentrations in the lake, suggesting that for this parameter the limnocorrals reasonably mimicked natural conditions (Fig. 6). Nevertheless, lake chlorophyll concentrations were slightly higher than levels in the controls during the last 5 weeks of the experiment. Note that mean epilimnetic chlorophyll concentrations in the nutrient treatments never exceeded levels recorded in Pettit Lake during spring overturn (cf. Fig. 4a and 6).

The phytoplankton community structure was similar in all 3 treatments at the start of the experiment (Fig. 4, see 03 July 1993), with a mixture of chlorophyta, diatoms, and *Dinobryon* at all depths. The dominant chlorophyta were Oocystis, *Chlorella*, and small *Chlorococcales* spp., while the dominant diatom was *Cyclotella*. No Dinophyta were found in the corrals during the experiment, although some were present in the lake (Chap. 2, Fig. 9).

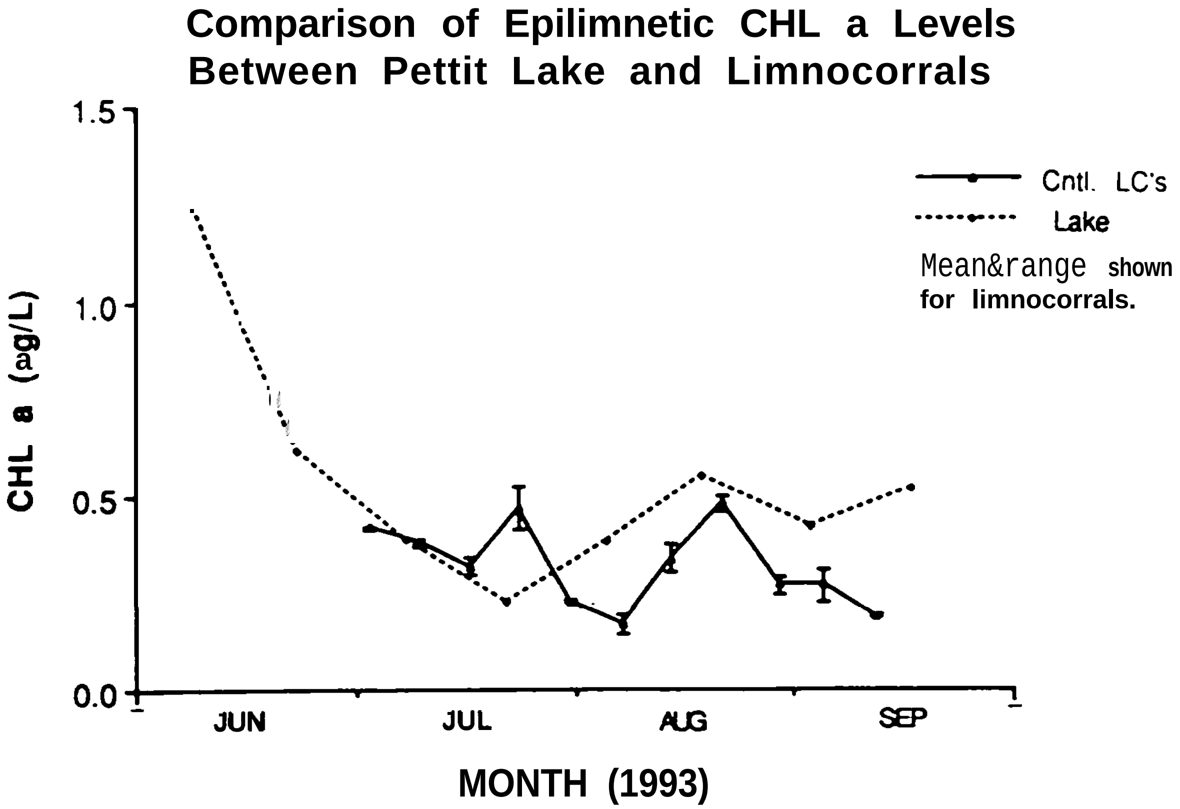

Figure 6. Comparison of lake and control limnocorral epilimnetic chlorophyll a levels in Pettit Lake, 1993. Error bars show range for limnocorrals (n=2).

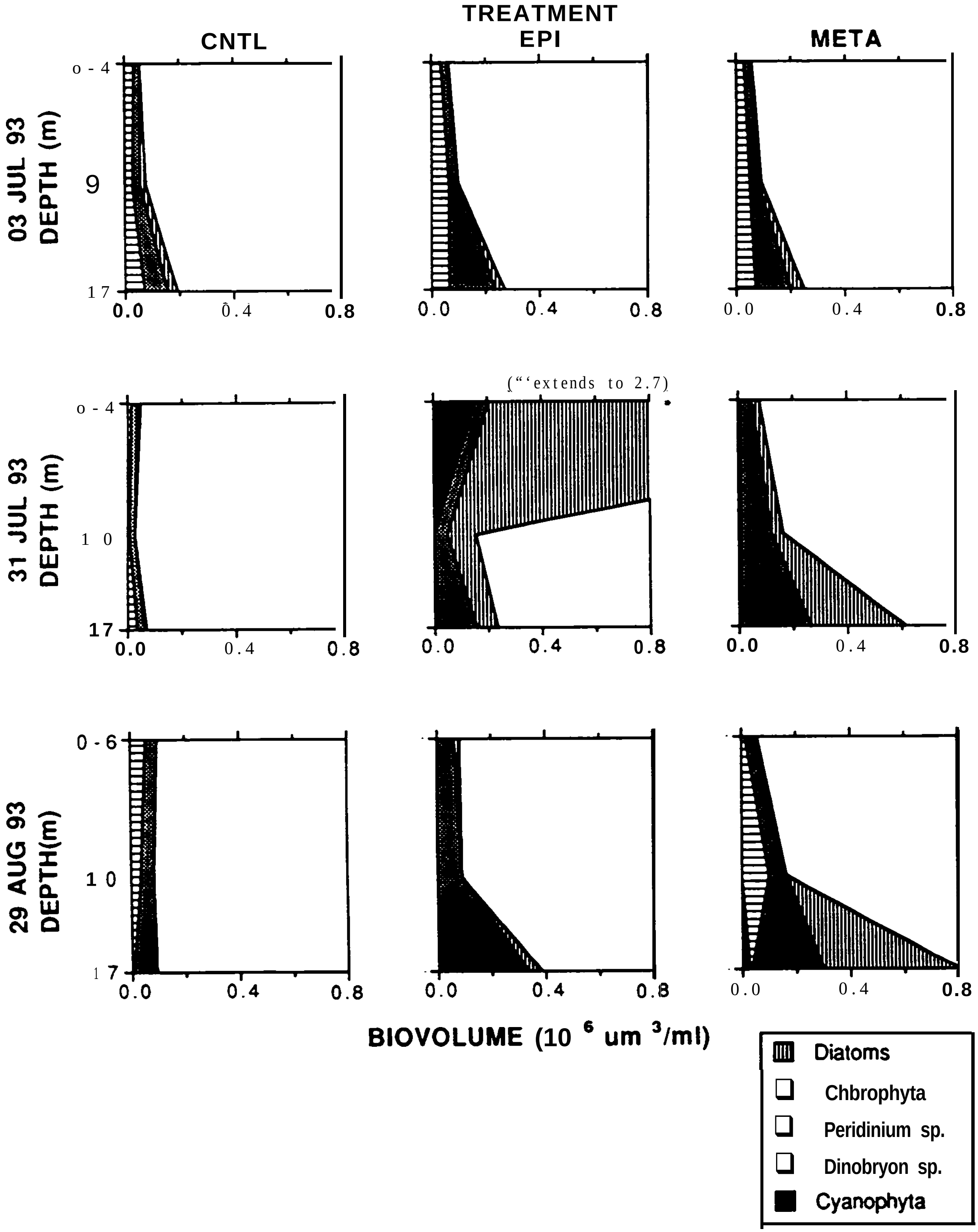

Figure 7. Mean algal biovolumes for Pettit Lake limnocorrals at epilimnetic, metalimnetic, and near-bottom depths, 1993. Graphs from same date shown horizontally; graphs from same treatment shown vertically.

After 4 weeks (Fig. 4, see 31 July 1993), phytoplankton biovolumes had increased most at the depths of the nutrient additions in each treatment. The dominant increase was in the diatoms, primarily due to blooms of *Fragilaria.* The subdominant *Tabellaria, Asterionella,* and *Synedra* spp. also increased. The *Cyclotella* and *Dinobryon* spp. declined in all three treatments. The phytoplankton community structure in the control treatments was similar to that of 03 July 1993, but the biovolume declined by 15-60%, with the largest decrease at 17 m. In the EPI treatments, biovolumes increased from 03 July 1993 in the epilimnia (an enormous 4300%) and at 10 m (52%), but decreased slightly (14%) at 17 m. The cyanophyte *Oscillatoria* also increased in the epilimnia, but only made up 6% of the total biovolume. Epilimnetic increases in the Chlorophyta were due to *Spondylosium* sp. In the META treatments, increases in mean algal biovolumes over 03 July 1993 were greatest at depth (37% in the epilimnion, 76% and at 10 m, and 150% at 17 m). Chlorophyta increases of -100% in the META treatments were primarily from *Oocystis.*

After 8 weeks (Fig. 4, see 29 Aug. 1993), the Fragilaria bloom from 31 July 1993 was virtually gone from the EPI treatments, but had increased by 45% in the META treatments. The biovolumes in the control treatments increased by between 30-160% from 31 July 1993, attaining levels comparable to the EPI and META treatments in the epilimnia and at 10 m. In the EPI treatments, increases in biovolume of the Chlorophytes of up to 130% were primarily due to *Spondylosium* and Gloeocystis. The chlorophyte increase at 17 m in the META treatments were primarily due to *Chlorella,* although some *Spondylosium* were present.

Nutrient additions markedly increased primary productivity in the limnocorrals, and the stimulation was greatest in the strata where the nutrients were added (Fig. 8). Primary production was first measured on 5 July, one day after nutrients were first added. On this date productivity was generally similar among the treatments, but algal growth at 0.5 m in the EPI treatment was nearly 50% above control treatments, suggesting that the

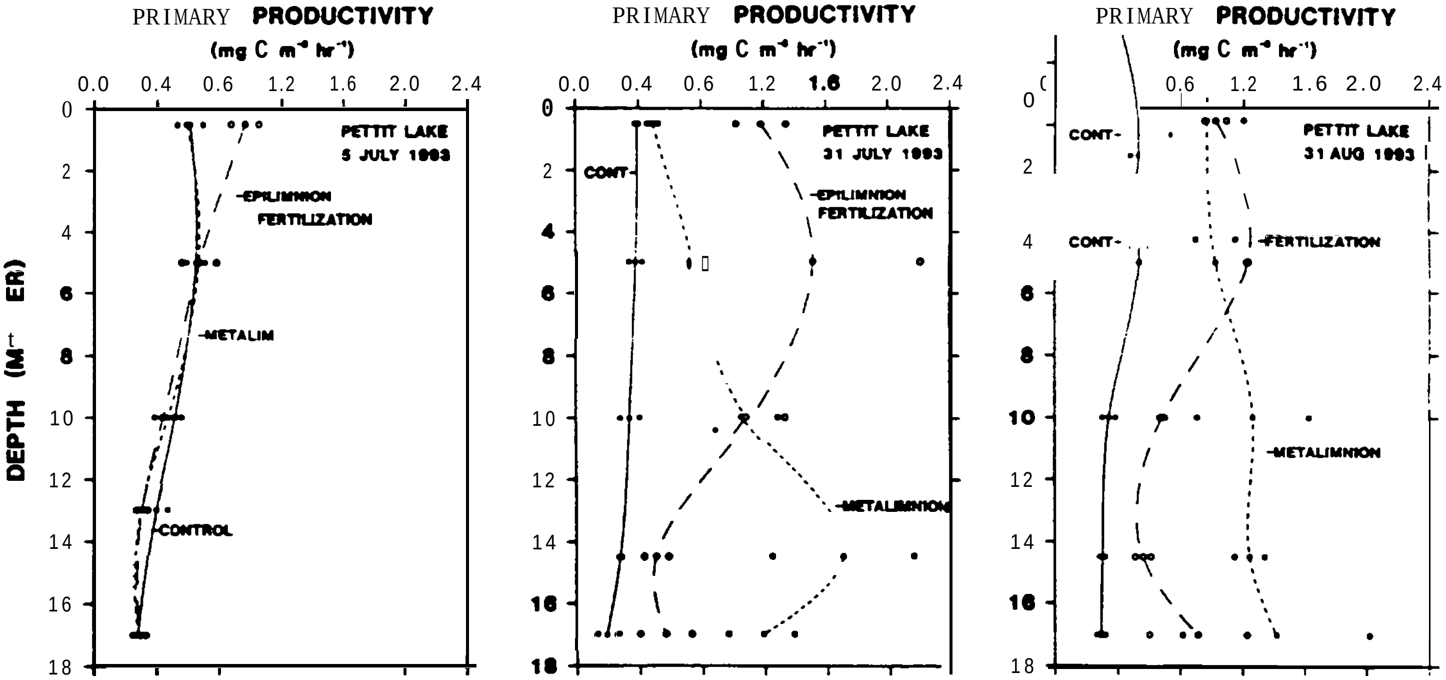

Figure 8. **Depth profiles** of primary production measured on **three dates** in **the Pettit Lake limnocorrals**. Individual **profiles** arm shown for **the** controls **(CONT)**, ● pilimnmtic **fertilizations**, and **fertilizations** in thr **metalimnion (METALIM)**. Data point8 **show the** replicates and means of **measurements** at **each** depth.

phytoplankton responded quickly to the nutrient addition in the surface where light levels were high.

By the end of July the spatial differences in productivity among the three treatments were well established (Fig. 8, center). Productivity in the control treatments remained low throughout the water column. In the epilimnetic fertilization, production in the epilimnion was 3-4 times greater than in the controls. This enhanced productivity extended to 10 m, but declined in the deeper water. In the metalimnetic fertilization production was stimulated five times above control levels in the deeper water, whereas in the epilimnia the stimulation was less pronounced. By the end of August, the spatial differences in productivity among the treatments were less distinct, but the pattern established in July was still evident (Fig. 8, right).

Fig. 9 summarizes how the different fertilization strategies effected primary production in the different strata of the limnocorrals. In the control treatments, integrated productivity in the epilimnetic (O-7.5 m) and metalimnetic (7.5-17 m) strata were approximately equal, and overall production was relatively low. On 5 July total integrated productivity was similar in all three treatments, indicating that there had been insufficient time for significant response to the fertilizations. However, in late July and August, fertilization of the epilimnia markedly stimulated primary production in that strata, but also increased algal growth in the deeper water. The metalimnetic fertilization had the reverse effect, primarily stimulating production in the metalimnia of the limnocorrals. Overall rates of primary production were approximately equal in the epilimnetic and metalimnetic fertilization treatments, and both were approximately 3-times higher than in the controls.

Primary production was significantly correlated with chlorophyll concentrations in the limnocorrals (Fig. 9b), indicating that the more frequently measured chlorophyll provides a reasonable index of algal productivity. Phytoplankton assimilation rates (carbon fixed/mg chlorophyll a), however, were

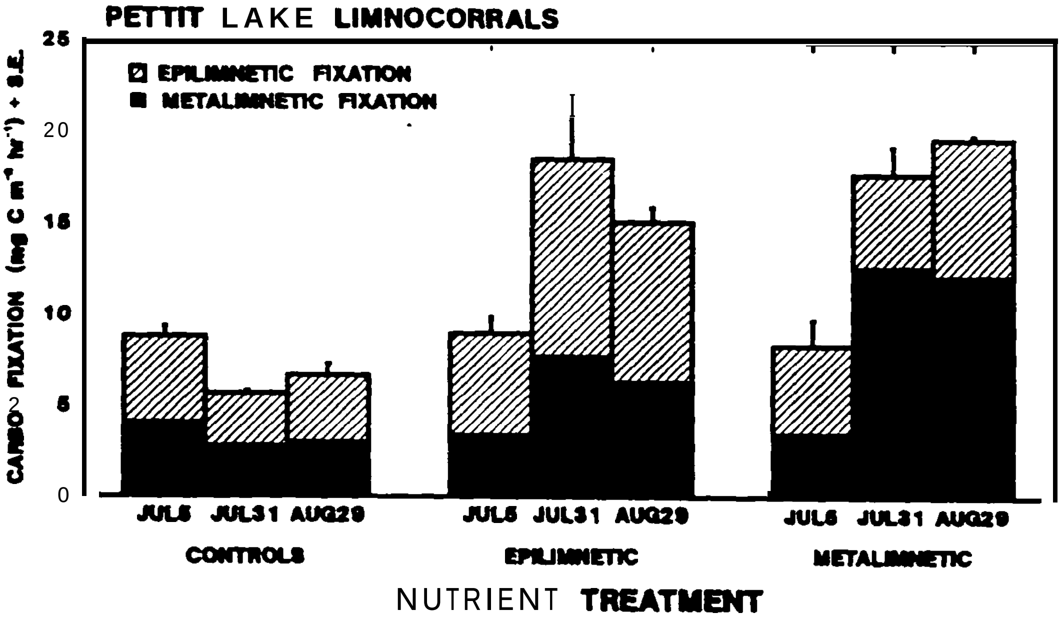

Figure 9. Summary of primary production rates in the control treatments, epilimnetic fortiliratioas, and metalimnetic fortilisatioa treatments of the Pettit Lake limnocorrals oa three dates. The measurements oa 5 July were done only one day ● ft8r fertilization was initiated, and consequently th8 results for this data indicate the relative similarity of tha control and fertilization treatments prior to manipulation. The dark bars indicate primary production in th8 metalimnia (7.5-17 m) of th8 limnocorrals, and th8 hatched sections indicate fixation in th8 epilimnia. Th8 total height of 8ach bar indicates total water column productivity for that treatment (Mean + S.E.).

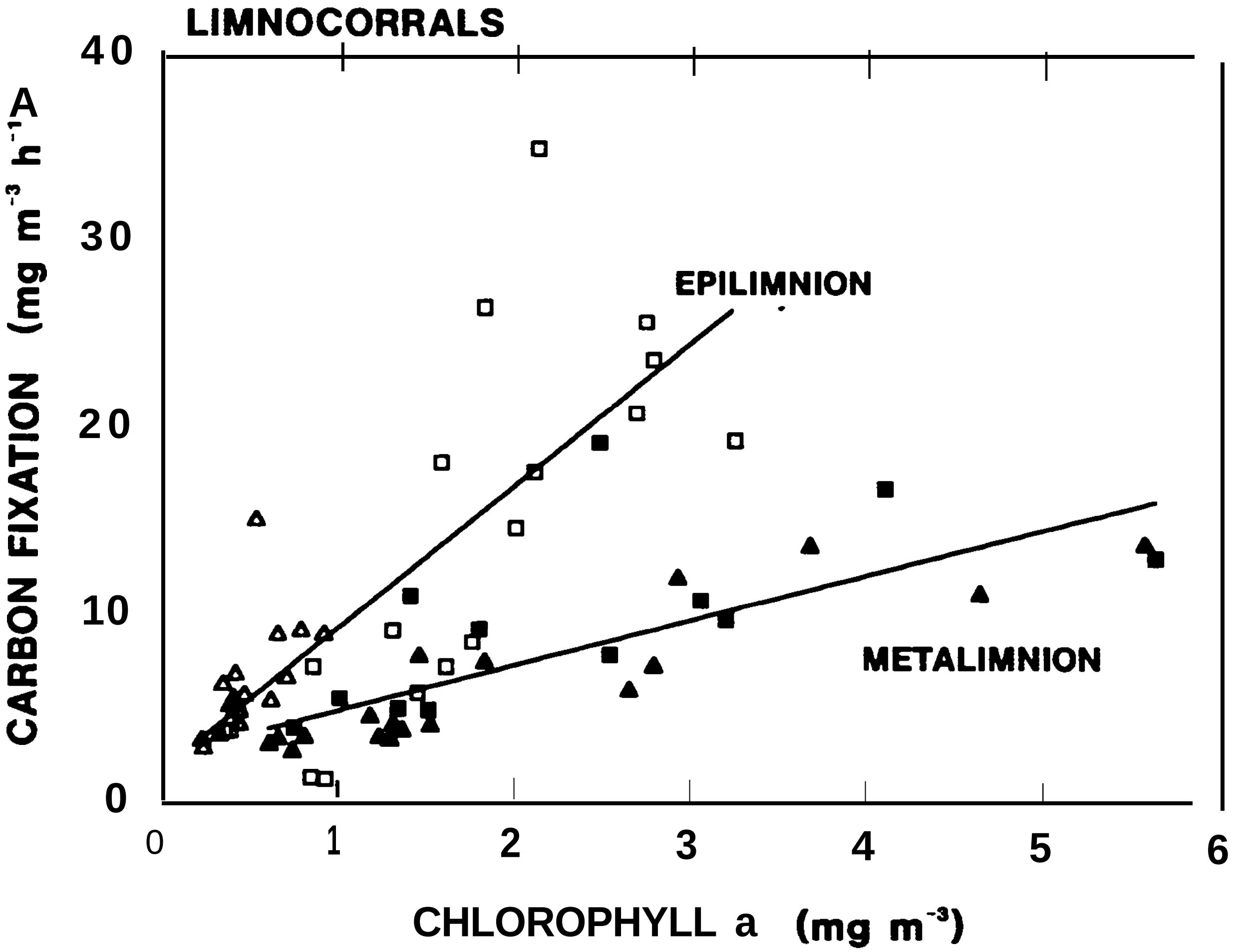

Figure 9b. Relationship between chlorophyll a concentrations and photosynthetic rates (PPR) in th8 epilimnia and metalimnia of the limnocorrals. Triangles and squares indicate data from Pettit Lake and Redfish Lakes, respectively. Filled and open symbols are for data from the metalimnia and epilimnia, respectively. Least squares regressions: Epilimnion - PPR = 1.64 + 7.58 (Chl a); $r^2 = 0.64$, $p < 0.001$. Metalimnion - PPR = 2.32 + 2.41 (Chl a); $r^2 = 0.58$, $p < 0.001$).

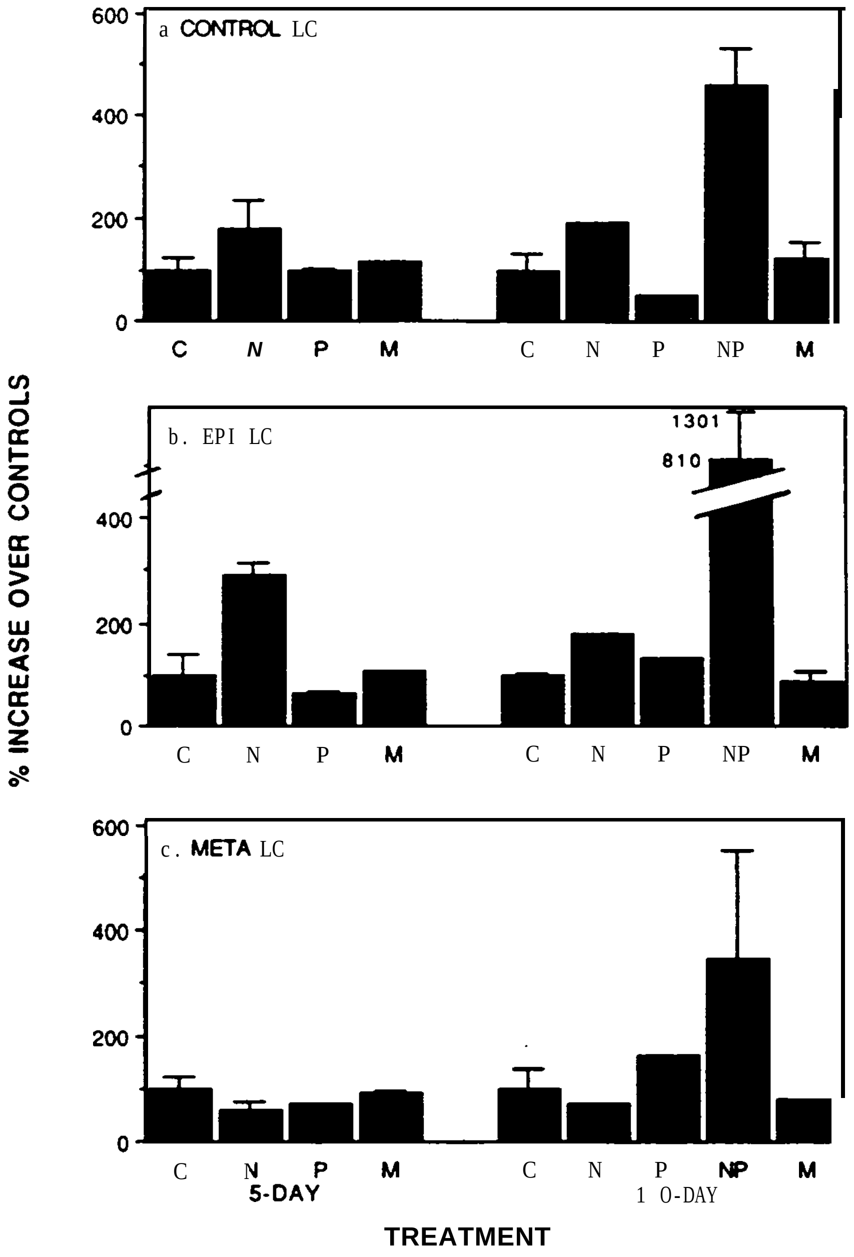

Figure 9c. Effects of nutrient additions on chlorophyll production in the Pettit Lake in vitro experiment. Water from the control limnocorrals (CNTL), epilimnetic nutrient addition limnocorrals (EPI), and metalimnetic nutrient addition corrals (META) was used in the laboratory experiment. Water from each of these field treatments was then amended with nitrogen (N), phosphorus (P), nitrogen plus phosphorus (N+P), micronutrients (M), or maintained as a control (C). The plankton were analyzed for chlorophyll a after 5 or 10 d. Two replicates were used for each treatment except for +N and +P at after 10 days, when only one replicate was available (see methods). Means ± S.E.

2-3 times higher in the epilimnia than in the metalimnia of the limnocorrals. This was not unexpected, as phytoplankton growing in light-limited conditions often synthesize additional pigments to maximize light capture (Reynolds 1984, p. 130)

The *in vitro* bioassay conducted in August indicated that the phytoplankton in the limnocorrals were limited by nitrogen and/or phosphorus (Fig. 9c). Phytoplankton taken from the CNTL limnocorral were stimulated by nitrogen additions after 5 and 10 days, and particularly by N + P additions after 10 days (the N+P treatment was not sampled at 5 days). This indicates that the plankton were colimited by N and P. Adding phosphorus or a micronutrient mixture alone did not stimulate chlorophyll production from this limnocorral. Phytoplankton from the EPI limnocorral were also stimulated by N and particularly by the N+P additions, but not by P or micronutrients alone. Plankton from the META limnocorral did not respond to any nutrients after 5 days, suggesting that they were not nutrient limited when they were sampled. After 10 days, however, P and N+P additions both stimulated chlorophyll production above control levels.

Periphyton Growth

Periphyton biomass, as indicated by chlorophyll levels, inside the limnocorrals varied by as much as three orders of magnitude over the course of the experiment (Fig. 10a-c), with density always lowest in the control treatments. Algae was barely detectable by eye in the control treatments, but in the epilimnetic fertilization treatments 5-20 mm thick patches of periphyton developed on the limnocorral walls. Periphyton developed first in the epilimnion of the EPI treatments (Fig. 10a) and then spread deeper into the water column as the experiment progressed (Fig. 10b). By the end of the experiment, however, EPI periphyton levels started to decline (Fig. 10c). Overall, epilimnetic periphyton chlorophyll was always greatest in the EPI treatments (Figs. 10a-c).

In the metalimnetic fertilization treatments periphyton was stimulated throughout the water column, but particularly in the lower depths. Periphyton levels increased as the experiment progressed, eventually exceeding EPI levels at 13 and 17 m on 14 Sept. 1993 (Figs. 10a-c).

To compare the relative amounts of algal biomass in the phytoplankton and in the periphyton in the different treatments we calculated the total mass of chlorophyll in each of these communities in each limnocorral (Fig. 10, lower). Throughout the experiment the amount of periphyton chlorophyll in the control treatments was negligible. In the EPI treatments, however, between 20 and 50% of the chlorophyll in limnocorrals was in the periphyton community. In the metalimnetic fertilization treatments, periphyton developed slowly, and it never represented more than 20% of the chlorophyll in these limnocorrals. Note that the mean of the combined periphyton and phytoplankton chlorophyll levels for the META treatments were greater than that of the EPI treatments on all 3 sampling dates (Figs. 10d-f).

Zooplankton

Zooplankton biomass in the limnocorrals was initially low, peaked in late July or early August, and declined during the last part of the experiment (Fig. 11). Zooplankton biomass was not, however, significantly related to nutrient treatments (ANOVA; $P=0.704$). The initial biomass of zooplankton in individual limnocorrals varied considerably, ranging from <10 $\mu g\ L^{-1}$ to >80 $\mu g\ L^{-1}$ (Fig. 12). This wide range in starting conditions undoubtedly made treatment effects difficult to distinguish, especially near the beginning of the experiment.

Nutrient additions significantly increased the number of eggs per adult *Daphnia* in the top 10 m ($P= 0.036$) of the limnocorrals (Fig. 13), but not in the bottom 7 m ($P=0.843$). The highest egg production was observed in the META treatments followed by the EPI treatments, with the lowest production observed in the control treatments.

The zooplankton species biomass composition in each corral, as a percent of the total biomass on each sample date are shown in

Perlphyton Levels in Pettit Limnocorrals, 1993

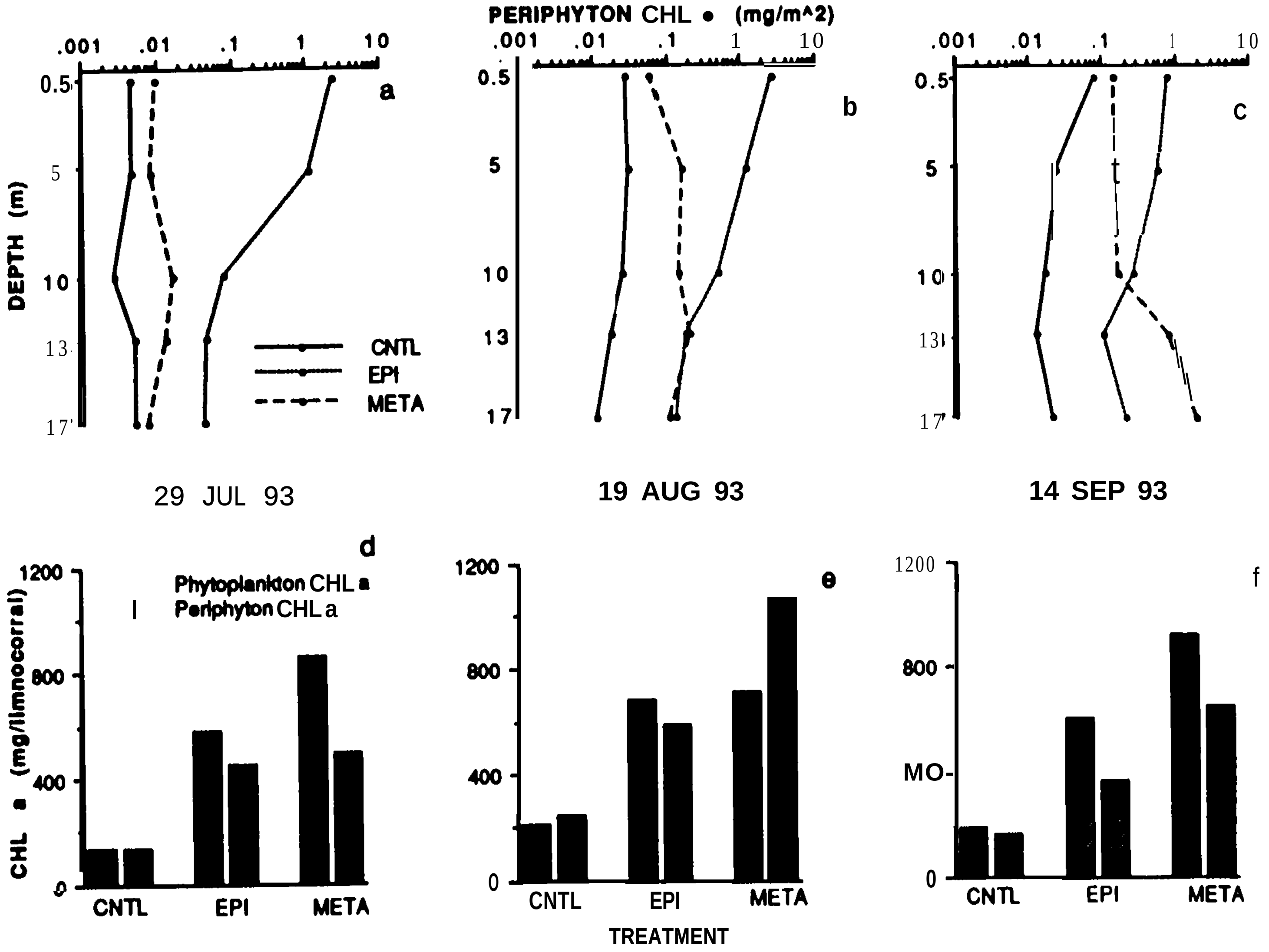

Figure 10. Periphyton levels in Pettit Lake limnocorrals, 1993, as **measured** from periphyton strip sampling. Mean periphyton density at 5 depths for control, EPI, and **META** treatments **(n=2),** as **mg/m**^{2} of chlorophyll a (a, b **&** c) and whole limnocorral chlorophyll a, **expressed** a s **mg/limnocorral,** i n phytoplankton ● Bd periphyton assemblages in each limnocorral **for** 3 dates in **1993** ☎♎📪 ● **&** ♐✆📬

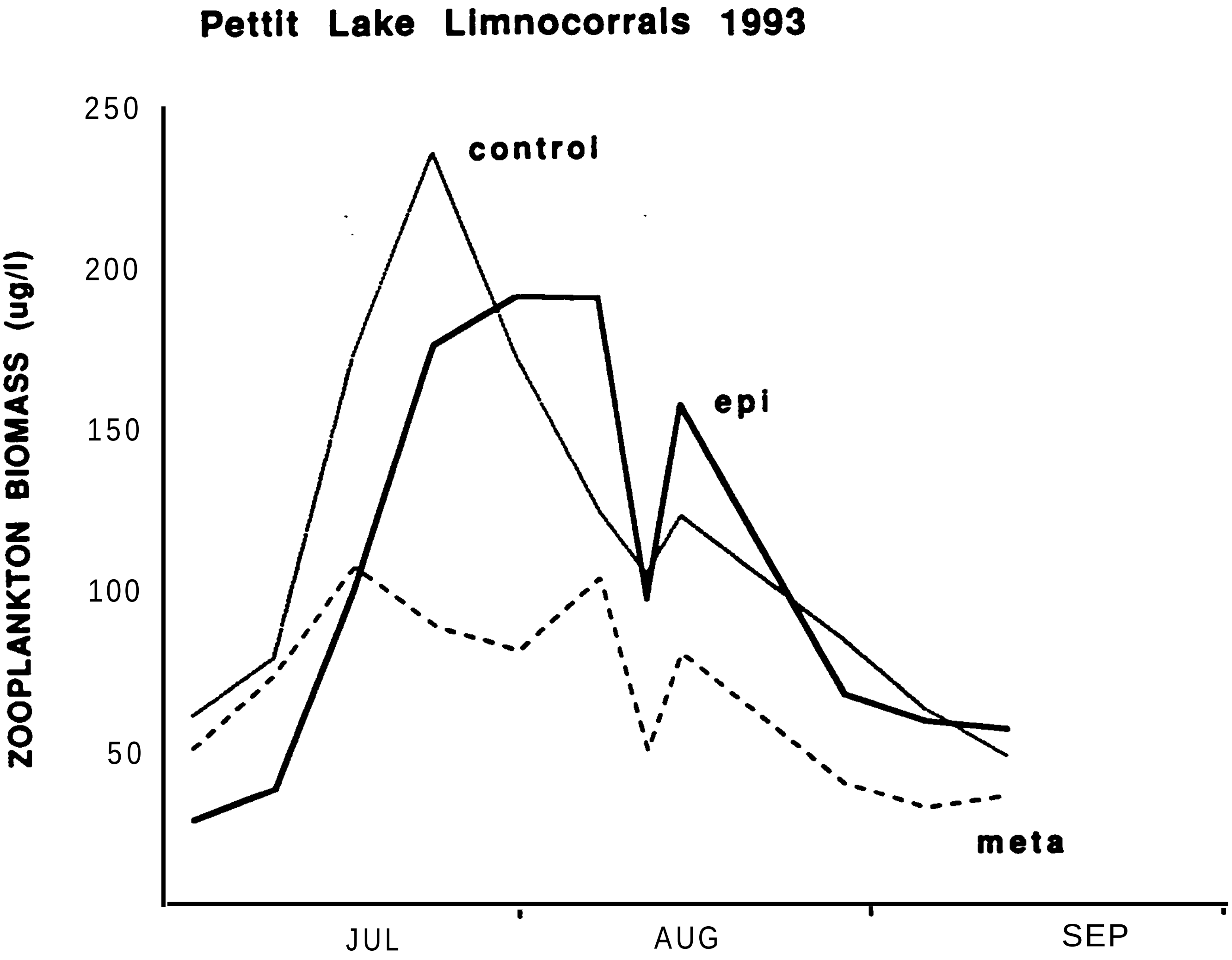

Figure 11. **Zooplankton** biomass **(µg/l)**, all **species** combined, in **each** limnocorral from **the** start of **the** experiment in July to the **end** in September. 10-0 **m** tows and **17-10 m** tows **are** summed.

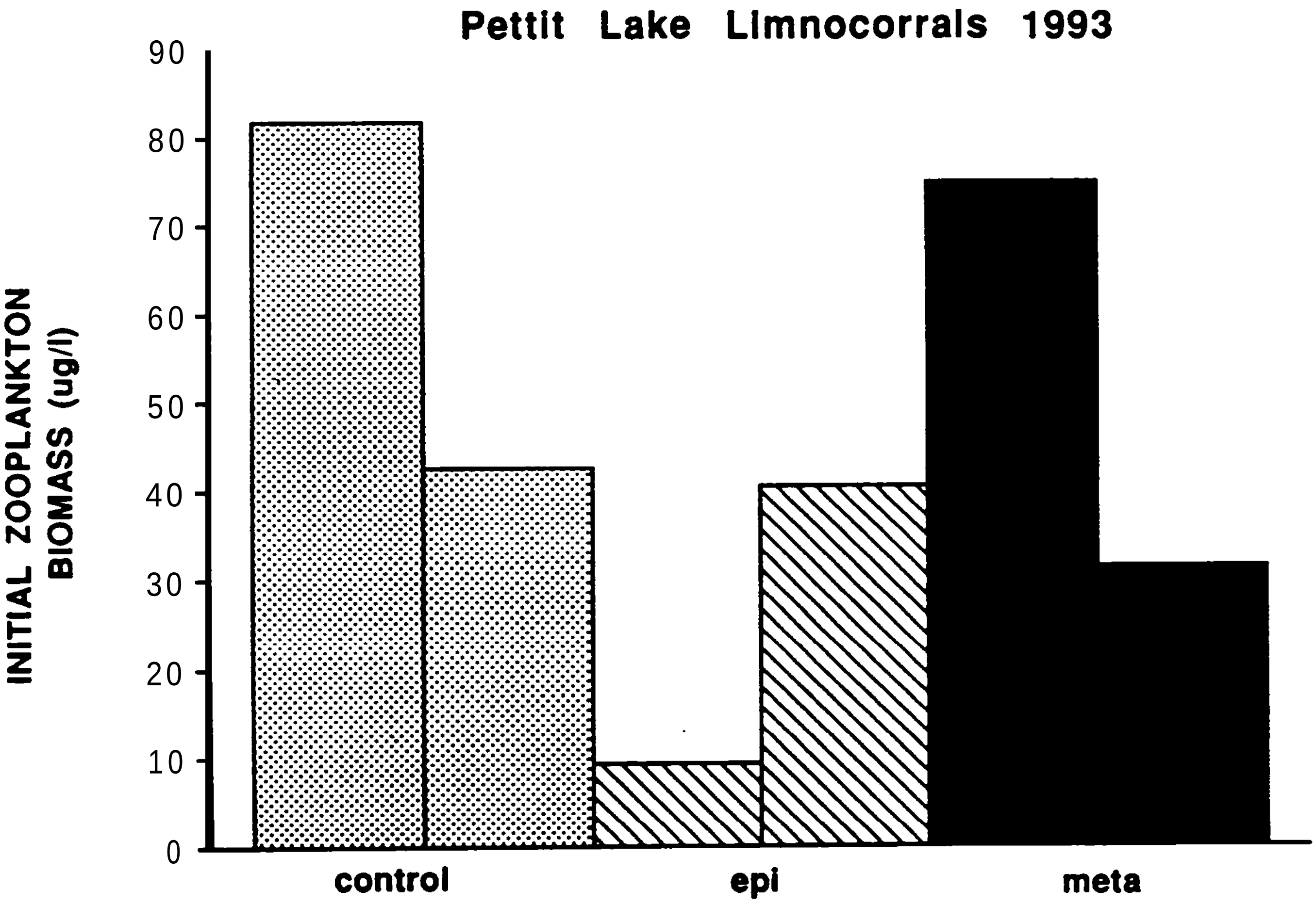

Figure 12. Initial rooplankton **biomass (µg/l)**, all **species** combined, in **each limnocorral** at **the** start of **the** experiment on 3 July 1993.

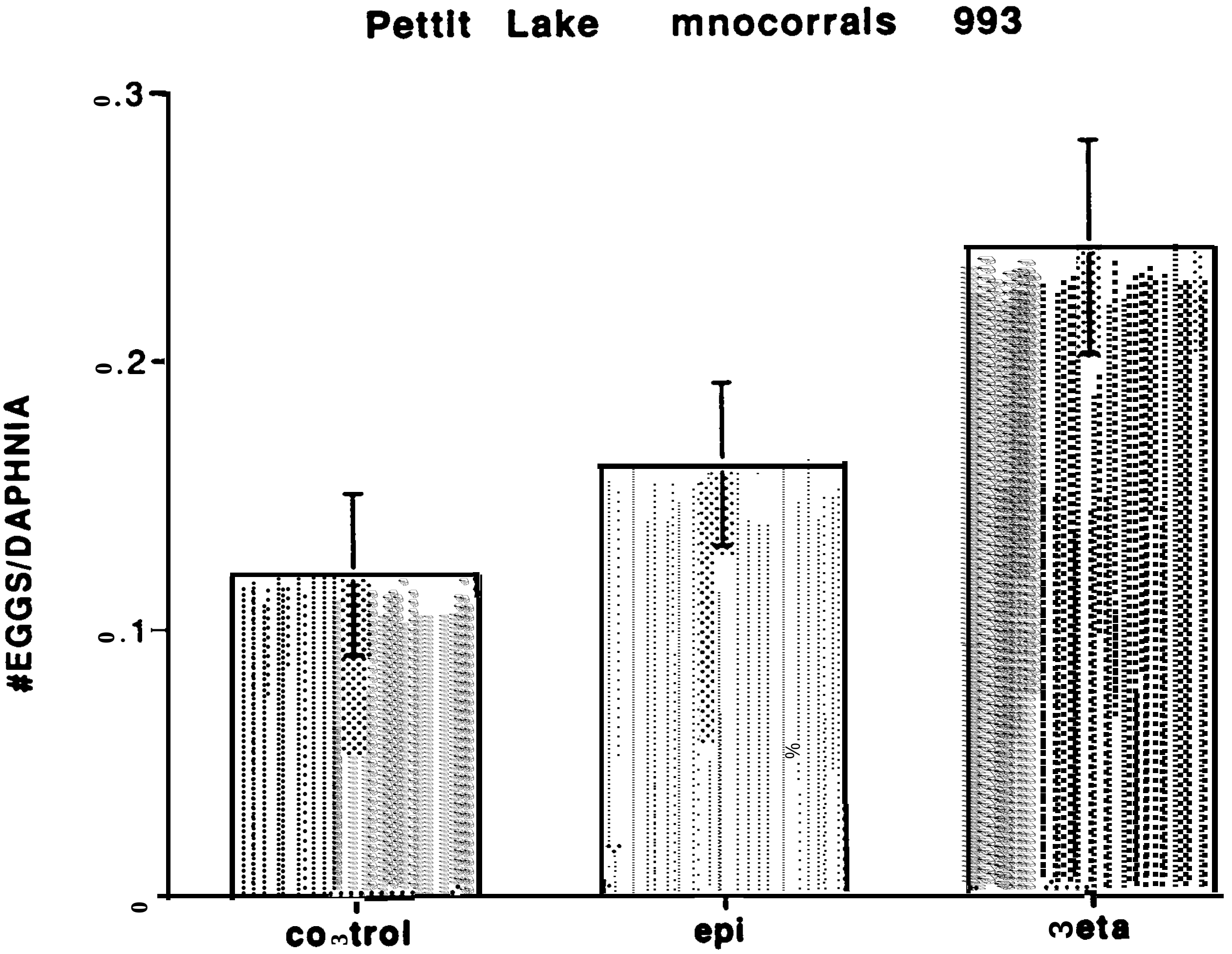

Figure 3. Egg production estimated as the number of eggs per female *Daphnia* in top ten meters of each treatment limnocorral; two replicates are averaged and standard error bars are shown (P= 0.036).

Fig. 14. Daphnia dominated in all limnocorrals throughout the duration of the experiment. All limnocorrals exhibited smaller, isolated concentrations of Bosmina in early July with second peaks also observed later in the experiment in both the EPI and META treatments. No other zooplankton species were present in high concentrations with the exception of one isolated representation of *Holopedium* in late August in one EPI treatment (LC3, Fig. 14).

Fish

Unfortunately the minnows used in the experiment seemed to be extremely sensitive to handling and few survived. Three fish were caught from a control treatment (LC4) at the intermediate sampling in July but no other fish were observed or caught at this time. Due to uncertainty concerning survivor numbers in the corrals, the three fish caught at the intermediate sampling were returned immediately to the limnocorrals.

During the fish removal at the end of the experiment, only four live fish were recovered from two limnocorrals (3 from LC4 and 1 from LC3) using the dry ice method. A few other dead fish were recovered by SCUBA divers from the bottom of the limnocorrals, and they were largely decomposed, suggesting that much of the mortality had occured early in the experiment. Further, most fish initially placed in the limnocorrals could not be accounted for at the end of the experiment. Because these fish could not be accounted for, no conclusions about planktivory or fish growth in response to nutrient addition could be made.

DISCUSSION

The 1993 Pettit Lake limnocorrals experiments demonstrated that addition of nutrients to either the epilimnia or the metalimnia of the lakes was effective in enhancing primary productivity and phytoplankton standing crop. However, increases in phytoplankton biomass, chlorophyll, and primary productivity rates in the metalimnetic fertilizations were similar or greater

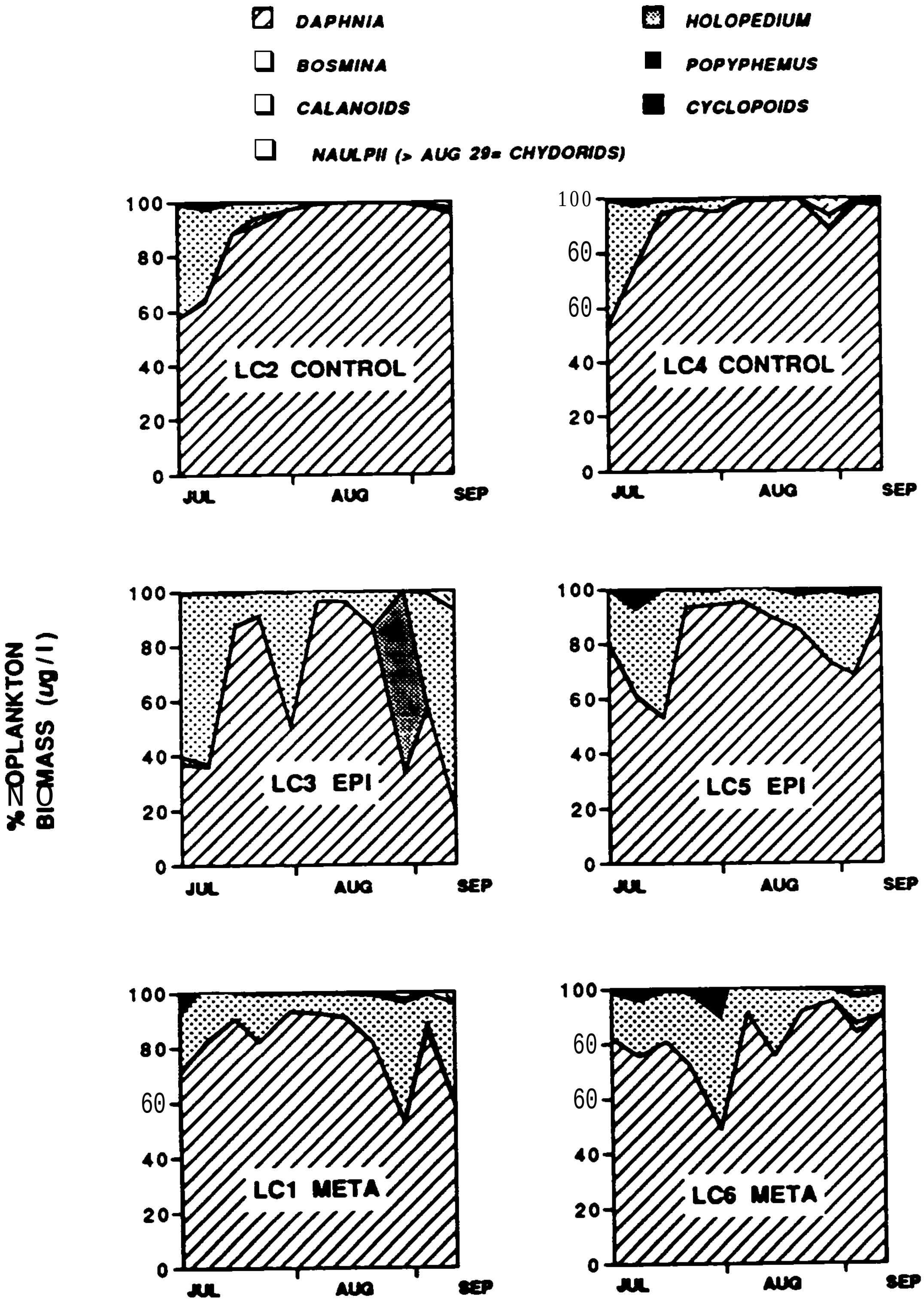

Figure 14. **Species** biomass composition **as a percent** of **the** total **zooplankton** in **each limnocorral** throughout **the** experiment.

than that in the epilimnetic treatments, yet water transparency in the metalimnetic treatments remained higher.

Nutrients added to either the epilimnia or the metalimnia of the corrals were apparently quite mobile, and stimulated algae in the adjoining strata. Nutrient concentrations were always elevated the most at the depths where they were added (Fig. 3), but concentrations also increased at other depths. In part, this was likely due to vertical mixing of limnocorral water during weekly sampling with zooplankton nets, Secchi disks and other equipment. While we were cautious to avoid unnecessary mixing during our sampling activities, some mixing was unavoidable. Little anthropogenic mixing would occur in a whole-lake fertilization, but this might be counterbalanced by increased turbulance due to internal waves in the lakes. In addition, zooplankton can redistribute nutrients if they have vertical migrations, although this redistribution can be downward as well as upward (Kitchell et al. 1979, Dini et al. 1987). Consequently, we would expect that a whole lake fertilization would stimulate algal production most in the layer where the nutrients were added, but that nutrients would be mixed between layers and stimulate phytoplankton throughout the photic zone.

Nutrient additions stimulated not only phytoplankton production, but the periphyton growth as well. The large increase of periphyton on the walls of the limnocorrals, is consistent with the results of other researchers who have also found these algae to be nutrient limited (Fairchild et al. 1985; Mazumder et al. 1989). The periphyton was stimulated much more by epilimnetic nutrient additions than by metalimnetic additions.

The large periphyton communities that developed near the tops of the fertilized enclosures may have competed for nutrients and limited the growth of the phytoplankton in the epilimnion. This was particularly evident in the epilimnetic fertilizations, where 20-50% of the chlorophyll in the enclosures was in the periphyton. The competition for nutrients may have allowed total phytoplankton chlorophyll levels to be higher in the metalimnetic nutrient treatments than the epilimnetic treatments (Figs 4a and 5). When combined periphyton and phytoplankton chlorophyll are considered, the differences between the epilimentic and metalimnetic nutrient treatments are reduced (Figs 10d-f).

A whole-lake fertilization would also promote periphyton growth in the littoral zone of a lake, particularly if nutrients were added to the epilimnion. The relative amounts of nutrients that would be diverted to the periphyton in a lake, however, would be much less than in the limnocorrals because of the much larger water volume:surface area ratio in the lakes than in the corrals. Because increased periphyton biomass is probably more noticible visually than is increased phytoplankton (Goldman 1974), it would be desireable to minimize the growth of the algae in the littoral zone. A metalimnetic fertilization again appears promising in this regard, as adding nutrients deeper in the water column promoted much less periphyton growth than did the epilimentic fertilization.

If whole-lake fertilizations are attempted in the Sawtooth Basin lakes, a more evenly-spaced addition of nutrients would be desireable to the early, large addition we used in the 1993 experiments. Distributing the nutrients evenly over the summer would likely decrease the strong pulse in chlorophyll that we observed in the fertilized limnocorrals in July (Fig. 3), and maintain production above controls latter in the summer. This might be particularly important in the Sawtooth Basin lakes because our bioassay experiments in 1992 indicated that the phytoplankton were more nutrient limited in July than in June (Gross et al. 1993).

The phytoplankton taxa in the limnocorrals in both the controls and in the fertilized treatments were generally in size ranges grazable by Daphnia. Porter (1973) found that phytoplankton >40 μm were rare in *Daphnia* guts, while Sommer (1988) found algae grazed by Daphnia to be <30 μm in all dimensions. In the limnocorrals, the only algae >30 μm were (1) some of the Gloeocystis colonies, *Dinobryon,* and the larger *Oocystis,* which are of lower food quality, and; (2) the colonial diatoms, which Daphnia can separate and break (Lampert 1978). Bosmina are more restricted by phytoplankton size than Daphnia (Sarnelle 1986), thus they probably benefitted much less from the diatom increases than the Daphnia did. However, biovolumes of the small *Chlorococcales* spp. <5 µm in diameter, highly usable by Bosmina, increased only slightly during the experiment.

Although size is a particularly useful parameter to assess whether phytoplankton can be grazed on by the zooplankton, there are many species-specific effects that complicate the analyses. For example, Knisely and Geller (1986) found that different species of zooplankton in the same genus have different ingestion and/or digestion rates for the same phytoplankton. Conversely, Infante and Litt (1985) found that a single zooplankton species may ingest different species of algae in the same genus at different rates. Nevertheless, we observed some recognizable trends in the limnocorrals that allow conclusions to be drawn about the effects of the nutrient additions on the zooplankton. The Pettit Lake zooplankton assemblage was dominated by Dapbnia *rosea* and Bosmina longirostris. Both are herbivores - *D. rosea* is a generalist, while *B.* longirostris is known to be a more selective feeder (DeMott 1982).

The major increases in biovolumes observed in the EPI and **META** fertilizations were caused primarily by the diatom *Fragilaria*, but also by Tabellaria, Asterionella, and *Synedra*. These diatoms occurred in colonial assemblages, except for the *Synedra*, which were solitary. Daphnia can separate individual diatoms from their colonies in order to ingest them (Lampert 1978, Infante and Litt 1985).

The quality of *Fragilaria* spp. as food varies from low to high, depending on the species of Dapbnia, but unforuntatly *D. rosea* has not yet been evaluated (Knisely and Geller 1986, Infante and Litt 1985). Infante and Litt also found *Tabellaria fenestrata* to be of intermediate and high value for two species of Daphnia. Sarnelle (1986) found that brood size of *D.* catawba increased when

Fragilaria and *Asterionella* were dominant. Most investigators have found that *Asterionella formosa* to be of low to intermediate edibility for a variety of Daphnia spp., as well as being highly used by cyclopoid copepods (Schindler 1971, Porter 1973, Lehman and Sandgren 1985, Tóth and Zdnkai 1985, Knisely and Geller 1986, Sommer 1988). The solitary *Synedra* sp. in the limnocorrals was <5 µm wide and ranged in length from 25-115 µm. Sommer (1988) found that two species of *Synedra* were used at medium to optimal rates by *Daphnia longispina* and *D. magna.*

Increases in the chlorophyta in the epilimnetic fertilizations on 29 Aug. 1993 resulted from species of poor to medium food quality. The Gloeocystis sp. was contained in a gelatinous sheath, which is indicative of algae that is a poor planktonic food source (Porter 1973, Sarnelle 1986, Sommer 1988). The other contributor to this increase was *Spondylosium* sp., a filamentous desmid. Filamentous desmids are loosely connected (Hutchinson 1967, p. 325), *so Daphnia* should be able to detach and ingest *Spondylosium*. The chlorophyta increases at 17 m in the META treatments were primarily due to *Chlorella* sp. >5 µm in diameter, which are of medium to high food quality (Knisely and Geller 1986).

In summary, the nutrient additions primarily increased the abundance of diatoms in the limnocorrals. These increases came without substantial coinciding increases innuisance Cyanobacteria. The diatoms should have been most usable by the *Daphnia* in Pettit Lake. While diatoms are not uniformly of the highest food quality for cladocera, they should have been sufficient to- increase zooplankton productivity.

Although fertilization increased primarily desireable species of algae, the epilimnetic treatment did show increases in the cyanobacteria *Oscillatoria.* Because cyanobacteria often do well when N:P ratios are low, we had attempted to add excess nitrogen fertilizer. Nevertheless, the *in* vitro bioassay (Fig. 9c) demonstrated that phytoplankton in the EPI limnocorral treatment were still primarily nitrogen limited in August. If lake fertilizations are attempted in the Sawtooth Basin lakes, it would be desireable to add an even higher N:P ratio than the 20:1 ratio used in the limnocorral experiments.

Our analyses of the zooplankton and fish responses to nutrient additions were confounded by high variability in zooplankton abundance between treatments, and the loss of fish in most of the enclosures. Nevertheless, the significant effect of fertilization on *Daphnia* egg production (Fig. 12) suggests that zooplankton can potentially be enhanced by this procedure in the Sawtooth Basin lakes. The metalimnetic nutrient addition increased egg production more than did the epilimnetic additions, again suggesting that this method of fertilization may be beneficial. Futher tests of will be conducted in 1995 to determine if zooplankton abundance, and more importantly, *Onchorhynchus nerka* production, will respond to metalimnetic fertilizations.

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Chapter 5

Assessment of Fish Growth Potential using Bioenergetics Simulations and a Field Test with Kokanee Salmon

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INTRODUCTION

Understanding the factors determining fish growth is an important aspect of managing endangered fish populations. The combined effect of temperature and food availability is particularly important in determining growth and consequently survival of juvenile fish (Brett 1985; Miller et al. 1988; Werner and Gilliam 1984; Houde 1987). The quantity of food available determines the energy available for growth after maintenance costs and the optimal temperature for growth. The temperature inhabited by a fish affects how rapidly that food is metabolized and the energy losses due to maintenance costs (Adams and Breck 1990; Brett et al. 1969; Brett 1983; Shelbourn et al. 1973; Wooton 1990). Understanding the interaction between temperature and food consumption on growth will aid in management decisions concerning the rehabilitation of endangered Snake River Sockeye Salmon.

This endangered stock historically spawned and reared in five high altitude (6,800'), oligotrophic lakes located in the Sawtooth Valley, central Idaho. These lakes exhibit different depth regimes and vary considerably in the food available to the rearing juvenile sockeye salmon. Zooplankton biomass can be considered a measure of the food available to fish in the different lakes because sockeye are predominantly zooplanktivores (Burgner 1992; Cordone 1971). In oligotrophic lakes such as these, small differences in food availability can have a considerable effect on growth both directly through energy consumed and indirectly through the combined effect of temperature and ration on energy conversion and growth.

Brett et al. (1969) have shown that for juvenile sockeye grown with excess ration, the optimal temperature for growth is 15°C, while at reduced ration (1.5% of body weight/day) the optimal temperature for growth is 5°C. This drop in the optimal temperature for growth with decreased ration reflects the decrease in maintenance metabolism that accompanies reduced temperature. They also showed that the effect of temperature on growth is most pronounced in the early stages of life when growth rates are high, but lessens as growth rates decline in older fish. Thus temperature, in combination with food availability, will have its most pronounced effect in the juvenile, rearing stage of sockeye salmon (Brett 1983; Post 1990).

As part of the assessment of rearing habits for juvenile sockeye salmon, we performed a series of growth experiments at different depths and consequently different growth regimes in two of the five Sawtooth sockeye lakes. Kokanee salmon, a conspecific strain of sockeye salmon (both *Oncorhynchus nerka)* were used as a surrogate for the endangered sockeye salmon in these experiments. Fish were grown in net pens in Stanley and Redfish Lakes. These two lakes were chosen because they provide the greatest difference in phytoplankton productivity and zooplankton biomass of the five lakes.

Given the higher ration available for rearing salmon in the more productive Stanley Lake, we anticipated that temperature would have a more pronounced effect on growth here than in the less productive Redfish Lake. Further, we expected that increased zooplankton biomass would enhance growth when temperatures were similar.

A simulation model designed for predicting juvenile sockeye growth was also tested and evaluated. The model used the bioenergetics simulation approach (Beauchamp et al. 1989; Hewett and Johnson 1992) in combination with a zooplankton consumption to zooplankton availability relationship derived from kokanee growth experiments (Teuscher 1993; Bowler and Riemen 1981; Post 1990). Zooplankton biomass and water temperature provide parameters specific to a given lake and time period while physiological estimates of consumption and metabolic rates are generalized across lakes (Beauchamp et al. 1989). Therefore, a bioenergetic model driven by food availability and temperature provides the opportunity for general predictions based on easily measured limnological data.

In this paper, we compare temperature, zooplankton availability, and fish growth responses among net pens in the two lakes. Measurements of temperature and zooplankton availability were used to predict growth of juvenile kokanee using the bioenergetics simulation model. Growth rate predictions were compared to observations from our field experiment. This type of modeling approach presents the opportunity to compare growth conditions for kokanee among different systems with varying environmental factors that determine growth.

METHODS

Net pen experiments were performed in Redfish and Stanley Lakes, over a 2 1/2 month period (June-September) in the summer of 1993. The two lakes were chosen because: (1) emergent kokanee salmon were available from these lakes, and; (2) Stanley and Redfish provide differences in productivity, allowing us to study factors affecting growth both among different productivity levels and throughout varying depth and consequently temperature regimes.

Juvenile kokanee for the Stanley net pens were seined the day the experiment was started, 16 June 1993, along the south-west shore of Stanley Lake. Fish were anesthetized, measured (TL), and weighed (g) before being added to net pens. Only those fish that appeared to show no signs of handling stress were used in the experiment. A sub-sample of fish was measured and frozen to estimate the initial dry weights of fish used in the experiments.

Juvenile kokanee used in the Redfish Lake net pens were caught immediately after emergence from the gravel in a Redfish inflow--Fishhook Creek, 16-19 May 1993. Fish were then transported to a hatchery in Eagle, ID where they were reared until the start of the experiment on 30 June 1993. The day before the experiment, fish were anesthetized, measured, and weighed, and then observed over twenty-four hours. Only survivors not affected by handling were added to the net pens.

Six net pens were used in each of the two lakes with three treatments and two replicates. The treatments were as follows: two epilimnetic net pens placed at 6 m, two metalimnetic net pens placed at 12 m, and two hypolimnetic net pens placed at 18 m. The net pens used were 2x2 m cylinders constructed with 1/8" ace netting, with a floating spreader bar at the surface at the top and a sinking spreader bar on the bottom. At the start of the experiment, net pens were weighted at the bottom, fish were added to the pens, and then the pens were slowly dropped such that the middle of the net pen was at the desired depth.

The fish from net pens were sampled for an intermediate growth measure approximately half way through the experiment (21 July - Stanley, 28 July Redfish). The pens were slowly brought to the surface, fish were removed, anesthetized, measured, and weighed. After all fish had recovered from handling, they were added back to the pens which were then repositioned as close to their original depth and location as possible.

At the end of the experiment, the pens were brought to the surface, fish were anesthetized until death, measured, and weighed. After a sub-sample of fish was preserved for stomach analysis, remaining fish were frozen for dry weights. Stomach samples were counted by species, individuals were measured, and biomass was calculated (McCauley, 1984).

Every 7 d, the net pens were sampled for temperature, oxygen, and zooplankton availability. An oxygen and temperature profile was taken at a hypolimnetic net pen from 0-20 m using a YSI meter, with readings recorded every meter. A closing, Wisconsin style zooplankton net equipped with an anti-reverse Oceanics flow meter was used to take a zooplankton tow at each net pen. A 4 m tow taken immediately outside the pen, from one meter below the net pen to one meter above the net pens, was chosen as the sample representing zooplankton available to those net pen fish. Zooplankton samples were counted, measured, and biomass was calculated, and corrected for volume filtered, according to length-mass regressions published by McCauley (1984) and Koenings and Burkett (1987).

To account for potential differences in zooplankton biomass inside the pens versus outside the pens, twice during the experiment, zooplankton were sampled inside the net pens using a pump and SCUBA divers. Towards the beginning of the experiment (16 and 30 July) 24 1 of water were pumped from inside each pen and then compared with 24 1 of water pumped outside the pens in both Redfish and Stanley Lakes. Samples were enumerated and measured according to methods described above. The same procedure was again repeated towards the end of the experiment, on 7 and 8 September 1993 in Stanley and Redfish Lakes respectively, but the volume filtered was increased from 24 1 to 120 1. A linear regression of inside zooplankton biomass versus outside zooplankton biomass in addition to a step-wise multiple regression (minimum tolerance =.05) of inside zooplankton as dependent on outside zooplankton biomass, temperature, and density were done to explain variance.

A step-wise multiple regression analysis was done using net pen fish growth (g/day) against fish density (#/pen), food availability (zooplankton biomass μg/l), and temperature (°C) for results from both lake experiments combined (minimum entry tolerance = .05). Repeated measures ANOVAs were done on fish weight through time for both lake experiments. Linear regressions were applied to fish growth (g/day) as a function of temperature and as a function of food availability for both lakes independently. All statistical tests were done in SYSTAT (Wilkinson 1990).

The Model:

The model parameters are reported in Beauchamp et al.'s (1989) sockeye bioenergetics model. The model was constructed and runs in Stella Computer programming software for Macintosh computers. Food availability, temperature, and initial fish weight were inputs to the model and food availability was calculated as total available zooplankton biomass (Brett et al. 1969, Brett 1988, Goodland et al. 1974, Kitchell et al. 1977; Mills and Forney 1981).

Two different approaches to consumption were modeled and evaluated. The first method relates the proportion maximum daily consumption (P-value) exhibited by a fish to total zooplankton biomass (μg/l) (Post 1990)(hereafter referred to as the P-value model). P-value is expressed as the percentage of maximum consumption (Beauchamp et al. 1989).

(1) **P-value=** (0.035' Zooplankton Biomass (μg/l) •
(1+0.077* Zooplankton Biomass)"

The second method relates weight specific consumption to zooplankton biomass (hereafter referred to as the Consumption Model:. Consumption cannot exceed maximum consumption (Cmax) with this relationship.

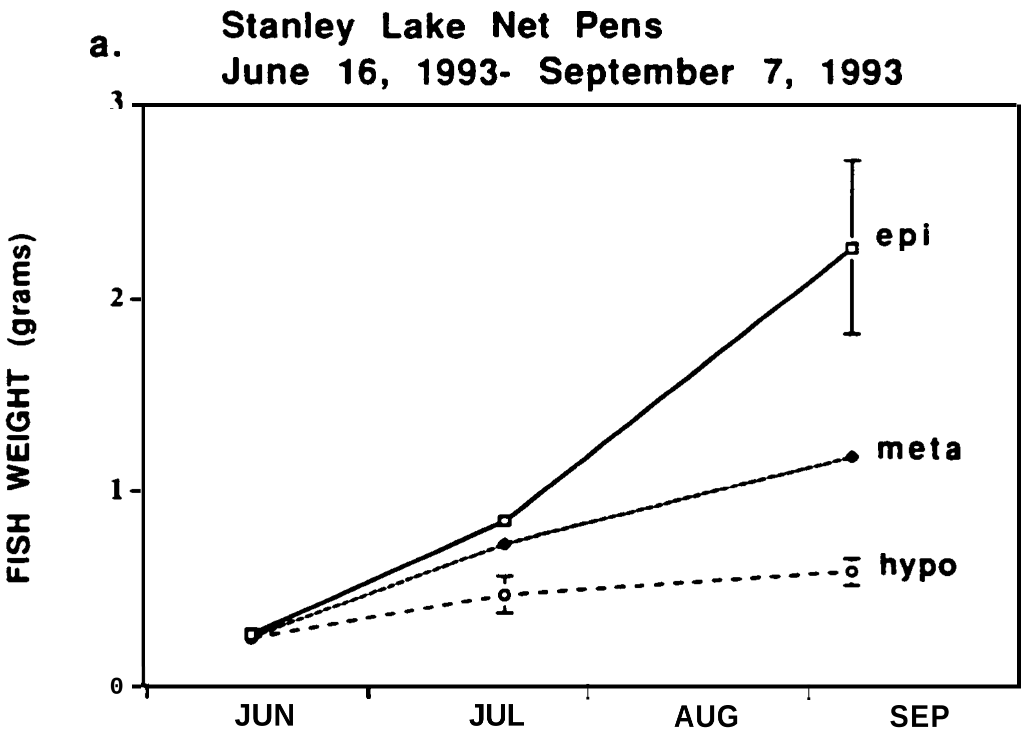
(2) **Consumption=** IF (.0011470 + .038486 * LOG10 (Zooplankton
Biomass (μg/l) • (Cmax/0.2147))) ELSE (Cmax)

Both relationships were derived from bioenergetic simulations of several data sets from experiments involving kokanee or sockeye (*O. nerka)* grown under varying environmental conditions (Teuscher 1993; Bowler and Rieman 1981). In these simulations P-value and consumption were estimated from reports of fish growth and temperature. These P-values or consumption estimates were regenerated as a function of zooplankton biomass to derive equations 1 and 2.

Weekly temperature and zooplankton values and initial fish weights were used as inputs to the model. Simulations representing the time interval of the net pens were run, and model predictions were compared to observed growth.

RESULTS

Fish weights throughout the experiment are shown in Figure 1 a-b. Results are given as the average of all fish from a treatment and as the mean of both replicates. Stanley Lake fish showed the highest final fish wet weight in the epilimnion, second highest in the metalimnion, and lowest in the hypolimnion (RM ANOVA df= 4, F= 8.85763, P= .01083). In Redfish Lake final fish weights in the net pens were highest in the metalimnion, slightly lower in the epilimnion, and lowest in the hypolimnion, but no significant difference among treatments was observed (RM ANOVA df= 4, F=.85928, P=.53781). Removing the metalimnetic treatment from the



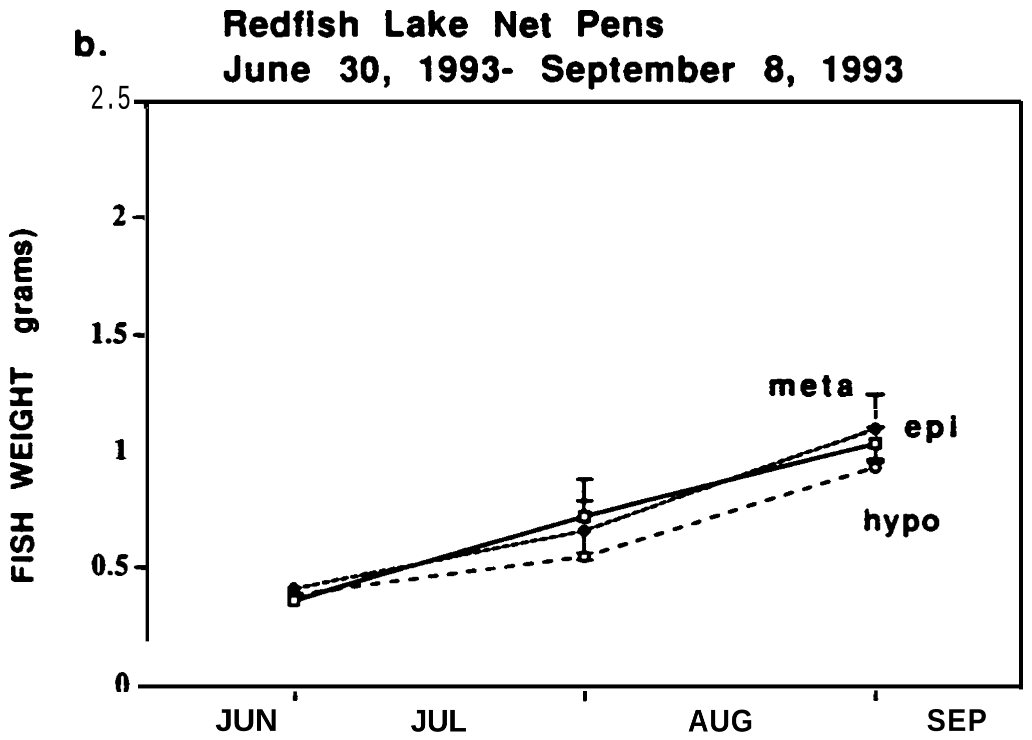


Figure 1. Fish growth measured as wet weight (grams) in a.) Stanley Lake net pens from the start of the experiment 16 June, 1993 to the end 7 September, 1993 and in b.) Redfish Lake from the start of the experiment 30 June, 1993 to the end 8 September, 1993 with one intermediate data point. Points shown are the mean of two replicates with the range given.

statistical analysis in order to determine if an effect between epilimnion and hypolimnion existed proved futile ($P=.358$).

Final dry weights of fish exhibited similar patterns in fish size as compared to initial size (Figs. 2 a-b.). Final weight was highest in the epilimnion, second highest in the metalimnion, and lowest in the hypolimnion in both Stanley and Redfish Lakes. However, the difference was much more pronounced in Stanley Lake (Fig. 2a) than in Redfish Lake net pens (Fig. 2b). Stanley net pen fish showed a greater overall increase in weight in the epilimnion and metalimnion treatments than Redfish Lake net pen fish. In Stanley Lake, the metalimnion fish had a slightly higher % dry mass (22.1%) than did fish in the epilimnion (21.9%) or hypolimnion (19.2) in Stanley, but the standard error was high. In Redfish, the highest % dry mass was observed in the epilimnion (19.6%), followed by the metalimnion (19.23%) and then hypolimnion (18.0%). The % dry mass present at the end of the experiment was positively correlated with growth rates among all net pens.

Growth was most rapid after the intermediate fish sampling point on 21 July, 1993 in Stanley and on 26 July, 1993 in Redfish. Final fish growth, in g/day, was correlated with temperature (Fig. 3a) in the Stanley Lake net pens. Highest growth rates were observed in the warm epilimnion treatment (average temperature = 12.8°C, Table 1) and lowest in the colder, hypolimnion treatment (average temperature = 7.0°C, Table 1) ($R^2=.809$, $P=.015$). Growth of Redfish Lake net pen fish, however, was not significantly affected by temperature differences among treatments ($R^2=.056$, $P=.653$) (Fig. 4).

Considerably more zooplankton biomass was present outside Stanley Lake net pens than for Redfish Lake (Table 1). Fish growth, g/day, as a function of zooplankton availability (μg/l) immediately outside the pens (4m tow) is shown for Stanley Lake net pens in Figure 4a and for Redfish Lake net pens in Figure 4b. Neither fish growth in Stanley nor Redfish Lake net pens was significantly correlated with average food availability outside the pens (Stanley $R^2=.082$, $P=.582$; Redfish $R^2=.007$, $P=.872$).

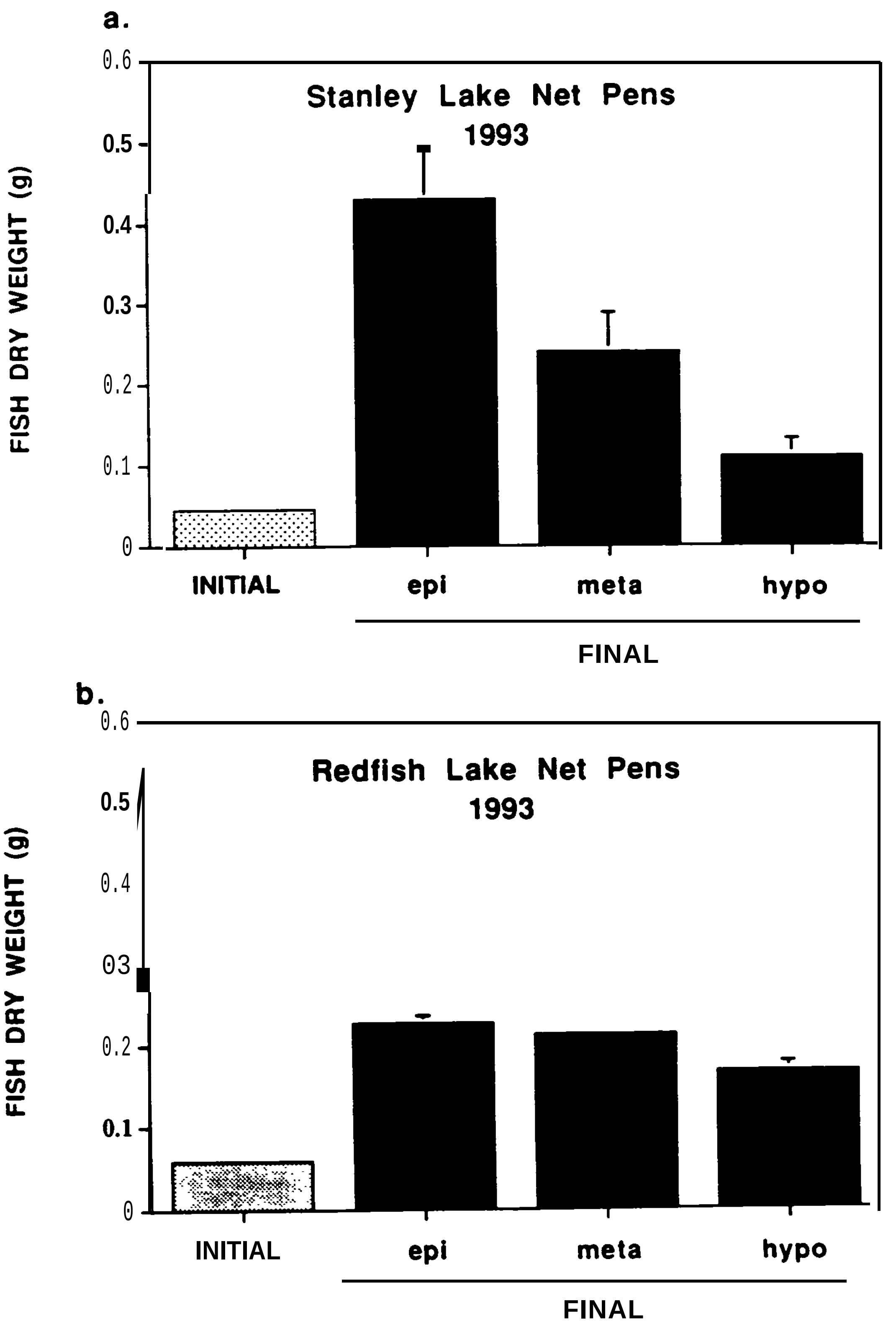


Figure 2. Fish growth measured as dry weight (grams) comparing initial fish samples to final fish samples in a.) Stanley Lake and b.) Redfish Lake. The bars shown are the mean of two replicates with the standard error bar.

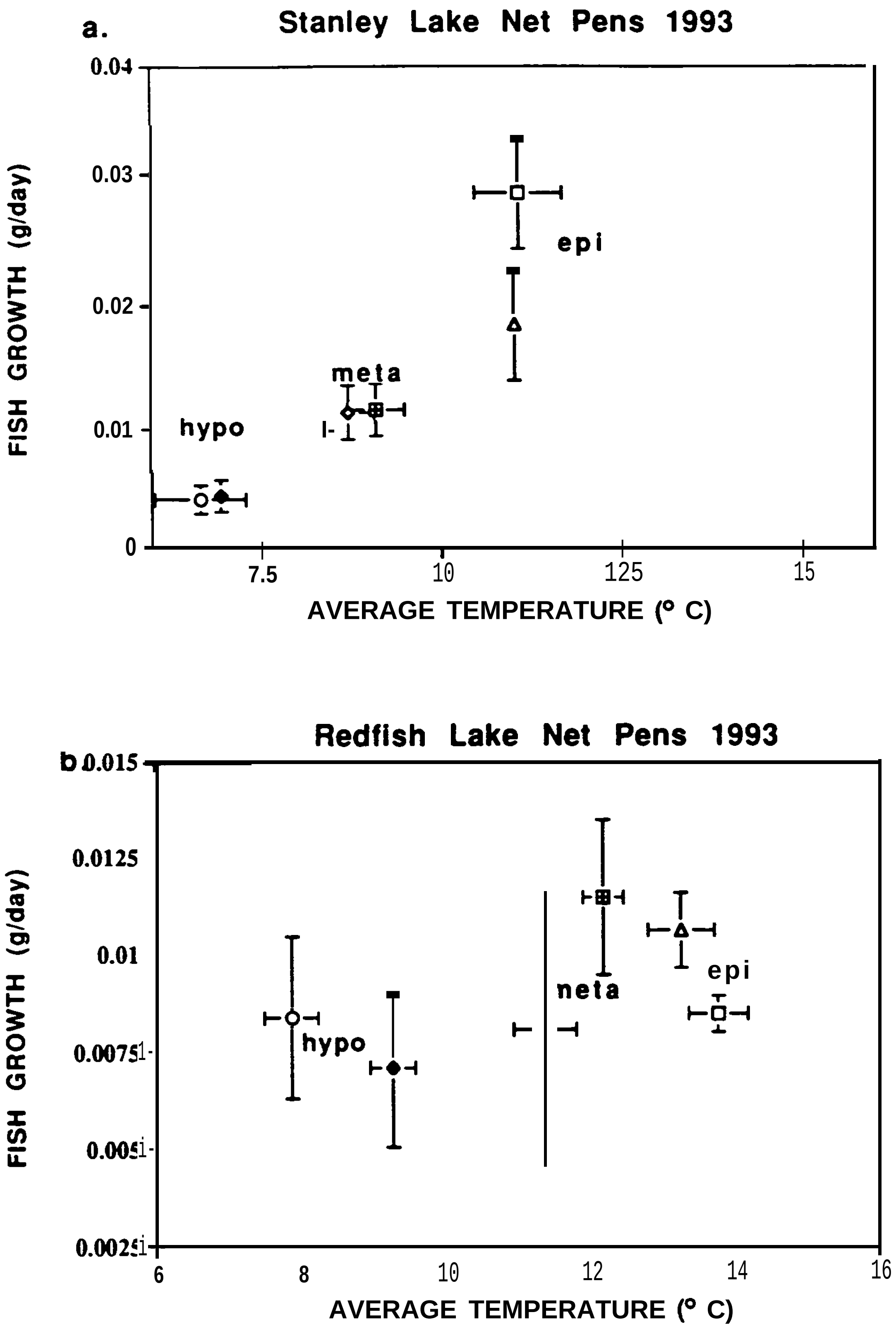


Figure 3. Fish growth (g/day) versus mean **temperature (°C)** at the mid-point of each pen in a.) Stanley Lake and **b.) Redfish** Lake. Points represent individual net pens, and error bars represent one S.E. for both axes.

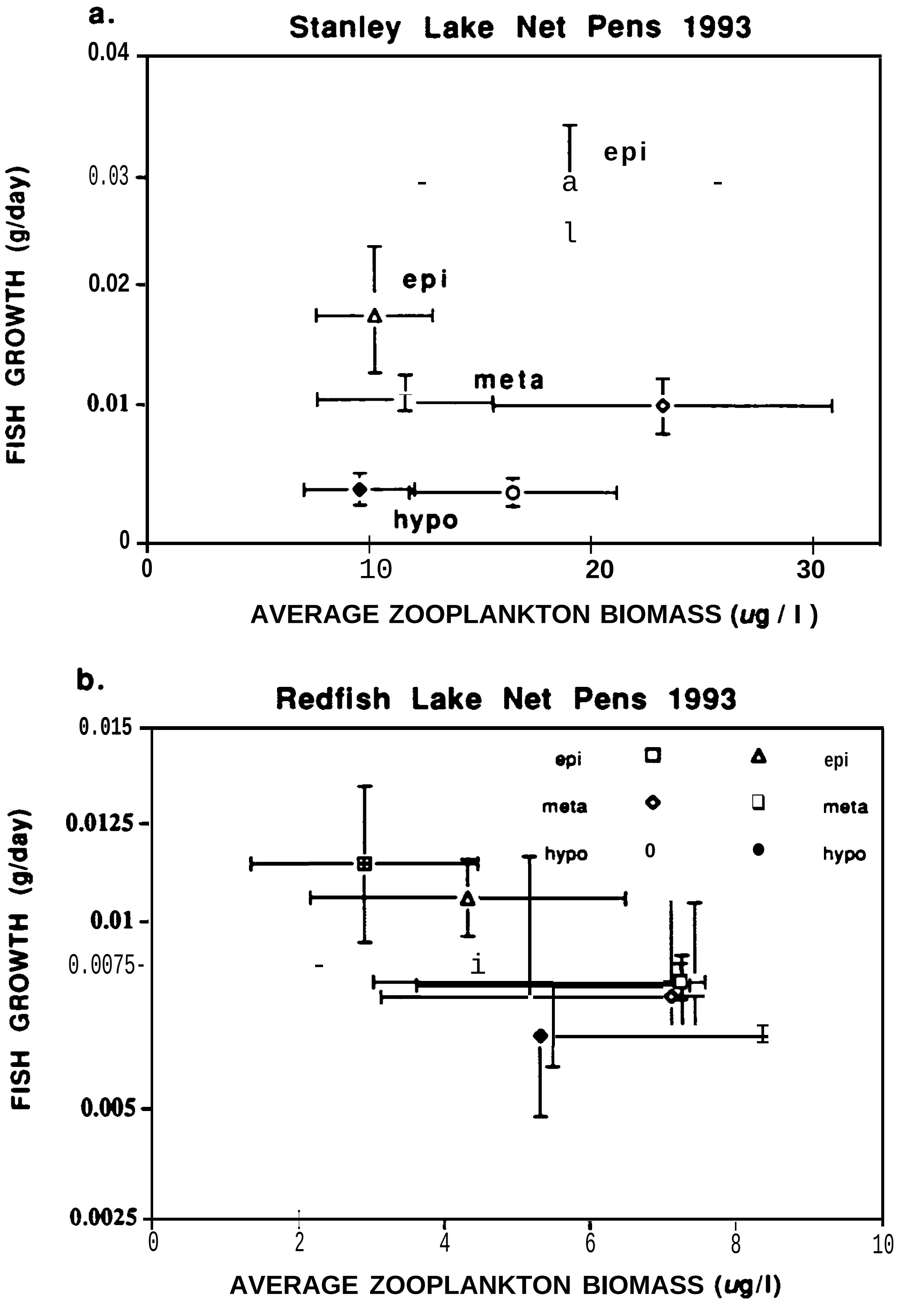


Figure 4. Fish Growth (g/day) versus mean total zooplankton biomass (µg/l) measured **immediately** outside each net pen, over a 4 meter tow in a.) Stanley Lake and **b.) Redfish** Lake. Points represent individual net pens, and error bars represent one S.E. for both axes.

Zooplankton biomass inside the net pens was positively related to zooplankton biomass outside the net pens on two sample dates but with a slope of 0.4 (Fig. 5, R^2= 0.413, P=.0076). This comparison suggests that fish inside the net pens were depleting zooplankton resources faster than they were being replaced by movement from outside. Zooplankton biomass inside the pens was variable, but distinct patterns were detectable among lakes and depths and fish survival observations. A step-wise regression of factors influencing inside zooplankton biomass indicated that both outside zooplankton biomass and water temperature explained a significant portion of the variation in inside-pen biomass patterns (outside zooplankton biomass (44.1%), temperature (24.2%)(R_2=.688, df= 3, F= 16.44127, P=.00002). Together the two factors explained over 68% of the variation in zooplankton biomass measured inside the net pens. Fish density had a pronounced effect only in pens placed in the warm epilimnetic waters. These results suggests that higher fish consumption rates in warmer water casued a greater depletion of zooplankton inside these net pens.

A step-wise multiple regression of net-pen fish final weight as dependent on temperature, food, and density of surviving fish at the end of the experiment was significant and ranked the variables with temperature first explaining 28% of the variance, food second explaining 20% of the variance, and kokanee density last explaining 15% of the variance (R^2=.489, df= 3, F= 4.51427, P= 0.0392).

Oxygen levels remained above 5 mg/l throughout the water column for the duration of the experiment in both lakes. Therefore, we assumed that oxygen was not a limiting factor affecting growth (Brett 1979). Unfortunately the light levels in the Stanley Lake hypolimnion likely reduced visual planktivory in that treatment. The 1% light level ranged from 12 to 18 meters, and the hypolimnion net pens were placed at 19-21 m at the mid-point of the pen. Consequently, the hypolimnetic growth results in Stanley were not modeled as the model predicts growth based on a feeding response that assumes high light levels for visual planktivory.

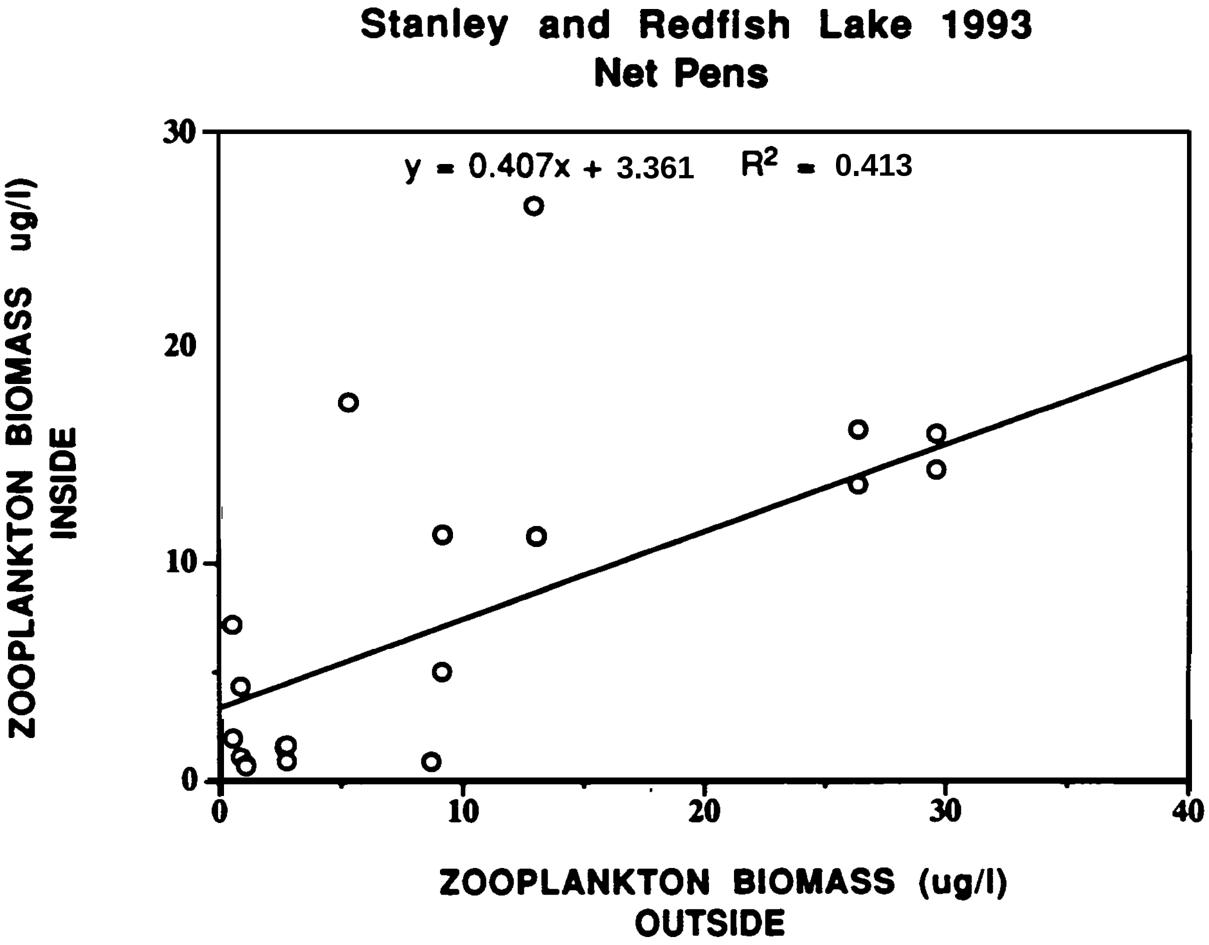


Figure 5. Zooplankton biomass (µg/l) measured inside each net pen with a pump versus zooplankton biomass (µg/l) measured outside each net pen with a pump.

The observed fish growth versus the fish growth predicted by the model is shown in Figure 6 a-b. The P-value model was closer in predictions to observed growth than the Consumption model in six cases overall. The P-value model and the Consumption model had almost equal predicting accuracy by treatment, but the P-value model did a better job at predicting differences in growth between the two lakes.

DISCUSSION

Most lakes undergo large seasonal changes in temperature and zooplankton biomass and the degree of thermal stratification (Goldman and Horne 1983, Wetzel 1983). This results in fish occupying habitats of varying temperatures and food availability. Vertical migration patterns can further complicate the effects of habitat use by fish (Clark and Levy 1988, Levy 1987). Further, in the Sawtooth Valley Lakes, lake productivity, food availability and temperature are generally low and consequently small differences in any of these factors can result in pronounced differences in growth and survival (Budy et al. 1992; Rieman and Meyers 1992; Stockner 1981).

Our experiment allowed us to examine how the interaction of temperature and food determined growth both across temperature regimes and at food levels. The experiment showed that temperature had a more pronounced effect on growth in Stanley Lake where zooplankton food availability was greatest. In contrast, fish growth in Redfish Lake was similar at all temperatures.

The experiment also revealed differences in growth of fish grown at similar temperatures but at different food levels. Juvenile kokanee in Stanley Lake net pens grew faster in both the epilimnion and the metalimnion as compared to individuals at similar temperatures in Redfish Lake. These differences are the result of the higher overall food availability in Stanley Lake as well as the differences in zooplankton community species composition in the two lakes. Redfish Lake is an ultra-oligotrophic lake dominated by Bosmina Zongirostris and *Holopedium*

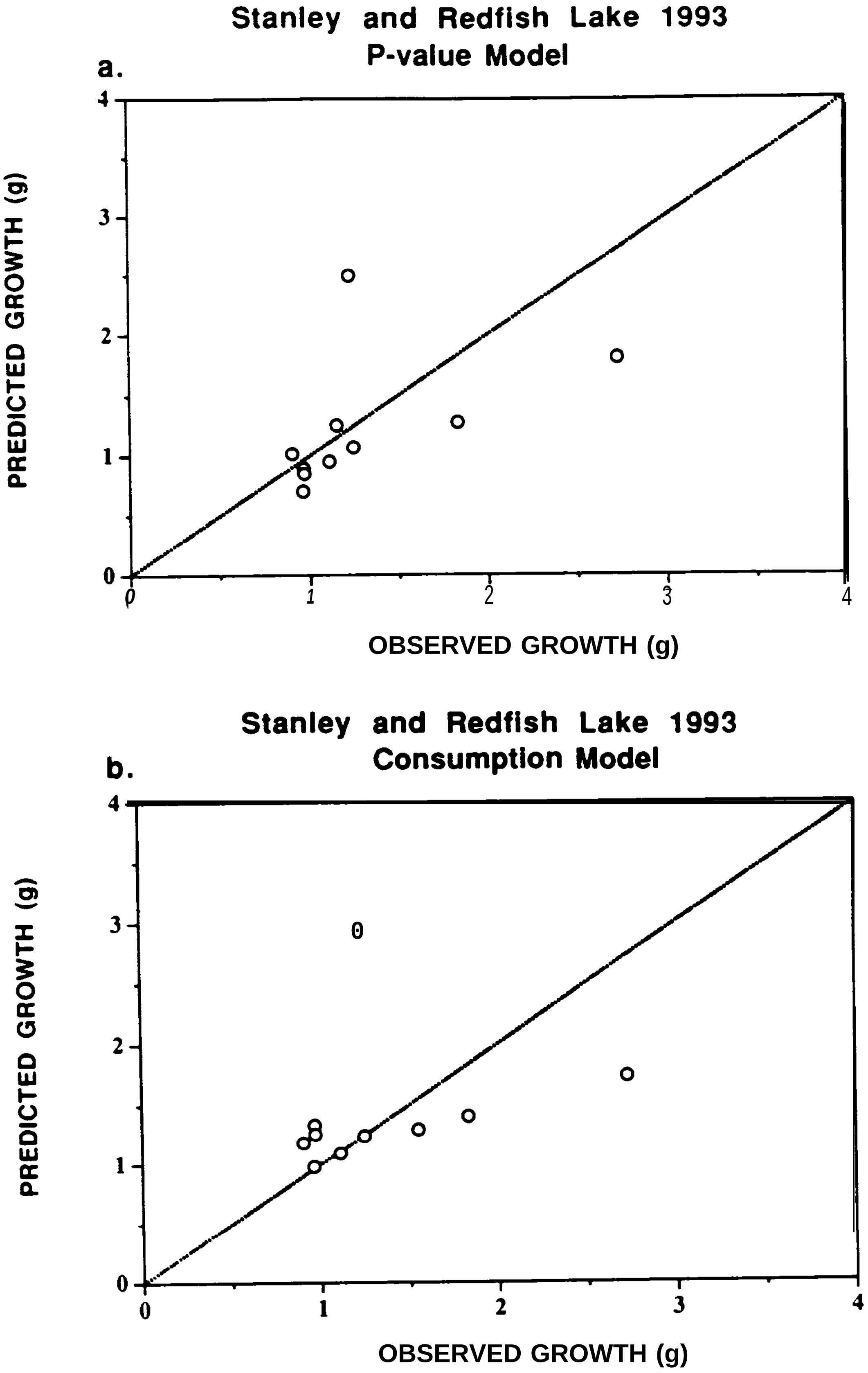


Figure 6. Observed versus predicted growth by the simulation model using the two different approaches a.) P-value model and b.) Consumption Model. The one to one line is given as a reference. The mean size of initial fish was .25 g and in Stanley Lake and .39 g in Redfish Lake.

gibberum while Stanley is considerably more productive (although still considered oligotrophic) and zooplankton species composition is characterized by larger, cladoceran zooplankton species *(Daphnia rosea*) and calanoids *(Epischuira nevadensis)* (Budy et al. 1992; Rieman and Meyers 1992).

Cordone et. al. (1971) found the diet of kokanee fry in Lake Tahoe, California, to consist primarily of cladocerans. Mills and Fourney (1981) have shown that YOY yellow perch exhibit lower cohort mortality when *Daphnia pulex* is abundant but switch to alternative prey when *Daphnia pulex* abundance is low resulting in an overall decline in growth. The greater abundance of a larger, more preferred and energetically profitable cladoceran prey likely contributed to the higher growth rates observed in Stanley Lake compared to Redfish Lake.

These results occurred even though differential survival of fish in pens likely confounded model predictions. The net pen with the greatest deviation in from predicted growth had high final fish density (10 fish/pen). Competitive effects of this high density of fish may have reduced the mean growth of these individuals (Kyle et a1.1993; Goodland et. al. 1974; Carpenter and Kitchell 1988).

The most important result of these experiments was that measurements of zooplankton biomass alone were poor predictors of juvenile kokanee growth (Fig. 4). Zooplankton biomass varied among depth strata by 400% in Redfish and by 50% in Stanley Lake. In spite of this large variation in zooplankton biomass, growth responses of kokanee within a like were not affected by food availability. These results indicated that comparison of zooplankton biomass among lakes are only relevant to growth of juvenile 0. *nerka* if temperatures occupied by the salmon in each lake are similar.

We believe the model is suitable for making predictions and generalizations concerning juvenile sockeye growthwhentemperature and food availability are known. Both versions of the bioenergetics simulation model did a reasonably good job of predicting observed growth differences of kokanee among net pens.

Our results suggest that the P-value method of relating consumption to food availability produced better predictions of growth. Consequently the P-value model will likely be used for future assessment of juvenile 0. nerka growth potential. The model will be especially useful for ranking the relative growth conditions of different lakes. Model predictions of potential growth among the five historical spawning and rearing lakes of Snake River Sockeye Salmon will allow managers to place emphasis on lakes with a higher growth potential. Thus, the model provides a useful tool for examining management options concerning the rehabilitation of this endangered salmon population.

TABLES

Table 1. Mean temperature over the duration of the experiment (range, °C) and zooplankton biomass ($\mu g/l$) for each net pen in both Redfish and Stanley Lakes. All pens started with an initial density (fish/pen) of 10 fish, the number of survivors at the intermediate sampling are shown in column *Ni* and those survivors at the end of the experiment in column *Nf*.

			Stanley **Lake** (.81 KM^2)				**Redf**ish **Lake** **(6.2 KM^2)**	
Net Pen	Ni	*Nf*	mean temp	mean zoop	Ni	*Nf*	mean temp	mean zoop
epi 1	8	5	7.7-13.9	19.1	10	8	**11.3-15.4**	**5.3**
meta 2	9	8	7.0-11.1	23.3	4	4	**8.6-10-0**	**5.2**
hypo 3	8	8	5.9- 7.3	16.5	7	7	6.7- **9.8**	**5.5**
epi 4	7	5	7.1-13.5	10.3	7	7	**11.3-15.4**	**4.3**
meta 5	9	8	6.4-11.1	11.6	10	6	**10.8-13.6**	**2.9**
hypo 6	9	8	6.2- 7.6	9.6	4	4	**8.0-10.5**	**5.3**

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Chapter 6

Limnological Models of Sockeye Salmon Production in the Sawtooth Valley Lakes

by

Wayne A. Wurtsbaugh and Chris Luecke

INTRODUCTION

The recovery of endangered Snake River sockeye salmon will depend on the limnological characteristics of the Sawtooth Valley lakes and the management decisions made for the lakes and fish. We are now accumulating a good deal of limnological information on these lakes that will help in the decision-making process. We have constructed relatively simple models that synthesize some of this information and make predictions on how management decisions might effect the sockeye salmon populations. The models rely on empirical relationships drawn from a variety of sources. In many cases these relationships are not precisely defined, and consequently our models cannot be expected to give highly accurate predictions. Their primary utility, then, is to help us conceptualize the processes that are driving the population dynamics of the salmon and their habitat, and to make general predictions on how management decisions might effect recovery.

We have constructed two models. The first is a simple empirically-based regression model that predicts salmon production at a given chlorophyll level. The second model integrates the first relationship with the well-known relationship between phosphorus loading and chlorophyll levels to simulate how many sockeye salmon would return to **Redfish** Lake under different scenarios of lake fertilization and improved survival through the migratory coordior. This model also simulates how the returning salmon would effect the productivity of their rearing lakes by bringing nutrients back to the system.

The chlorophyll-salmon model

This empirical model is based on Rieman and Meyers (1992) research cn kokanee salmon and limnological parameters in Idaho Lakes. They used trawling to estimate salmon sizes and abundances in 10 lakes that varied from ultra-oligotrophic to mesotrophic. We used their published information to calculate salmon production in each lake as the product of mean standing stock biomass and growth rates between size classes according to:

$$\sum_{i=1}^{3} [(B_i + B_{i+1})/2] \bullet (\ln W_{i+1} - \ln W_i)/1 \text{ year}$$

where: B_i = Standing stock biomass (density * mean weight) of year class i in the lake.

W_i = Mean weight of fish in year class i. Mean weights were calculated from their fork length data using the following length-weight relationship:
$\log w = -4.803 + 2.886 \text{ Log FL};$

This model provides an approximation of salmon production from age 1 to 3, but it does not include the production of salmon moving from age class 0' to 1, because the densities and sizes of the YOY salmon in the lakes were not available.

The resulting annual production values were plotted against the summer chlorophyll values for each of the lakes. We have shown the production data in both kg ha^{-1} yr^{-1}, and as the equivalent number of smolts assuming that the fish migrate at a size of 10 g (Bjornn et al. 1968). This allows us to predict how many smolts would be produced at a given chlorophyll concentration, assuming that *the current kokanee* production *in these lakes would be* realized *by* sockeye salmon that *would* grow *to smolt size in one year.*

The correlation between chlorophyll concentration and *0. nerka* production was signficant ($p < 0.05$), but there was a good deal of scatter in the data indicating that many additional factors besides phytoplankton abundance will effect fish production in these Idaho lakes. A part of the scatter in the points is also likely due to errors in assessing population abundance and fish sizes in these diverse lakes. Although the chlorophyll-kokanee production relationship was variable, the general increase in fish production with increases in chlorophyll, or other trophic-state indices, is

PREDICTED SMOLTS (100 mm) FOR IDAHO LAKES
BASED ON TOTAL KOKANEE SALMON PRODUCTION
(RIEMAN & MEYERS 1992)

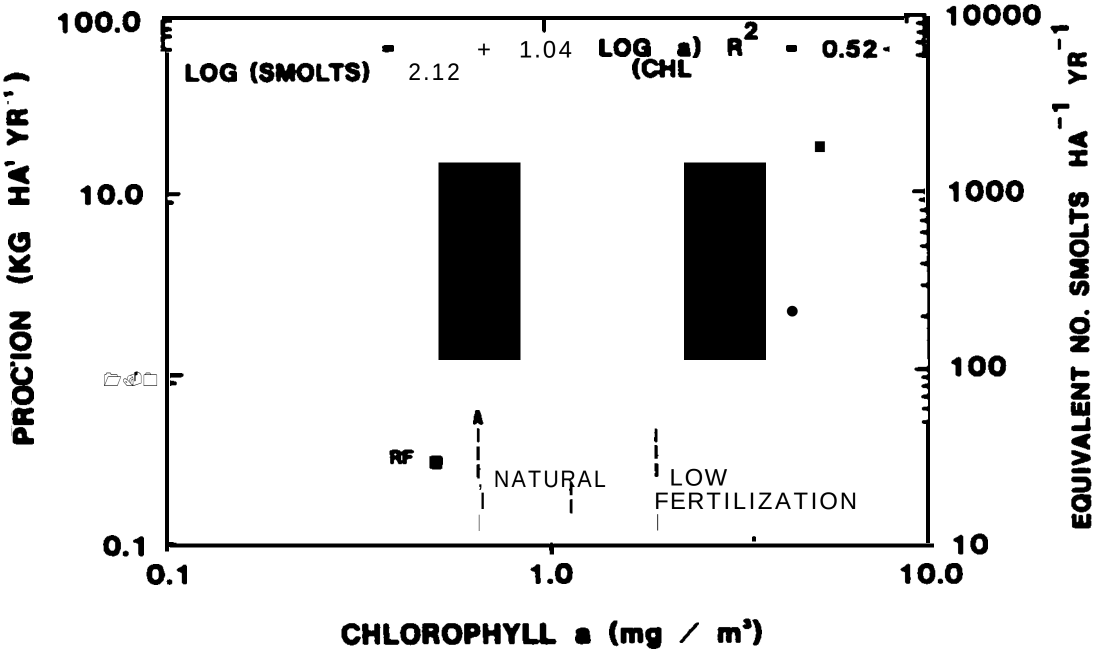

Figure 1. Relationship between mean summer chlorophyll a levels and kokanee salmon production in ten Idaho lakes. The relationship was derived from estimates of fish biomass and the growth rates of salmon in these lakes reported by Rieman and Meyers (1992). The equivalent number of 10-g sockeye salmon smolts that could be produced at given chlorophyll levels is also shown. Arrows indiate natural chlorophyll *a* levels in Redfish Lake, and those resulting from low levels of nutrient additions to Redfish Lake water. "RF" indicates the data point derived for Redfish Lake by Rieman and Meyers (1992).

consistent with many other studies (see Carline 1966; Plante and Downing 1993). The chlorophyll-salmon model predicts that as chlorophyll levels rise from 0.5 to 5 mg m^{-3}, 0. *nerka* production will rise from 0.6 to 7.0 kg $ha^{-1}yr^{-1}$, or in terms of potential smolts, from 64 to 702 fish $ha^{-1}yr^{-1}$.

This relationship can be used to help understand the current production of 0. *nerka* in Redfish Lake and how fertilization might increase fish production. The mean summer chlorophyll concentration in Redfish Lake in 1993 was 0.62 mg m^{-3}, near that reported by Rieman and Myers (1992) when they measured the abundance of 0. *nerka* in the lake. At this chlorophyll level, the model predicts that about 85 smolts ha“, or 52,000 smolts/year could be produced if the current production all went into smolts. If the lake were fertilized at a level similar to that in our "low" limnocorral treatment, chlorophyll levels would rise to near 2 mg m^{-3}, and the model would predict that 0. *nerka* production would be near 255 smolts ha^{-1}, or 158,000 yr^{-1}. The model thus suggests that levels of whole-lake nutrient additions contemplated within species recovery plans have the potential to considerably increase production of stocked sockeye salmon.

It is not clear, however, that this production potential would be realized in the Sawtooth Valley lakes. In field experiments in 1992 and 1993, we assessed the effects of increased nutrient additions on trophic function of Sawtooth Valley Lakes. In two limnocorral experiments we observed how increased nutrient loadings enhanced phytoplankton production, but no increase in zooplankton biomass was observed. Similarly, the increased runoff and nutrient loadings to the lakes in 1993 compared to 1992 resulted in enhanced chlorophyll concentrations, but no increase in zooplankton biomass. Research plans in 1994 need to fully assess the mechanisms of trophic transfer under nutrient additions to determine what factors reduce the ability of zooplankton to respond to these increases in phytoplankton production.

Simulation model

We developed a simulation model to assess the potential number of adult returns expected under a variety of management options. These simulations allowed us to compare the relative effectiveness of different management strategies and to estimate potential historic runs of sockeye salmon prior to construction of the hydroelectric dams on the Columbia and Snake Rivers.

The outline of the model is shown in Figure 2. In the model, annual phosphorus loading drives mean chlorophyll concentration based on the relationships in Vollenweider (1976). When Redfish Lake 1992 discharge, lake area, and mean depth are used in the calculations, chlorophyll concentration ($\mu g/L$) was defined as:

$$\text{Chl a} = 0.0182\,(P / A)^{0.91}$$

where P is phosphorus loading in mg and A is lake area in m^2.

Resultant chlorophyll concentration is used to derive production of age-0 *O.nerka* as calculated from Reiman and Meyers (1992 see above). As age-l smolts outmigrate from Redfish Lake at 9.8 g they reduce P-loading by the phosphorus content in their bodies. Survival of smolts during their downstream migration past eight hydroelectric dams to the ocean was assumed to be 0.0253 (Bowles and Cochnauer 1984). Ocean survival was assumed to be 0.73 per year for 2 years. Upstream survival of adults past the eight dams was assumed to be 0.37 (Bowles and Cochnauer 1984). Adults return at 2.2 kg to Redfish Lake and increase the subsequent year's P-loading by the phosphorus in their body mass (0.35%). This loading was coupled with annual P-loading from streams and used to generate chlorophyll concentration in the subsequent year.

These growth and survival assumptions provide a smolt-returning adult survivorship of 0.0049 and a ratio of smolt biomass leaving the lake to adult biomass returning of 0.0045 (9.8:2200 g). As long as smolt-adult survivorship exceeds the ratio of smolt to adult biomass, the returning adult sockeye salmon can increase the nutrient loading of the lake. Under the baseline simulations, this nutrient enhancement occurs but at exceedingly small rates.

SOCKEYE-NUTRIENT MODEL

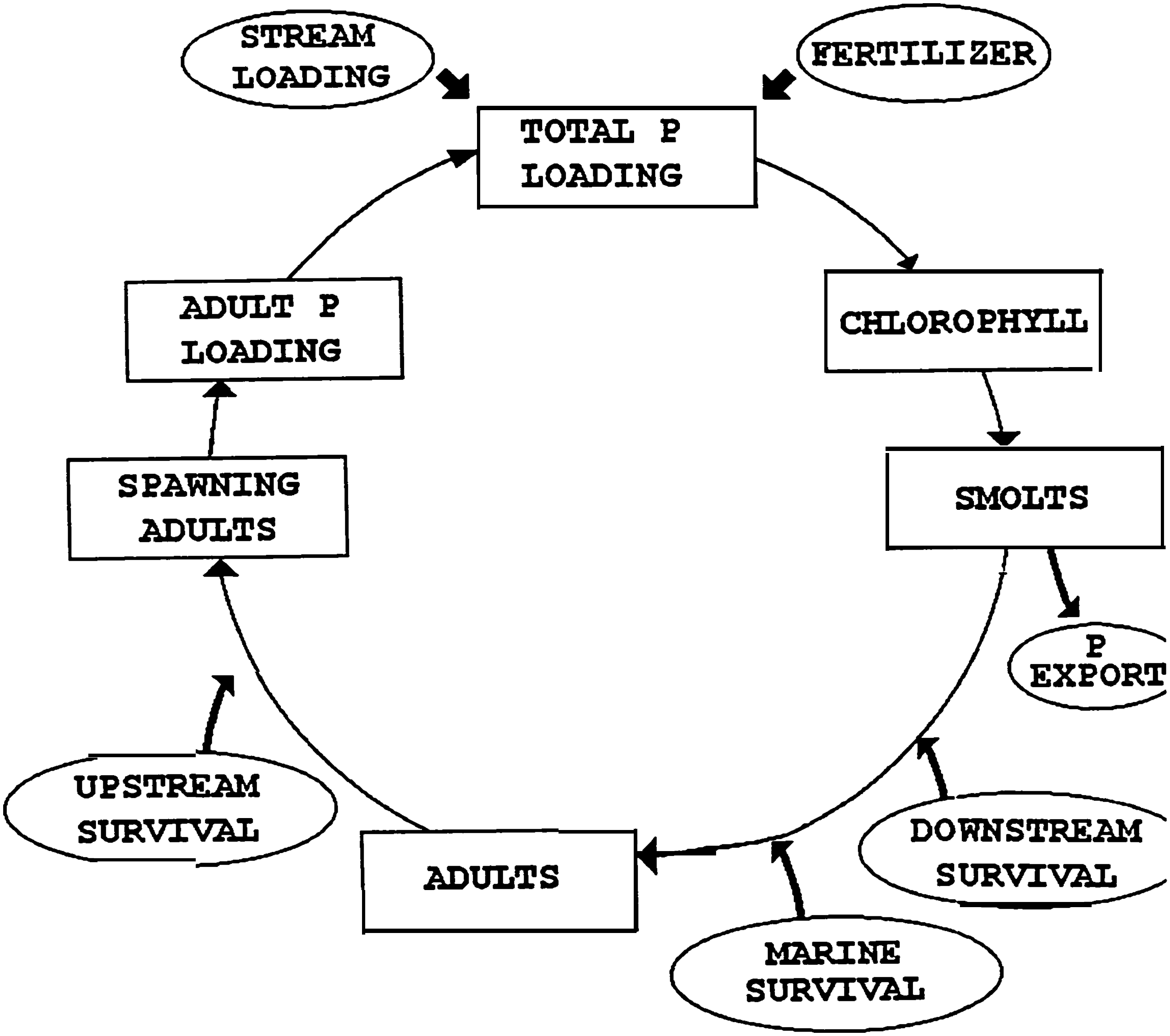

Figure 2. Diagram of the simulation **model** in which consideration of phosporus inputs and outputs allowed comparison of the number of adult **sockeye salmon** returning to **Redfish** Lake under different survival and lake fertilization scenarios.

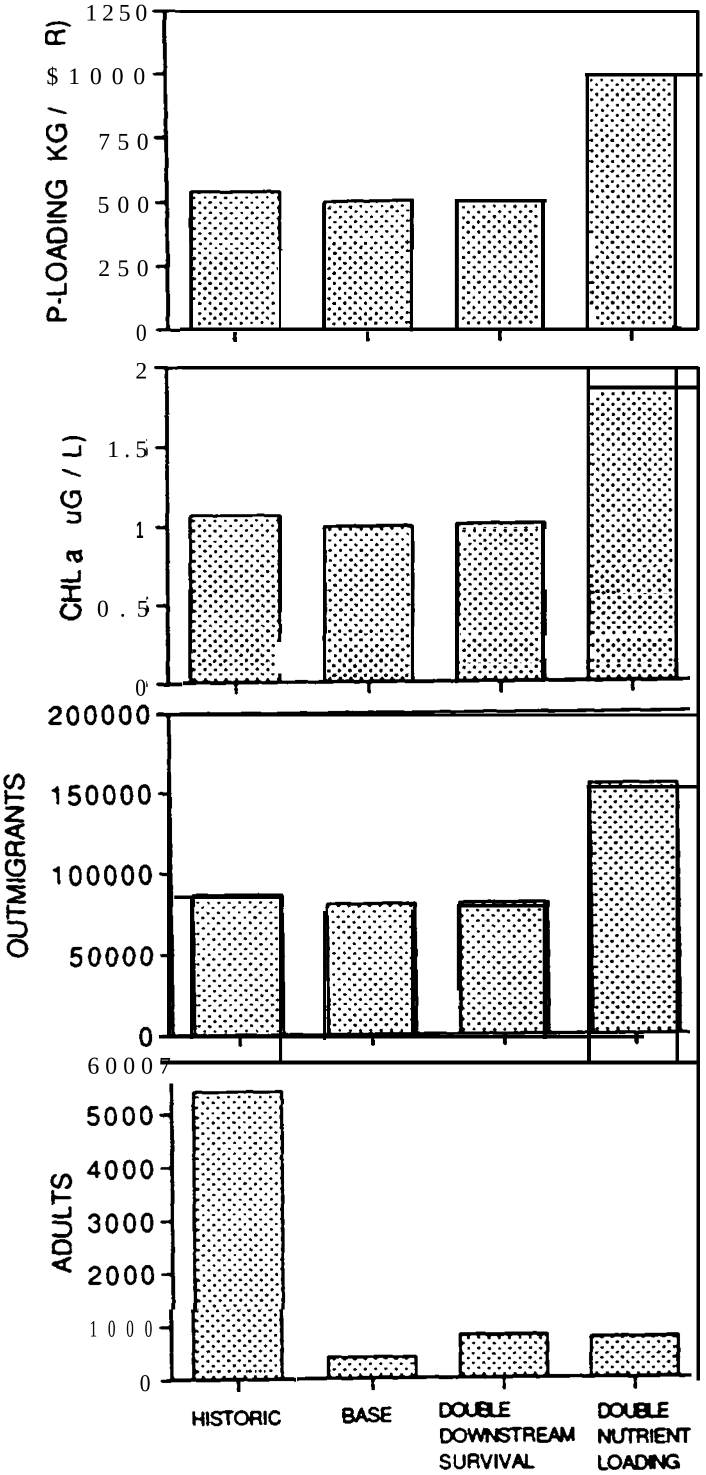

Figure 3. Equilibrium conditions of simulation model of **sockeye** salmon production and survival in **Redfish** Lake. Simulation runs included historic survival estimates before **mainstem** dams were constructed, present day baseline conditions, doubling of survival during sownstream migration, **and** double nutrient loadings with lake fertilization. In all cases, equilibrium conditions were reached within 12 years.

Five management scenarios were simulated with this model. A base run was conducted in which P-loadings from streams were assumed to be 500 kg annually and survival rates stated above. This base run represents current conditions with no lake fertilization and no improvements in survival frommodifications in dam operations. A second scenario simulated sockeye production when survival of smolts migrating downstream was doubled to 0.0506, representing improvements in **mainstem** passage. A third scenario was simulated in which P-loading was doubled, representing the potential impacts of a lake fertilization program. A fourth scenario simulated sockeye production under historic estimates of sockeye survival (0.13 for downstream migration, 0.9 for upstream migration) representing the time before the 8 dams were constructed. A fifth scenario simulated effects of a three year program of lake fertilization with no improvements in outmigrant survival.

Equilibrium values for total annual P-loading, mean annual chlorophyll concentration and number of returning adults were plotted for each of the first four management scenarios (Figure 3). P-loading and chlorophyll were highest under the lake fertilization simulation. The base-line simulations indicated that present **lake** production and sockeye survival estimates could produce approximately 400 returning adult salmon. Doubling nutrient loading or doubling downstream survival of smolts had similar effects on the number of returning adult salmon, increasing these values by approximately a factor of two. Doubling downstream survival more than doubled the number of returning adults in that these additional returning adults increased P-loading and enhanced chlorophyll concentrations slightly. The number of returning adult salmon was greatest in simulations where historic survival estimates were used. This simulation indicated that approximately 5500 adults could have returned to **Redfish** Lake before the dams were constructed on the Columbia and Snake Rivers. These returning salmon allowed average P-loadings and resultant chlorophyll concentrations to increase approximately 10% over baseline conditions.

In simulations where P-loading was doubled for three years, chlorophyll concentrations, sockeye outmigrants and returning adult sockeye increased in a manner similar to the previously described simulations of lake fertilization. After fertilization is stopped, these values return to base conditions within eight years, assuming a water residence time of three years. Results of these simulations suggest that a short-term lake fertilization program would have little impact on sockeye recovery without large improvements in sockeye survival rate.

Results of our simulations for Redfish Lake are qualitatively similar to Bowles and Cochnauer (1984) estimates of sockeye production in Alturas Lake. In both models variation in survival had a much greater effect on adult returns than did effects of fertilization. Both models indicate that improvements in mainstem passage survival has a greater effect on adult escapement than did variations in rearing lake environments. Bowles and Cochnauer (1984) simulations indicate that maximum adult escapement in Alturas Lake would not exceed 35 adults per hectare, whereas our simulations suggest that potential adult escapement to Redfish Lake would be limited to 10 adults per hectare. These differences derive from assumptions concerning the relationship between lake production and other components of the fish assemblage. Bowles and Cochnauer assume that kokanee populations would remain similar to previous conditions, but that piscivorous fish could be reduced substantially. Our simulations examine the potential sockeye production if no kokanee were present, but piscivores were present in abundances similar to that of other Idaho lakes.

The simulation model results provide three major conclusions. First, that historic runs of sockeye salmon adults to Redfish Lake may not have greatly exceeded numbers reported from the 1950's (Bjornn et al. 1968). Secondly, continued accumulation of additional marine-based nutrients from decomposition of returning adult salmon likely had small effects on lake nutrient budgets given the relatively high natural mortality rates associated with this long-distance migrant population. And lastly, that improved survival of Redfish Lake sockeye salmon had greater effects on sockeye production than did short-term increases in nutrient concentrations with a lake fertilization program.

Discussion

The simulation model, based on relationships among phosphorus loading, chlorophyll, and kokanee production (Rieman and Myers 1992), indicated that low levels of nutrient fertilization in Redfish Lake would have a relatively small impact on chlorophyll concentrations due to flushing rates and the departure of nutrients with outmigrating salmon. In these simulations, the enhanced production of juvenile 0. nerka that occurs with lake fertilization is largely lost because of high mortality of smolts during the downstream migrations. Greatest long-term production of sockeye salmon occurred in simulations where lake fertilization was coupled with improvements in smolt survival.

The relationship between chlorophyll concentration and production of 0. *nerka* in Idaho lakes (Rieman and Meyers 1992) indicates that ambient chlorophyll levels in the Sawtooth Valley Lakes could produce substantial numbers of sockeye smolts. In fact, production rates of 0. nerka in Redfish Lake calculated from their data suggested that 0.34 kg ha^{-1} was produced in 1990. This was primarily kokanee production. This level of production is identical to the mean annual biomass of sockeye salmon smolts migrating from Redfish Lake from 1956-1966 when fish migrations were monitored by Bjornn et al. (1968). The exact correspondance of current 0. *nerka* production (largely kokanee) and past sockeye smolt production is probably coincidental, as Bjornn et al.'s data indicated a lo-fold range in the biomass of smolts leaving the lake in a given year. Nevertheless, the data does suggest that a considerable proportion of production that once yielded smolts, now goes into kokanee production.

The construction of eight dams on the Snake and Columbia Rivers during the past 50 years has likely increased the abundance of kokanee in these lakes. Increased mortality imposed on anadromous strains of 0. nerka during this period caused decreases in the numbers of adult sockeye returning from the sea. With fewer progeny from anadromous adults entering the lakes, the natural production potential likely was subsumed by kokanee populations. Gross (1987) has provided a conceptual model of how changes in mortality or growth during various stages of a fish's life, can favor anadromy or non-anadromous life-history strategies. These conclusions suggest that reductions in the recruitment success of kokanee populations, may facilitate the return the system to pre-1900 conditions and re-establish runs of anadromous sockeye salmon in the lakes.

Ultimately, to restore the success of the anadromous life history, increased survival of juvenile and adults during mainstem passage must be achieved. Efforts to enhance production of juvenile sockeye salmon in the rearing lakes can work only as stopgap measures to eliminate further erosion of this endangered population. The potential successes of a broodstock program, lake fertilization, or reduction of competitors or predators will be tempered by problems facing sockeye salmon smolts after they leave the rearing lakes and negotiate the dams on the Snake and Columbia Rivers. Progress on mainstem passage problems is required to allow self-sustaining populations of anadromous sockeye salmon to exist in Sawtooth Valley Lakes.

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Chapter 7

General Discussion and Management Implications

Chris Luecke **and** Wayne Wurtsbaugh

Results from the sampling and field experiments conducted in Sawtooth Valley lakes and watersheds in 1993 provide five general conclusions that have special relevance for recovery efforts of endangered sockeye salmon. These conclusions relate to inter-annual climate variation, the productivity of deep chlorophyll layers, effects of nutrient additions on lake productivity, effects of temperature and zooplankton variation on growth of juvenile 0. *nerka,* and overall fish production in the lakes.

Climate effects on nutrient loading and temperature

Measurements of the hydrology and limnology of Sawtooth Valley lakes and streams in 1992 and 1993 allow us to compare effects of variable precipitation on lake structure and function. 1992 was the last year of a severe drought, providing the lowest runoff recorded for several decades. The end of the drought in 1993 provided an average runoff year that resulted in both flows of stream water and loading of nitrogen and phosphorus that were 2-3 times the values of 1992. This increased nutrient loading resulted in increased phosphorus and nitrogen concentrations of lake water during the spring overturn period. Increased nutrient concentrations in spring and continued nutrient loading from streams throughout the summer allowed phytoplankton biomass to accrue more rapidly in all of the lakes in 1993 compared to 1992. This increase in phytoplankton occurred in spite of lower surface illumination and cooler temperatures in 1993.

Although chlorophyll concentrations were higher in 1993 compared to 1992, biomass of crustacean zooplankton in the lakes in 1993 was similar or even lower than biomass in 1992. The delayed development of summer zooplankton biomass in several of the lakes suggests that cooler water temperatures may have reduced the ability of crustacean populations to respond to larger amounts of available food resources. The biomass of crustacean zooplankton was strikingly reduced in Alturas Lake in 1993 compared to 1992. The increased abundance of small planktivorous fish as estimated with acoustic surveys likely caused the collapse of these zooplankton populations. Ambient fish densities in the other lakes coupled with cool water temperatures likely kept crustacean zooplankton populations from responding to the increases in phytoplankton biomass observed in 1993.

Productivity of deep chlorophyll layers

Limnological observations from 1992 indicated that the majority of the chlorophyll present in the water column occurred below the epilimnion of each lake. At the time we speculated that phytoplankton present in these colder darker regions of the lakes might not be contributing much to primary production. Measurement of primary production in each of the lakes in 1993, however, indicated that phytoplankton populations in the metalimnia were growing and fixing carbon at rates not too different from epilimnetic phytoplankton. Indeed most of the primary production in each of the lakes occurred below the epilimnion.

These results suggest two points that impinge on future management activities. First, although the concentration of primary producers in the lakes are low, they are distributed through a long water column where light levels are adequate for photosynthesis, and this may allow for higher levels of carbon fixation than might be expected based on epilimnetic chlorophyll levels. Nevertheless, integral rates of primary production in the lakes were still extremely low. Secondly, these deep chlorophyll layers are providing production to regions of the lakes where juvenile sockeye salmon would likely congregate.

Nutrient budgets, nutrient additions to limnocorrals, and the potential for lake fertilization.

The nutrient budget analyses for the lakes in 1992 indicated that most nutrients enter the lake via streamflow and precipitation to the lake surface. Returning sockeye salmon may have contributed substantial amounts of nitrogen and phosphorus to the lakes, although the relative contribution is likely to be less than has been found in some other lakes. The loss of this nutrient source may now make the Sawtooth Valley lakes less fertile than they were under pristine conditions.

The limnocorral experiments clearly demonstrated that nutrient additions would stimulate production of phytoplankton and periphyton in the lakes, and egg production in cladocera. Low levels of nutrient additions to epilimnetic waters more than doubled chlorophyll concentrations and primary production in Redfish Lake and Pettit Lakes. High nutrient additions further increased chlorophyll concentrations, but also reduced water transparency. Higher abundances of phytoplankton allowed cladoceran zooplankton to produce more eggs in limnocorrals that received nutrient additions.

Results of the Pettit Lake limnocorral experiment indicated that additions of nitrogen and phosphorus to the metalimnion of the lakes would have a similar stimulating effect on phytoplankton populations as would epilimnetic nutrient additions, and would have a reduced effect on decreases in water transparency. Whole water column chlorophyll, phytoplankton biovolume, primary production and cladoceran egg production were all greater in the metalimnetic nutrient additions compared to epilimnetic treatments. These increases in measures of lake productivity were accompanied by generally greater Secchi depth transparency in metalimnetic compared to epilimnetic nutrient additions. These results suggest that a metalimnetic fertilization would likely be more benefical than would an epilimnetic fertilization. Additionally, if the lake surface were fertilized, it would stimulate algal growth in the epilimnion and this would shade likely shade the deep chlorophyll layer and curtail production there.

Although nutrients stimulated algal production, water transparencies in the fertilized treatments was generally similar to those occuring in the lakes during the spring and fall, and were greater than transparencies of many lakes used for fishing and recreation in Idaho (Rieman and Meyers 1992). Consequently, the productivity of the lakes can likely be substantially increased without compromising recreation use in the area.

Although out 1992 and 1993 experiments clearly demonstrated a positive effect of nutrients on algal production, we have not yet shown that this production is transfered to zooplankton and salmon. Several factors may have confounded our results and made it difficult to detect any increases in zooplankton and fish growth. In the Redfish Limnocorral experiment, fish densities and the resultant predation pressure on the plankton may have been too great to allow zooplankton populations to respond, despite the higher egg production of the cladocera in the fertilized treatments.

This hypothesis is supported by the inverse relationship between kokanee densities in the corrals and their growth, and by the negative correlation of zooplankton biomass with the density of age-0 kokanee remaining in the limnocorrals at the end of the experiment. Limnocorrals with high fish density contained low zooplankton biomass. Planktivory by these juvenile kokanee may have had a greater effect on zooplankton population dynamics than did effects of increased food resources. This effect could be particularly powerful if the fish preyed selectively on the larger egg-bearing cladocerans. These results indicated that at fish densities present in the Redfish limnocorrals (approximately twice the fish biomass present in the upper 30 m of the lake) cladocerans populations may not be capable of increasing in response to nutrient additions.

With the high variability between replicate treatments, it was not possible to determine if zooplankton biomass in the Pettit Lake limnocorrals responded to nutrient additions. Variable and unfortunately unknown abundances of planktivorous fish may have reduced the ability of zooplankton populations to respond to increased levels of algal biomass as we have suggested occurred in Redfish Lake limnocorrals. The relatively high densities of large-bodied cladocerans, however, suggest that levels of fish planktivory were low during the Pettit limnocorral experiment.

These observations suggest that factors other than fish and phytoplankton biomass could regulate zooplankton biomass in Pettit Lake. One possibility is that the phytoplankton taxa stimulated the most by nutrient additions were inedible to the dominant zooplankton taxa present. Examination of phytoplankton taxa in the different treatments indicated that most of the increased biovolume occurred in diatoms. Given that the diatoms we observed are suitable food for cladocerans (Infante and Litt 1985), we conclude that the lack of response of zooplankton biomass to nutrient additions was not entirely due to increases in inedible or toxic algae. For whatever reasons, the lack of response of zooplankton biomass to nutrient addition in both the Pettit and Redfish limnocorral experiments suggests that using nutrient additions to enhance zooplankton may be problematic in Sawtooth Valley lakes. Additional experiments planned for the summer of 1994 should help to clarify this question.

Temperature and zooplankton effects on fish growth

Juvenile 0. *nerka* placed in experimental net pens in Redfish and Stanley Lakes exhibited highly variable rates of growth. In the low food environment of Redfish Lake, growth was slow and water temperature had little effect on fish growth. In the high food environment of Stanley Lake, growth rate was generally higher and water temperature had a pronounced effect on fish growth. Juvenile fish grew much more rapidly in warm epilimnetic waters compared to meta- and hypolimnetic waters.

These results may partially explain the depth distributions and growth patterns of kokanee in the lakes. In poor food environments, like Alturas and Redfish Lakes, zooplankton food resources are limiting and thus depth (and correspondingly temperature) distribution has less effect on a fish's growth potential. In these lakes fish may remain in deep water during both day and night periods without sacrificing growth. Alturas Lake kokanee may be one example of this type of response in that nighttime acoustic targets were most dense at 20-25 m where water temperatures were 5°C. In high food environments, like Pettit and Stanley Lakes, depth distribution would have a large effect on potential growth of juvenile 0. *nerka* as temperature appears to limit fish growth. These results may explain the higher densities of fish present in epi- and meta-limnetic waters in these two lakes.

The interactions of temperature and available zooplankton food resources on growth of juvenile 0. *nerka* suggest that management strategies for stocked sockeye salmon should differ among lakes. In lakes with low zooplankton biomass, efforts to enhance zooplankton production (like lake fertilization) should result in increased production of juvenile sockeye salmon. In a system with high zooplankton food resources, like Stanley Lake, enhancements of zooplankton production may have little effect on sockeye growth if the presence of piscivores restricts the distribution of juvenile sockeye to colder waters. Reductions of piscivore density might have a greater effect on individual growth and cohort production in these lakes.

The reasonable correspondence between the growth of juvenile kokanee in net pens and predictions from the simulation model indicated that this model can be used as a management tool to assess production potential of introduced sockeye salmon in Sawtooth Valley Lakes. The strong interaction between temperature and food availability in both field and model results indicates that these two limnological parameters will be most important in assessing habitat suitability of the rearing lakes for juvenile sockeye salmon.

Fish production among the **lakes**

Empirical and simulation models suggest that the Sawtooth Valley lakes could support a substantial biomass of juvenile sockeye salmon. Presently much of that production potential is used to support kokanee populations. Results from simulation models indicate that lake fertilization could increase the production potential of the lakes, but that the increase would be short-lived once lake fertilization ceased given the relatively low water residence times in the lakes and the low potential for marine nutrients to contribute substantially to the lakes' nutrient income. The present low smolt-adult survival rates indicates that the nutrient loss due to outmigrating sockeye salmon exceeds the amount of nitrogen and phosphorus that enters the lakes with returning adult salmon. Salmon would only be able to enrich their rearing environments if survival rate greatly exceeds the ratio of smolt:adult weight. The higher pre-dam estimates of smolt-adult survival allow for some net accumulation of nutrients from the ocean, but results from the simulation model suggest that this increase in nutrients would only be on the order of 10%. These results suggest that Redfish Lake historically supported a population of sockeye salmon that was not more- than an order of magnitude greater than numbers of returning adults reported from the 1950's, with lake nutrient and chlorophyll concentrations that were not dramatically different from present day values.

Appendix 1.

Nutrient analyses of lakes and streams

Lake Redfish

All values are in ug/L, except TN TP ratio, which is by weight
TN = NO3-N + TKN

Date	Depth	TP	SRP	NO3-N	NH4-N	TKN	TN	TN TP
10-May-92	0-6m	6		2		64	66	11
31-May-92	0-6m	7	<2 0	9		47	56	8
12-Jun-92	0-6m	7	2 4	3		32	35	5
29-Jun-92	0-6m	7	<2 0	2		29	31	4
09-Jul-92	0-6m	6	2 7	5		38	43	7
22-Jul-92	0-6m	4	<2 0	7		20	27	7
01-Aug-92	0-6m	3	<2 0					
01-Aug-92	20m	3	<2 0					
01-Aug-92	75m	32	<2 0					
11-Aug-92	0-6m	6	<2 0	11		66	77	13
25-Aug-92	0-6m	25	4 2	6		63	69	3
08-Sep-92	0-6m	10	<2 0	11		45	56	6
18-Sep-92	0-6m	10	<2 0	12		45	57	6
11-Oct-92	0-6m	8	2 7	3		31	34	4
18-Nov-92	0-6m	5	<2 0	9		34	43	9
21-Mar-93	0-6m	16	<2 0	10	3	73	83	5
21-Mar-93	0m	19						
21-Mar-93	10m	5	<2 0	5	15	53	58	12
21-Mar-93	50m	12	<2 0					
21-Mar-93	80m	6	<2 0	30	9	59	89	14
16-May-93	0-6m	10		8		55	63	6
16-May-93	30m	7		5		43	48	7
16-May-93	70m	8		7		40	47	6
01-Jun-93	0-6m	12		0		68	68	6
10-Jun-93	0-6m	*	4 3	1	1	80	81	
10-Jun-93	15m	5	<2 0	0	4			
10-Jun-93	22m	3	<2 0	1	0	55	56	17
10-Jun-93	30m	4	<2 0	9	47			
10-Jun-93	40m	4	<2 0	1	5			
10-Jun-93	50m	4	<2 0	0	4			
10-Jun-93	60m	4	<2 0	1	8			
10-Jun-93	82m	24	<2 0	17	8	66	83	4
21-Jun-93	0-6m	9	<2 0	1	0	67	68	7
09-Jul-93	0-6m	7	<2 0	2	3	53	55	8
09-Jul-93	22m	5	<2 0	1	0	65	66	13
09-Jul-93	88m		<2 0	8	4			
23-Jul-93	0-6m	4		1		63	64	16
06-Aug-93	0-6m	3	<2 0	5	10	63	68	20
06-Aug-93	24m	4	<2 0	2	1	64	66	15
06-Aug-93	84m	2	<2 0	246	10	53	299	171
03-Sep-93	0-6m	6	<2 0	1	1	47	48	7
03-Sep-93	24m	7	<2 0	1	1	32	33	5
03-Sep-93	84m	5	<2 0	10	15	71	81	15
15-Sep-93	0-6m	5		1		66	67	14
08-Oct-93	0-6m	5	<2 0	1	4	68	69	15
08-Oct-93	31 5m	4	<2 0	0	2	29	29	7
08-Oct-93	86m	17	<2 0	46	14	59	105	6
07-Nov-93	0-6m	5		1		31	32	6
07-Nov-93	26 5m	5		1		34	35	6
04-Dec-93	0-6m	5		38		61	99	22
04-Dec-93	85m	6				36		

stream: Fishhook Creek

All values are in ug/L, except TN TP ratio.
which is by weight.
TN = NO3-N + TKN

Date	TP	NO3-N	TKN	T N	TN.TP
26-Mar-92	8				
11 -May-92	7	30	52	82	12
20-May-92	19	38			
25-May-92	10	18	78	96	10
31 -May-92	12	23			
12-Jun-92	14	22			
19-Jun-92	9	8	31	39	4
29-Jun-92	15	18	102	120	8
09-Jul-92	8	14	42	56	7
21-Jul-92	9	11	44	55	8
10-Aug-92	9	11	60	71	8
24-Aug-92	15	5	60	65	4
11-Sep-92	12	5	56	61	5
13-Sep-92	18	1			
1 s-sop-92	19	2			
18-Sep-92	19	2			
22-Sep-92	11	3	48	51	5
25-Sep-92	17	1			
1 O-ml-92	9	4	51	55	6
18-Nov-92	9	23	52	75	8
19-Apr-93	11	8	82	90	6
03-May-93	19	7	97	104	5
12-May-93	42	14	425	439	10
16-May-93	31	27	256	263	9
17-May-93	159	18	1499	1517	10
22-May-93	14	25	23	48	4
01 Jun-93	5	24	170	194	41
08-Jun-93	11	23	102	125	12
20-Jun-93	47	12	205	217	5
22Jun-93	17	13	172	185	11
01 Jul-93	8	13	50	63	8
14-Jul-93	12	11	39	50	4
23-Jul-93	3	8	119	127	44
01-Aug-93	7	13	75	88	13
08-Aug-93	5	9	59	68	15
14-Aug-93	13	12	95	107	8
22-Aug-93	9	16	56	72	8
28-Aug-93	4	18	75	93	26
04-Sep-93	8	14	114	128	17
11-Sep-93	5	15	58	73	14
18-w-93	10	13	46	59	6
08-Oct-93	8	6	34	40	5
06-Nov-93	10	36	34	70	7

Stream: Redfish Lake Creek Inflow

All values are in ug/L, except TN:TP ratio,
which is by weight.
TN = NO3-N + TKN

Date	TP	NO3-N	TKN	TN	TN:TP
26-Mar-92	7				
13-May-92	6	32	21	53	9
25-May-92	9	43	21	64	7
31-May-92	6	44			
12-Jun-92	5	49			
19-Jun-92	9	19	38	57	6
29-Jun-92	39	35	130	165	4
09-Jul-92	7	42	21	63	9
22-Jul-92	8	33	25	58	7
01-Aug-92	19	44			
10-Aug-92	7	37	19	56	8
24-Aug-92	9	34	18	52	6
13-Sep-92	7	24	12	36	5
18-Sep-92	7	36	6	42	6
11-Oct-92	6	31	25	56	9
18-Nov-92	7	66	2	68	10
21-Mar-93	7	44	15	59	9
19-Apr-93	4	50	12	62	16
16-May-93	13	28	103	131	10
22-May-93	5	32	97	129	25
01-Jun-93	7	33	75	108	15
07-Jun-93	9	31	46	77	9
10-Jun-93	10	31	44	75	7
14-Jun-93	7	35	44	79	12
23-Jun-93	10	35	44	79	8
09-Jul-93	5	6	70	76	16
10-Jul-93	3	49	50	99	34
17-Jul-93	3	39	31	70	23
23-Jul-93	7	30	39	69	10
01-Aug-93	9	45	57	102	12
08-Aug-93	4	34	54	88	25
16-Aug-93	6	23	63	86	14
22-Aug-93	6	49	42	91	15
28-Aug-93	9	45	24	69	8
05-Sep-93	5	39	106	145	30
14-Sep-93	2	40	24	64	27
08-Oct-93	4	47	21	68	16
07-Nov-93	5	81	11	92	19

Lake: Alturas

All values are in ug/L, except TN:TP ratio, which is by weight.
TN = NO3-N + TKN

Date	Depth	TP	SRP	NO3-N	NH4-N	TKN	TN	TN:TP
09-May-92	0-6m	10	2.8	2		72	74	7
30-Jun-92	0-6m	7	<2.0	2		55	57	8
20-Jul-92	0-6m	9	<2.0	14		70	84	9
30-Jul-92	0-6m	6	<2.0	2				
30-Jul-92	13m	96	<2.0	4				
30-Jul-92	48m	45	<2.0	2				
10-Aug-92	0-6m	7	<2.0	1		89	90	12
19-Sep-92	0-6m	7	<2.0	3		78	81	12
10-Oct-92	0-6m	9	<2.0	4		79	83	9
22-Mar-93	0-6m	7	2.0	12	66	74	86	12
22-Mar-93	10m	7	2.9	9	32	49	58	8
22-Mar-93	46m	8	3.5	35	4	27	62	8
17-May-93	0-6m	15		7		80	87	6
17-May-93	25m	9		3		69	72	8
17-May-93	45m	7		2		47	49	8
23-May-93	0-6m	7		1		81	82	12
09-Jun-93	0-6m	10	2.0	2	1	77	79	8
09-Jun-93	10m	7	<2.0	1	6			
09-Jun-93	18m	6	<2.0	2	7	86	88	16
09-Jun-93	25m	6	<2.0	0	5			
09-Jun-93	35m	8	<2.0	1	8			
09-Jun-93	45m	14	2.4	21	1	68	89	7
23-Jun-93	0-6m	14		2		170	172	12
08-Jul-93	0-6m	8	<2.0	10	3	100	110	14
08-Jul-93	18m	7	<2.0	60	2	93	153	22
08-Jul-93	45m	8	2.6	52		53	105	13
22-Jul-93	0-6m	9		1		73	74	8
05-Aug-93	0-6m	7	<2.0	2	3	67	69	10
05-Aug-93	24m	8	<2.0	2	1	21	23	3
05-Aug-93	45m	13	2.0	22	2	47	69	5
02-Sep-93	0-6m	5	<2.0	1	2	54	55	11
02-Sep-93	21m	6	<2.0	1	2	61	62	11
02-Sep-93	48m	7	3.7	5	1	70	75	11
16-Sep-93	0-6m	7				50		
09-Oct-93	0-6m	5	<2.0	1	4	130	131	27
09-Oct-93	23m	6	<2.0	1	4	82	83	13
09-Oct-93	45m	6	<2.0	1	3	66	67	11
06-Nov-93	0-6m	7		1		94	95	13
06-Nov-93	20m	6		1		76	77	12
06-Nov-93	47m	8		1		73	74	9
05-Dec-93	0-6m	8		1		57	58	8
05-Dec-93	16m	9		1		82	83	9
05-Dec-93	47m	7		1		55	56	8

Stream: Alturas Inflow

All values are in ug/L, except TN:TP ratio,
which is by weight.
TN = NO3-N + TKN

Date	T P	NO3-N	TKN	TN	TN:TP
22-Mar-93	6	24	70	94	15
23-Apr-93	8	24	48	72	9
04-May-93	34	18	80	98	3
13-May-93	27	20	194	214	8
17-May-93	29	15	154	169	6
23-May-93	17	13	96	109	6
01 -Jun-93	9	7	160	167	19
13-Jun-93	5	4	44	48	10
23-Jun-93	10	6	54	60	6
01 -Jul-93	8	4	54	58	7
10-Jul-93	6	3	58	61	10
13-Jul-93	5				
17-Jul-93	7	10	21	31	4
23-Jul-93	6	4	57	61	10
01 -Aug-93	8	11	49	60	7
08-Aug-93	5	6	47	53	10
14-Aug-93	8	8	38	46	6
21 -Aug-93	4	6	35	41	10
28-Aug-93	9	11	40	51	6
04-Sep-93	5	10	101	111	24
11 -Sep-93	5	15	57	72	14
18-Sep-93	5	14	30	44	10
04act-93	5	11	33	44	9
06-Nov-93	7	30	25	55	8

Lake Pettit

All values are in ug/L. except TN TP ratio,which is by weight
TN = NO3-N + TKN

Date	Depth	TP	SRP	NO3-N	NH4-N	TKN	TN	TN TP
09-May-92	0-6m	9	<2.0	3		84	87	10
30-May-92	0-6m	4	<2 0	11		91	102	27
30-Jun-92	0-6m	4	<2 0	4		63	67	19
30-Jul-92	0-6m	5	<2 0	2		94	96	21
30-Jul-92	11m	4	<2 0	2				
30-Jul-92	47m	10	<2 0	20				
10-Aug-92	0-6m	7	2 9	4		82	86	12
19-Sep-92	0-6m	8	2 2	4		84	88	11
10-Oct-92	0-6m	6	3 9	4		82	86	15
19-Mar-93	0-6m	3						
19-Mar-93	0m	5	<2 0	21	39	110	131	28
19-Mar-93	2om	3	2 0	9	79	101	110	43
19-Mar-93	46m	14	7 2	19	253	231	250	17
17-May-93	0-64-n	9		5		104	109	12
17-May-93	25m	2		3		76	79	46
17-May-93	45m	6		58		74	132	24
23-May-93	0-6m	7						
09-Jun-93	0-6m	10	<2 0	3	4	97	100	10
09-Jun-93	15m	4	<2 0	0	4	90	90	25
09-Jun-93	22m	3	<2 0	10	14			
09-Jun-93	30m	4	<2.0	0	3			
09-Jun-93	40m	6	<2 0	0	4			
09-Jun-93	47m	5	<2 0	55	106	136	191	36
23-Jun-93	0-6m	6		2		45	47	8
08-Jul-93	0-6m	4	·	3	4	80	83	23
08-Jul-93	17 5m	4	·	21	1	79	100	27
08-Jul-93	45m	14	<2 0	44	12	75	119	8
22-Jul-93	0-6m	7		1		66	67	10
05-Aug-93	0-6m	7	<2 0	2	2			
05-Aug-93	24m	6	<2 0	3	2			
05-Aug-93	45m	5	<2 0	59	69			
02-Sep-93	0-6m	3	2.6	1	2	38	39	14
02-Sep-93	23m	2	<2 0	1	1	61	62	29
02-Sep-93	45m	6	<2.0	89	16	104	193	30
16-Sep-93	0-6m	4		0		92	92	24
16-Sep-93	48m	12						
09-Oct-93	0-6m	9	2.0	1	3	65	66	8
09-Oct-93	25m	3	<2.0	1	2	66	67	22
09-Oct-93	45m	6	<2.0	2	59	149	151	25
06-Nov-93	0-6m	6		5		162	167	28
06-Nov-93	24m	5		0		85	85	16
06-Nov-93	46m	11		5		278	283	25

Stream: **Pettit North Inflow**

All values are in ug/L, except TN:TP ratio,
which is by weight.
TN = NO3-N + TKN

Date	T P	NO3-N	TKN	TN	TN:TP
28-Apr-93	13	60	53	113	9
05-May-93	10	26	52	78	8
15-May-93	9	22	117	139	16
23-May-93	5	20	85	105	22
01 -Jun-93	7	23	21	44	6
08-Jun-93	8	13	70	83	11
16-Jun-93	6	15	78	93	15
23-Jun-93	8	21	91	112	15
01 -Jul-93	3	17	54	71	24
10-Jul-93	3	9	79	88	33
17-Jul-93	3	12	91	103	34
23-Jul-93	6	10	94	104	18
01 -Aug-93	4	17	48	65	18
08-Aug-93	2	18	41	59	27
14-Aug-93	3	12	88	100	33
21 -Aug-93	2	13	67	80	33
28-Aug-93	10	22	61	83	8
04-Sep-93	13	22	76	98	8
11-Sep-93	4	28	87	115	32
18-Sep-93	3	25	56	81	25
09-Oct-93	6	18	60	78	13
06-Nov-93	5	64	48	112	25

Stream: Pettit South Inflow

All values are in ug/L, except TN:TP ratio,
which is by weight.
TN = NO3-N + TKN

Date	T P	NO3-N	TKN	TN	TN:TP
28-Apr-93	6	52	21	73	12
05-May-93	10	48	67	115	11
15-May-93	8	14	116	130	17
23-May-93	5	17	78	95	21
01 -Jun-93	4	16	56	72	19
08-Jun-93	4	10	87	97	26
16-Jun-93	3	12	45	57	19
23-Jun-93	3	15	68	83	28
01 -Jul-93	4	14	97	111	31
10-Jul-93	5	9	71	80	17
17-Jul-93	4	12	47	59	15
23-Jul-93	4	9	91	100	26
01 -Aug-93	3	10	110	120	39
08-Aug-93	3	12	115	127	46
14-Aug-93	2	16	42	58	27
21 -Aug-93	2	14	66	80	52
28-Aug-93	4	16	60	76	17
04-Sep-93	11	22	83	105	9
11-Sep-93	6	23	104	127	21
18-Sep-93	3	16	13	29	10
09-Oct-93	6	18	111	129	22
06-Nov-93	4	48	12	60	15

Lake Stanley

All values are in ug/L except TN TP ratio which is by weight
TN = NO3-N + TKN

Date	Depth	TP	SRP	NO3-N	NH4-N	TKN	TN	TN TP
10-May-92	0-6m	11	<2 0	3		99	102	9
31-May-92	0-6m	10	<2 0	7	4	78	85	9
29-Jun-92	0-6m	6	<2 0	5		60	65	10
21-Jul-92	0-6m	6	<2 0	3		71	74	13
31-Jul-92	0-6m	4						
31-Jul-92	9m	3						
31-Jul-92	23m	5						
24-Aug-92	0-6m	15	3 4	4		140	144	10
18-Sep-92	0-6m	7	3 4	2		70	72	10
10-Oct-92	0-6m	5	2 1	3		91	94	19
20-Mar-93	0m	20		9		280	289	14
20-Mar-93	0-6m		4 1	60	10			
20-Mar-93	11m	5	<2 0	22	8	69	91	17
20-Mar-93	23m	4	<2 0	58	37	114	172	43
16-May-93	0-6m	16		11		156	167	10
16-May-93	15m	8		6		108	114	14
16-May-93	20m	15		10		88	98	7
22-May-93	0-6m	10		7		133	140	14
10-Jun-93	0-6m	16	<2 0	3	39	93	96	6
10-Jun-93	10m	6	<2 0	4	2	117	121	19
10-Jun-93	13m	7	<2 0	1	5			
10-Jun-93	15m	4	2 6	3	5			
10-Jun-93	17m	3	2 5	2	5			
10-Jun-93	21m	5	<2 0	8	5	98	106	22
24-Jun-93	0-6m	6		2		99	101	16
09-Jul-93	0-6m	6	<2 0	2	1	68	70	12
09-Jul-93	13m	7	<2 0	7	4	84	91	13
09-Jul-93	24m	7	<2 0	21	17	108	129	18
23-Jul-93	0-6m	6		1		75	76	13
06-Aug-93	0-6m	5	2 6	8	12	80	88	20
06-Aug-93	15m	5	2 0	7	10	74	81	15
06-Aug-93	23m	6		52	10	60	112	18
03-Sep-93	0-6m	3	2 4	1	3	61	62	18
03-Sep-93	16m	5	<2 0	6	3	75	81	16
03-Sep-93	23m	6	<2 0	62	3	59	121	21
16-Sep-93	0-6m	7		1		61	62	9
10-Oct-93	0-6m	4	<2 0	1	3	63	64	15
10-Oct-93	17m	5	2 0	11	3	54	65	12
10-Oct-93	21m	3	<2 0	76	5	59	135	39
07-Nov-93	0-6m	5		11		107	118	25
07-Nov-93	14m	7		12		57	69	10
07-Nov-93	22m	7		13		62	75	11
17-Dec-93	0-6m	8						
17-Dec-93	10m	7						
17-Dec-93	23m	5						

Stream: **Stanley Lake Inflow**

All values are in ug/L, except TN:TP ratio,
which is by weight.
TN = NO3-N + TKN

Date	TP	NO3-N	TKN	TN	TN:TP
20-Mar-93	3				
18-Apr-93	6	21	47	68	12
03-May-93	9	27	87	114	12
13-May-93	75	38	*	45	1
16-May-93	35	18	278	296	8
22-May-93	24	17	123	140	6
01 -Jun-93	11	13	64	77	7
08-Jun-93	7	13	65	78	11
16-Jun-93	7	10	49	59	9
24-Jun-93	8	12	61	73	9
02-Jul-93	6	11	69	80	14
10-Jul-93	2	9	37	46	19
17-Jul-93	2	22	26	48	31
23-Jul-93	8	9	45	54	7
01 -Aug-93	7	2	94	96	15
08-Aug-93	7	14	38	52	8
14-Aug-93	3	10	50	60	19
21 -Aug-93	7	16	58	74	10
28-Aug-93	4	11	135	146	33
04-Sep93	3	16	47	63	25
11-Sep-93	6	11	43	54	10
18-Sep-93	5	13	15	28	6
10-Oct-93	4	13	7	20	5
06-Nov-93	6	23	2	25	4

Lake: Yellow Belly

All values are in ug/L, except TN:TP ratio, which is by weight.
TN = NO3-N + TKN

Date	Depth	TP	SRP	NO3-N	NH4-N	TKN	TN	TN:TP
09-May-92	0-6m	9	2.2	5		69	74	9
31-Jul-92	0-6m	7	<2.0	13		96	109	15
31-Jul-92	10m	4	<2.0	10				0
31-Jul-92	20m	5	3.2	8				0
24-Aug-92	0-6m	9	3.7	11		100	111	13
20-Sep-92	0-6m	8	4 0	8		138	146	17
22-Mar-93	0-6m	5	<2.0	15		76	91	18
22-Mar-93	10m	21	<2.0	1		58	59	3
22-Mar-93	21m	4	<2.0	3		114	117	32
11-Jun-93	0-6m	5	2.3	6	40	100	106	22
11-Jun-93	10m	7	<2.0	5	0			
11-Jun-93	14m	5	<2.0	4		143	147	28
11-Jun-93	17m	3	<2.0					
11-Jun-93	19m	4	2.2					
11-Jun-93	21m	3	<2.0	18	62	170	188	57
12-Jul-93	0-6m	5	2.0	9	6	64	73	15
12-Jul-93	15m	8	3.0	14	27	210	224	27
12-Jul-93	21m	5	2.0	10	67	232	242	51
09-Aug-93	0-6m	6	<2.0	4	1	53	57	9
09-Aug-93	15m	4	<2.0	15	3	122	137	37
09-Aug-93	21m	4	<2.0	17	70	188	205	58
09-Sep-93	0-6m	1	<2.0	1	3	77	78	71
09-Sep-93	18m	7	<2.0	14	45	136	150	21
09-Sep-93	21m	11	<2.0	14	103	187	201	18

Stream: Yellow Belly Inflow

All values are in ug/L, except TN:TP ratio,
which is by weight.
TN = NO3-N + TKN

Date	TP	NO3-N	TKN	TN	TN:TP
22-Mar-93	6	36	39	75	12
27-Apr-93	12	21	60	81	7
05-May-93	11	16	93	109	10
11-Jun-93	8	20	65	85	11
12-Jul-93	5	12	158	170	33
09-Aug-93	9	12	49	61	7
08-Sep-93	6	23	149	172	29

Lake Redfish

All values are in ug/L, except TN TP ratiowhich is by weight
TN = NO3-N + TKN

Date	Depth	TP	SRP	NO3-N	NH4-N	TKN	TN	TN TP
10-May-92	0-6m	6		2		64	66	11
25-May-92	0-6m	5						
25-May-92	15m	7						
25-May-92	30m	5						
25-May-92	80m	7						
31-May-92	0-6m	7	<2 0	9		47	56	8
12-Jun-92	0-6m	7	2 4	3		32	35	5
29-Jun-92	0-6m	7	<2 0	2		29	31	4
09-Jul-92	0-6m	6	2 7	5		38	43	7
09-Jul-92	14m	6						
09-Jul-92	35m	9						
09-Jul-92	80m	5						
22-Jul-92	0-6m	4	<2.0	7		20	27	7
01-Aug-92	0-6m	3	<2.0					
01-Aug-92	20m	3	<2.0					
01-Aug-92	75m	32	<2.0					
11-Aug-92	0-6m	6	<2.0	11		66	77	13
11-Aug-92	12m	9						
1 1-Aug-92	35m	5						
11-Aug-92	79m	5						
25-Aug-92	0-6m	25	4 2	6		63	69	3
08-Sep-92	0-6m	10	<2.0	11		45	56	6
08-Sep-92	15m	10						
08-Sep-92	28m	9						
08-Sep-92	84 5m	13						
18-Sep-92	0-6m	10	<2 0	12		45	57	6
11-Oct-92	0-6m	8	2.7	3		31	34	4
11-Oct-92	30m	6						
11-Oct-92	85m	11						
18-Nov-92	0-6m	5	<2.0	9		34	43	9
21-Mar-93	0-6m	16	<2 0	10	3	73	83	5
21-Mar-93	0m	19						
21-Mar-93	10m	5	<2 0	5	15	53	58	12
21-Mar-93	50m	12	<2.0					
21-Mar-93	80m	6	<2.0	30	9	59	89	14
16-May-93	0-6m	10		8		55	63	6
16-May-93	30m	7		5		43	48	7
16-May-93	70m	8		7		40	47	6
01-Jun-93	0-6m	12		0		68	68	6
10-Jun-93	0-6m	*	4.3	1	1	80	81	
10-Jun-93	15m	5	<2.0	0	4			
10-Jun-93	22m	3	<2 0	1	0	55	56	17
10-Jun-93	30m	4	<2 0	9	47			
10-Jun-93	40m	4	<2 0	1	5			
10-Jun-93	50m	4	<2 0	0	4			
10-Jun-93	60m	4	<2.0	1	8			
10-Jun-93	82m	24	<2.0	17	8	66	83	4
21-Jun-93	0-6m	9	<2.0	1	0	67	68	7
09-Jul-93	0-6m	7	<2.0	2	3	53	55	8
09-Jul-93	22m	5	<2.0	1	0	65	66	13
09-Jul-93	88m		<2 0	8	4			
23-Jul-93	0-6m	4		1		63	64	16
06-Aug-93	0-6m	3	<2.0	5	10	63	68	20
06-Aug-93	24m	4	<2.0	2	1	64	66	15
06-Aug-93	84m	2	<2.0	246	10	53	299	171
03-Sep-93	0-6m	6	<2.0	1	1	47	48	7
03-Sep-93	24m	7	<2 0	1	1	32	33	5
03-Sep-93	84m	5	<2.0	10	15	71	81	15
15-Sep-93	0-6m	5		1		66	67	14
08-Oct-93	0-6m	5	<2.0	1	4	68	69	15
08-Oct-93	31.5m	4	<2.0	0	2	29	29	7
08-Oct-93	86m	17	<2 0	46	14	59	105	6
07-Nov-93	0-6m	5		1		31	32	6
07-Nov-93	26 5m	5		1		34	35	8
04-Dec-93	0-6m	5		38		61	99	22
04-Dec-93	85m	6				36		

Stream: Fishhook Creek

All values are in **ug/L,** except TN:TP ratio,
which is by **weight.**
TN = **NO3-N** + TKN

Date	TP	NO3-N	TKN	TN	TN:TP
26-Mar-92	8				
11-May-92	7	30	52	82	12
20-May-92	19	38			
25-May-92	10	18	78	98	10
31 -May-92	12	23			
12-Jun-92	14	22			
19 Jun-92	9	-8	31	39	4
29 Jun-92	15	18	102	120	8
09 Jul-92	8	14	42	58	7
21 Jul-92	9	11	44	55	8
10-Aug-92	9	11	80	71	8
24-Aug-92	15	5	80	85	4
11 -sep-92	12	5	58	81	5
13-Sep-92	18	1			
16-Sep-92	19	2			
18-Sep-92	19	2			
22-Sep-92	11	3	48	51	5
25-Sep-92	17	1			
1 o-act-92	9	4	51	55	8
18-Nov-92	9	23	52	75	8
19-Apr-93	11	8	82	90	8
03-May-93	19	7	97	104	5
12-May-93	42	14	425	439	10
16-May-93	31	27	258	283	9
17-May-93	159	18	1499	1517	10
22-May-93	14	25	23	48	4
01 Jun-93	5	24	170	194	41
08 Jun-93	11	23	102	125	12
20 Jun-93	47	12	205	217	5
22 Jun-93	17	13	172	185	11
01-Jul-93	8	13	50	83	8
14 Jul-93	12	11	39	50	4
23 Jul-93	3	8	119	127	44
01-Aug-93	7	13	75	88	13
08-Aug-93	5	9	59	88	15
14-Aug-93	13	12	95	107	8
22-Aug-93	9	18	58	72	8
28-Aug-93	4	18	75	93	28
04-Sep-93	8	14	114	128	17
11-Sep-93	5	15	58	73	14
18-Sep-93	10	13	48	59	8
08-Oct-93	8	8	34	40	5
06-Nov-93	10	38	34	70	7

Stream: Redfish Lake Creek Inflow

All values are in ug/L, except TN:TP ratio,
which is by weight.
TN = NO3-N + TKN

Date	TP	NO3-N	TKN	TN	TN:TP
26-Mar-92	7				
11-Apr-92	10	11			
13-May-92	6	32	21	53	9
25-May-92	9	43	21	64	7
31-May-92	6	44			
12-Jun-92	5	49			
19-Jun-92	9	19	38	57	6
29-Jun-92	39	35	130	165	4
09-Jul-92	7	42	21	63	9
22-Jul-92	8	33	25	58	7
01-Aug-92	19	44			
10-Aug-92	7	37	19	56	8
24-Aug-92	9	34	18	52	6
13-Sep-92	7	24	12	36	5
18-Sep-92	7	36	6	42	6
11-Oct-92	6	31	25	56	9
18-Nov-92	7	66	2	68	10
21-Mar-93	7	44	15	59	9
19-Apr-93	4	50	12	62	16
16-May-93	13	28	103	131	10
22-May-93	5	32	97	129	25
01-Jun-93	7	33	75	108	15
07-Jun-93	9	31	46	77	9
10-Jun-93	10	31	44	75	7
14-Jun-93	7	35	44	79	12
23-Jun-93	10	35	44	79	8
09-Jul-93	5	6	70	76	16
10-Jul-93	3	49	50	99	34
17-Jul-93	3	39	31	70	23
23-Jul-93	7	30	39	69	10
01-Aug-93	9	45	57	102	12
08-Aug-93	4	34	54	88	25
16-Aug-93	6	23	63	86	14
22-Aug-93	6	49	42	91	15
28-Aug-93	9	45	24	69	8
05-Sep-93	5	39	106	145	30
14-Sep-93	2	40	24	64	27
08-Oct-93	4	47	21	68	16
07-Nov-93	5	81	11	92	19

Lake Alturas

All values are in ug/L, except TN TP ratio, which is by weight
TN = NO3-N + TKN

Date	Depth	TP	SRP	NO3-N	NH4-N	TKN	TN	TN.TP
09-May-92	0-6m	10	2.8	2		72	74	7
24-May-92	0-6m	5						
24-May-92	15m	7						
24-May-92	44m	14						
30-May-92	0-6m	6						
20-Jun-92	0-6m	7						
30-Jun-92	0-6m	7	<2 0	2		55	57	8
10-Jul-92	0-6m	6						
10-Jul-92	16m	6						
10-Jul-92	40m	8						
20-Jul-92	0-6m	9	<2 0	14		m	84	9
30-Jul-92	0-6m	6	<2 0	2				
30-Jul-92	13m	96	<2 0	4				
30-Jul-92	48m	45	<2 0	2				
10-Aug-92	0-6m	7	<2 0	1		89	90	12
10-Aug-92	10m	8						
10-Aug-92	45m	15						
25-Aug-92	0-6m	7						
09-Sep-92	0-6m	10						
09-Sep-92	25m	8						
09-Sep-92	48m	8						
19-Sep-92	0-6m	7	<2 0	3		78	81	12
10-Oct-92	0-6m	9	<2 0	4		79	83	9
10-Oct-92	18m	16						
10-Oct-92	48m	15						
22-Mar-93	0-6m	7	2.0	12	66	74	86	12
22-Mar-93	10m	7	2 9	9	32	49	58	8
22-Mar-93	46m	8	3.5	35	4	27	62	8
17-May-93	0-6m	15		7		80	87	6
17-May-93	25m	9		3		69	72	8
17-May-93	45m	7		2		47	49	8
23-May-93	0-6m	7		1		81	82	12
09-Jun-93	0-6m	10	2 0	2		77	79	8
09-Jun-93	10m	7	<2 0	1	6			
09-Jun-93	18m	6	<2 0	2	7	86	88	16
09-Jun-93	25m	6	<2.0	0	5			
09-Jun-93	35m	8	<2 0	1	8			
09-Jun-93	45m	14	2.4	21	1	68	89	7
23-Jun-93	0-6m	14		2		170	172	12
08-Jul-93	0-6m	8	<2 0	10	3	100	110	14
08-Jul-93	18m	7	<2 0	60	2	93	153	22
08-Jul-93	45m	8	2 6	52		53	105	13
22-Jul-93	0-6m	9		1		73	74	8
05-Aug-93	0-6m	7	<2 0	2	3	67	69	10
05-Aug-93	24m	8	<2 0	2		21	23	3
05-Aug-93	45m	13	2 0	22	2	47	69	5
02-Sep-93	0-6m	5	<2 0	1	2	54	55	11
02-Sep-93	21m	6	<2 0	1	2	61	62	11
02-Sep-93	48m	7	3 7	5		70	75	11
16-Sep-93	0-6m	7				50		
09-Oct-93	0-6m	5	<2 0	1	4	130	131	27
09-Oct-93	23m	6	<2 0	1	4	82	83	13
09-Oct-93	45m	6	<2 0	1	3	66	67	11
06-Nov-93	0-6m	7		1		94	95	13
06-Nov-93	20m	6		1		76	77	12
06-Nov-93	47m	8		1		73	74	9
05-Dec-93	0-6m	8		1		57	58	8
05-Dec-93	16m	9		1		82	83	9
05-Dec-93	47m	7		1		55	56	8

Stream: Alturas Inflow

All values are in ug/L, except TN:TP ratio,
which is by weight.
TN = NO3-N + TKN

Date	TP	NO3-N
11-Apr-92		29
11-May-92	9	11
24-May-92	7	9
30-May-92	9	7
08-Jun-92	11	5
15-Jun-92	12	9
20-Jun-92	12	7
30-Jun-92	13	2
10-Jul-92	9	6
20-Jul-92	12	10
30-Jul-92	11	9
10-Aug-92	8	11
25-Aug-92	8	5
11-Sep-92	8	8
19-Sep-92	9	10
20-Sep-92	8	8
10-Oct-92	7	17

Date	TP	NO3-N	TKN	TN	TN:TP
22-Mar-93	6	24	70	94	15
23-Apr-93	8	24	48	72	9
04-May-93	34	18	80	98	3
13-May-93	27	20	194	214	8
17-May-93	29	15	154	169	6
23-May-93	17	13	96	109	6
01-Jun-93	9	7	160	167	19
13-Jun-93	5	4	44	48	10
23-Jun-93	10	6	54	60	6
01-Jul-93	8	4	54	58	7
10-Jul-93	6	3	58	61	10
13-Jul-93	5				
17-Jul-93	7	10	21	31	4
23-Jul-93	6	4	57	61	10
01-Aug-93	8	11	49	60	7
08-Aug-93	5	6	47	53	10
14-Aug-93	8	8	38	46	6
21-Aug-93	4	6	35	41	10
28-Aug-93	9	11	40	51	6
04-Sep-93	5	10	101	111	24
11-Sep-93	5	15	57	72	14
18-Sep-93	5	14	30	44	10
09-Oct-93	5	11	33	44	9
06-Nov-93	7	30	25	55	8

Lake: **Pettit**

All values are in **ug/L,** except **TN:TP** ratio, which is by weight
TN = **NO3-N** + **TKN**

Date	Depth	TP	SRP	NO3-N	NH4-N	TKN	TN	TN:TP
W-May-92	0-6m	9	<2.0	3		84	87	10
24-May-92	0-6m	5						
30-May-92	0-6m	4	<2.0	11		91	102	27
20-Jun-92	0-6m	8						
30-Jun-92	0-6m	4	<2.0	4		83	87	19
10-Jul-92	0-6m	6						
10-Jul-92	16m	4						
10-Jul-92	45m	18						
20-Jul-92	0-6m	4						
29-Jul-92	0-6m	8						
30-Jul-92	0-6m	5	<2.0	2		94	96	21
30-Jul-92	11m	4	<2.0	2				
30-Jul-92	47m	10	<2.0	20				
10-Aug-92	0-6m	7	2.9	4		82	86	12
1O-Aug-92	1Om	6						
10-Aug-92	43m	16						
25-Aug-92	0-6m	12						
09-Sep-92	0-6m	9						
09-Sep-92	20m	10						
09-Sep-92	46m	23						
19-Sep-92	0-6m	8	2.2	4		84	88	11
10-Oct-92	0-6m	6	3.9	4		82	86	15
10-Oct-92	17m	5						
10-Oct-92	48m	21						
19-Mar-93	0-6m	3						
19-Mar-93	Om	5	<2.0	21	39	110	131	28
1g-Mar-93	20m	3	2.0	9	79	101	110	43
1g-Mar-93	46m	14	7.2	19	253	231	250	17
17-May-93	0-6m	9		5		104	109	12
17-May-93	25m	2		3		76	79	46
17-May-93	45m	6		58		74	132	24
23-May-93	0-6m	7						
09-Jun-93	0-6m	10	<2.0	3	4	97	100	10
09-Jun-93	15m	4	<2.0	0	4	90	90	25
09-Jun-93	22m	3	<2.0	10	14			
09 Jun-93	30m	4	<2.0	0	3			
09 Jun-93	40m	6	<2.0	0	4			
09 -Jun-93	47m	5	<2.0	55	106	136	191	36
23 -Jun-93	0-6m	8		2		45	47	8
08-Jul-93	0-6m	4	•	3	4	80	83	23
08-Jul-93	17.5m	4	•	21		79	100	27
08-Jul-93	45m	14	<2.0	44	12	75	119	8
22-Jul-93	0-6m	7		1		66	67	10
05-Aug-93	0-6m	7	<2.0	2	2			
05-Aug-93	24m	6	<2.0	3	2			
OS-Aug-93	45m	5	<2.0	59	69			
02-Sep-93	0-6m	3	2 6	1	2	38	39	14
02-Sep-93	23m	2	<2.0	1	1	61	62	26
02-Sep-93	45m	6	<2 0	89	16	104	193	30
16-Sep-93	0-6m	4		0		92	92	24
16-Sep-93	48m	12						
09-Oct-93	0-6m	9	2.0	1	3	65	68	8
09-Oct-93	25m	3	<2.0	1	2	66	67	22
09-Oct-93	45m	8	<2 0	2	59	149	151	25
06-Nov-93	0-6m	6		5		162	167	28
06-Nov-93	24m	5		0		85	85	16
06-Nov-93	46m	11		5		278	283	25

Stream: Pettit South Inflow

All values are in ug/L, except TN:TP ratio,
which is by weight.
TN = NO3-N + TKN

Date	TP	NO3-N
24-May-92	3	22
30-May-92	4	16
08-Jun-92	3	26
15-Jun-92	4	14
20-Jun-92	5	12
30-Jun-92	6	10
10-Jul-92	4	11
20-Jul-92	6	17
30-Jul-92	5	20
10-Aug-92	4	28
25-Aug-92	4	18
11-Sep-92	4	27
19-Sep-92		22
24-Sep-92	5	18
10-Oct-92	5	42

Date	TP	NO3-N	TKN	TN	TN:TP
28-Apr-93	6	52	21	73	12
05-May-93	10	48	67	115	11
15-May-93	8	14	116	130	17
23-May-93	5	17	78	95	21
01-Jun-93	4	16	56	72	19
08-Jun-93	4	10	87	97	26
16-Jun-93	3	12	45	57	19
23-Jun-93	3	15	68	83	28
01-Jul-93	4	14	97	111	31
10-Jul-93	5	9	71	80	17
17-Jul-93	4	12	47	59	15
23-Jul-93	4	9	91	100	26
01-Aug-93	3	10	110	120	39
08-Aug-93	3	12	115	127	46
14-Aug-93	2	16	42	58	27
21-Aug-93	2	14	66	80	52
28-Aug-93	4	16	60	76	17
04-Sep-93	11	22	83	105	9
11-Sep-93	6	23	104	127	21
18-Sep-93	3	16	13	29	10
09-Oct-93	6	18	111	129	22
06-Nov-93	4	48	12	60	15

Stream: Pettit North Inflow

All values are in ug/L, except TN:TP ratio,
which is by weight.
TN = NO3-N + TKN

Date	TP	NO3-N
11-Apr-92		39
12-May-92	5	17
24-May-92	4	26
30-May-92	1	20
08-Jun-92	1	19
15-Jun-92	1	15
20-Jun-92	4	16
30-Jun-92		15
10-Jul-92	2	12
20-Jul-92	5	16
30-Jul-92	2	29
10-Aug-92	4	42
25-Aug-92	6	24
11-Sep-92	3	20
19-Sep-92	6	21
24-Sep-92	7	6
10-Oct-92	5	29

Date	TP	NO3-N	TKN	TN	TN:TP
28-Apr-93	13	60	53	113	9
05-May-93	10	26	52	78	8
15-May-93	9	22	117	139	16
23-May-93	5	20	85	105	22
01-Jun-93	7	23	21	44	6
08-Jun-93	8	13	70	83	11
16-Jun-93	6	15	78	93	15
23-Jun-93	8	21	91	112	15
01-Jul-93	3	17	54	71	24
10-Jul-93	3	9	79	88	33
17-Jul-93	3	12	91	103	34
23-Jul-93	6	10	94	104	18
01-Aug-93	4	17	48	65	18
08-Aug-93	2	18	41	59	27
14-Aug-93	3	12	88	100	33
21-Aug-93	2	13	67	80	33
28-Aug-93	10	22	61	83	8
04-Sep-93	13	22	76	98	8
11-Sep-93	4	28	87	115	32
18-Sep-93	3	25	56	81	25
09-Oct-93	6	18	60	78	13
06-Nov-93	5	64	48	112	25

Lake. Stanley

All values are in ug/L, except TN TP ratio which is by weight
TN = NO3-N + TKN

Date	Depth	TP	SRP	NO3-N	NH4-N	TKN	TN	TN.TP
10-May-92	0-6m	11	<2 0	3		99	102	9
23-May-92	0-6m	14						
23-May-92	14m	6						
23-May-92	24m	5						
31-May-92	0-6m	9	<2 0	7	4	78	85	9
12-Jun-92	0-6m	14						
12-Jun-92	0-6m	17						
19-Jun-92	0-6m	10						
29-Jun-92	0-6m	6	<2 0	5		60	65	10
09-Jul-92	0-6m	10						
09-Jul-92	10m	12						
09-Jul-92	24m	9						
21-Jul-92	0-6m	6	<2 0	3		71	74	13
31-Jul-92	0-6m	4						
31-Jul-92	9m	3						
31-Jul-92	23m	5						
09-Aug-92	0-6m	7						
09-Aug-92	9m	11						
09-Aug-92	20m	18						
24-Aug-92	0-6m	15	3.4			140	144	10
08-Sep-92	0-6m	9						
08-Sep-92	12m	13						
08-Sep-92	23m	15						
18-Sep-92	0-6m	7	3.4	2		70	72	10
10-Oct-92	0-6m	5	2.1	3		91	94	19
10-Oct-92	17m	9						
10-Oct-92	22m	16						
18-Nov-92	0-6m	7						
20-Mar-93	0m	20		9		280	289	14
20-Mar-93	0-6m		4.1	60	10			
20-Mar-93	11m	5	<2 0	22	8	69	91	17
20-Mar-93	23m	4	<2.0	58	37	114	172	43
16-May-93	0-6m	16		11		156	167	10
16-May-93	15m	8		6		108	114	14
16-May-93	20m	15		10		88	98	7
22-May-93	0-6m	10		7		133	140	14
10-Jun-93	0-6m	16	<2 0	3	39	93	96	6
10-Jun-93	10m	6	<2 0	4	2	117	121	19
10-Jun-93	13m	7	<2 0	1	5			
10-Jun-93	15m	4	2.6	3	5			
10-Jun-93	17m	3	2.5	2	5			
10-Jun-93	21m	5	<2 0	8	5	98	106	22
24-Jun-93	0-6m	6		2		99	101	16
09-Jul-93	0-6m	6	<2 0	2	1	68	70	12
09-Jul-93	13m	7	<2 0	7	4	04	91	13
09-Jul-93	24m	7	<2 0	21	17	108	129	18
23-Jul-93	0-6m	6		1		75	76	13
06-Aug-93	0-6m	5	2 6	8	12	80	88	20
06-Aug-93	15m	5	2 0	7	10	74	81	15
06-Aug-93	23m	6		52	10	60	112	18
03-Sep-93	0-6m	3	2 4	1	3	61	62	18
03-Sep-93	16m	5	<2 0	6	3	75	81	16
03-Sep-93	23m	6	<2 0	62	3	59	121	21
16-Sep-93	0-6m	7		1		61	62	9
10-Oct-93	0-6m	4	<2 0	1	3	63	64	15
10-Oct-93	17m	5	2.0	11	3	54	65	12
10-Oct-93	21m	3	<2.0	76	5	59	135	39
07-Nov-93	0-6m	5		11		107	118	25
07-Nov-93	14m	7		12		57	69	10
07-Nov-93	22m	7		13		62	75	11
17-Dec-93	0-6m	8						
17-Dec-93	10m	7						
17-Dec-93	23m	5						

Stream: Stanley Lake Inflow

All values are in **ug/L,** except **TN:TP ratio,** which is by weight.
TN = **NO3-N** + TKN

Date	T P	**NO3-N**
11-**Apr-92**		34
13-May-92	15	17
23-May-92	11	12
31 -May-92	14	17
09 Jun-92	7	17
15 Jun-92	10	12
19-Jun-92	5	
29 Jun-92		17
09 Jul-92	6	9
21 Jut-92	7	12
31-**Jul-92**	3	**11**
IO-Aug-92	9	12
24-Aug-92	5	7
11-**Sep-92**	10	4
18-Sep-92	10	4
25-Sep-92	9	10
1**0-Oct-92**	3	10
18-Nov-92	4	

Date	TP	**NO3-N**	**TKN**	T N	**TN:TP**
20-Mar-93	3				
18Apr-93	8	21	47	68	12
03-May-93	9	27	87	114	12
13-May-93	75	38	•	45	1
16-May-93	35	18	278	298	8
22-May-93	24	**17**	123	140	6
01 Jun-93	11	13	64	77	7
08-Jun-93	7	13	65	78	11
16-Jun-93	7	10	49	59	9
24-**Jun-93**	8	12	61	73	9
02 Jtd-93	6	11	69	80	14
1**0-Jul-93**	2	9	37	46	19
17Juf-93	2	22	26	48	31
23-**Jul-93**	8	9	45	54	7
01-**Aug-93**	7	2	94	96	15
08-Aug-93	7	14	38	52	8
14-Aug-93	3	10	50	60	19
21-**Aug-93**	7	18	58	74	10
28-Aug-93	4	11	135	146	33
04-Sep-93	3	16	47	83	25
11-**Sep-93**	6	11	43	54	10
1**8-Sep-93**	5	13	15	28	8
1**0-Oct-93**	4	13	7	20	5
06-Nov-93	6	23	2	25	4

Lake. Yellow Belly

All values are in ug/L, except TN TP ratio, which is by weight
TN = NO3-N + TKN

Date	Depth	TP	SRP	NO3-N	NH4-N	TKN	TN	TN.TP
09-May-92	0-6m	9	2 2	5		69	74	9
25-May-92	0-6m	4						
25-May-92	10m	5						
25-May-92	22m	6						
01-Jul-92	0-6m	5						
11-Jul-92	0-6m	4						
11-Jul-92	8m	5						
11-Jul-92	20m	6						
21-Jul-92	0-6m	5						
31-Jul-92	0-6m	7	<2 0	13		96	109	15
31-Jul-92	10m	4	<2.0	10				0
31-Jul-92	20m	5	3 2	8				0
09-Aug-92	0-6m	5						
09-Aug-92	13m	5						
09-Aug-92	22m	9						
24-Aug-92	0-6m	9	3.7	11		100	111	13
10-Sep-92	0-6m	6						
10-Sep-92	14m	6						
10-Sep-92	20m	12						
20-Sep-92	0-6m	8	4 0	8		138	146	17
09-Oct-92	0-6m	5						
09-Oct-92	14m	6						
09-Oct-92	22m	11						
22-Mar-93	0-6m	5	<2 0	15		76	91	18
22-Mar-93	10m	21	<2.0	1		58	59	3
22-Mar-93	21m	4	<2.0	3		114	117	32
11-Jun-93	0-6m	5	2.3	6	40	100	106	22
11-Jun-93	10m	7	<2.0	5	0			
11-Jun-93	14m	5	<2.0	4		143	147	28
11-Jun-93	17m	3	<2.0					
11-Jun-93	19m	4	2.2					
11-Jun-93	21m	3	<2.0	18	62	170	188	57
12-Jul-93	0-6m	5	2.0	9	6	64	73	15
12-Jul-93	15m	8	3.0	14	27	210	224	27
12-Jul-93	21m	5	2.0	10	67	232	242	51
09-Aug-93	0-6m	6	<2.0	4	1	53	57	9
09-Aug-93	15m	4	<2.0	15	3	122	137	37
09-Aug-93	21m	4	<2.0	17	70	188	205	58
09-Sep-93	0-6m	1	<2.0	1	3	77	78	71
09-Sep-93	18m	7	<2.0	14	45	136	150	21
09-Sep-93	21m	11	<2.0	14	103	187	201	18

Stream: Yellow Belly Inflow

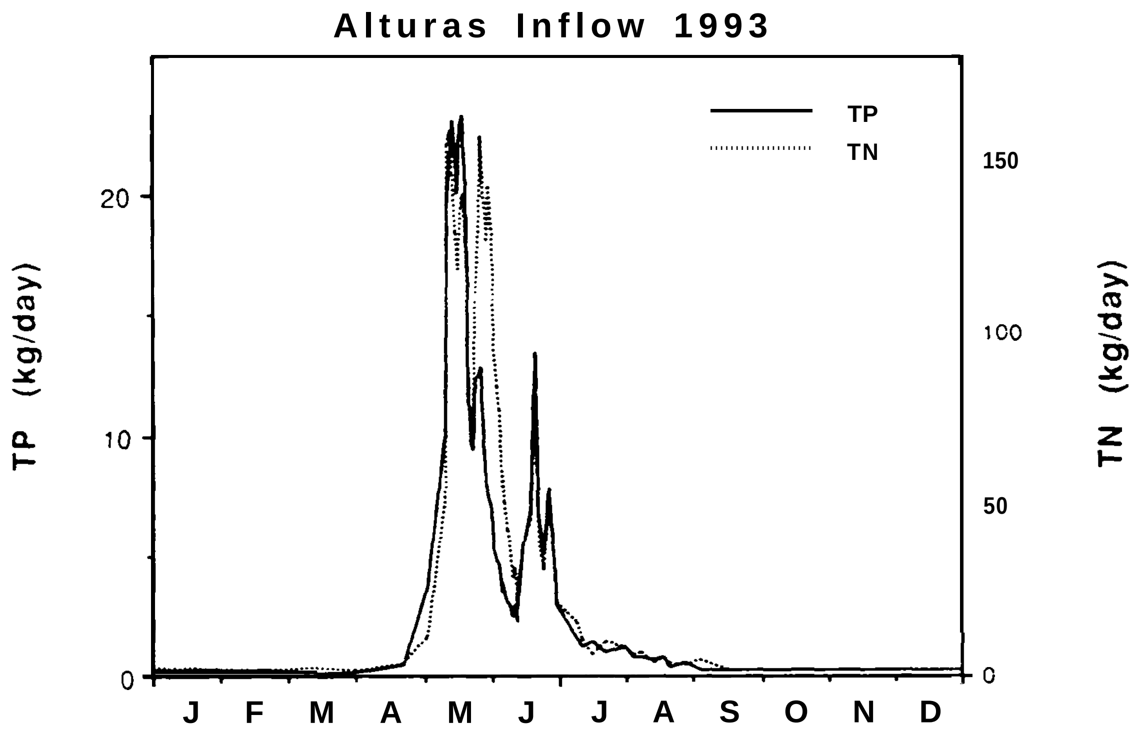
All values are in ug/L, except TN:TP ratio,
which is by weight.
NO3-N + TKN

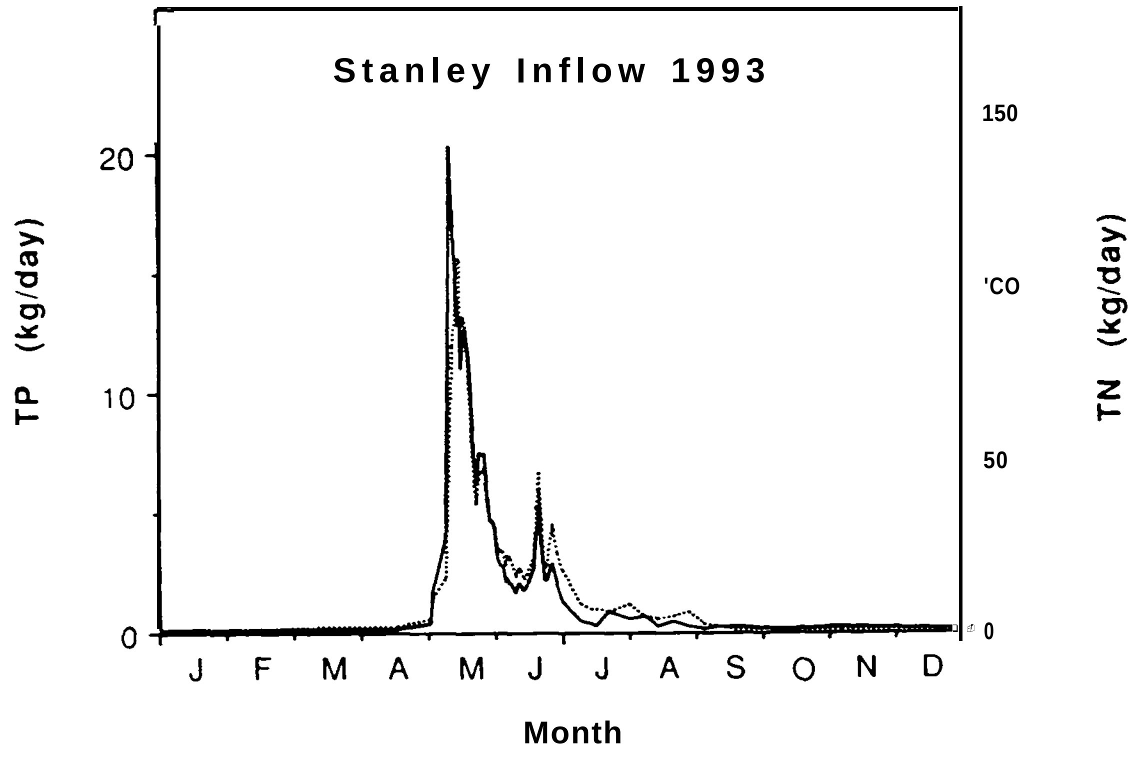
Date	TP	NO3-N
12-May-92	3	21
19-May-92	4	
25-May-92	16	31
01-Jun-92	7	18
11-Jun-92	4	15
15-Jun-92	6	13
20-Jun-92	5	5
01-Jul-92	10	12
11-Jul-92	8	49
21-Jul-92	6	10
31-Jul-92	10	10
09-Aug-92	8	6
24-Aug-92	10	
10-Sep-92	9	2
20-Sep-92	16	1
09-Oct-92	9	5

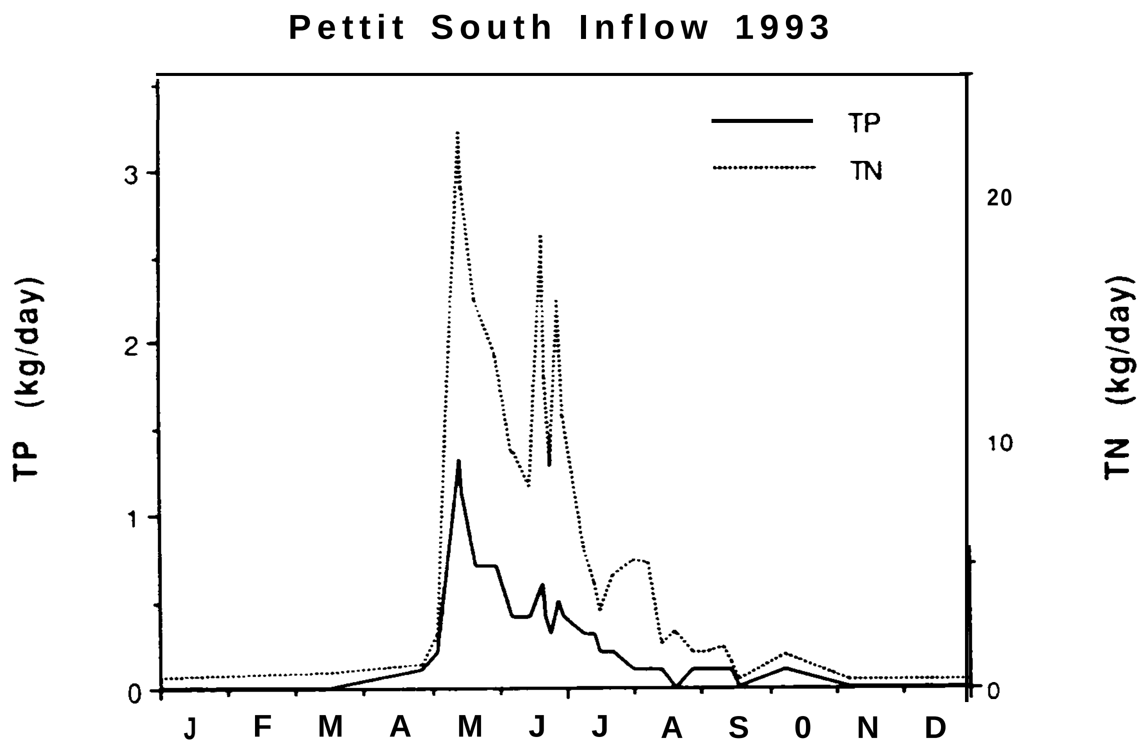
Date	TP	NO3-N	TKN	TN	TN:TP
22-Mar-93	6	36	39	75	12
27-Apr-93	12	21	60	81	7
05-May-93	11	16	93	109	10
11-Jun-93	8	20	65	85	11
12-Jul-93	5	12	158	170	33
09-Aug-93	9	12	49	61	7
08-Sep-93	6	23	149	172	29

Appendix 1B.

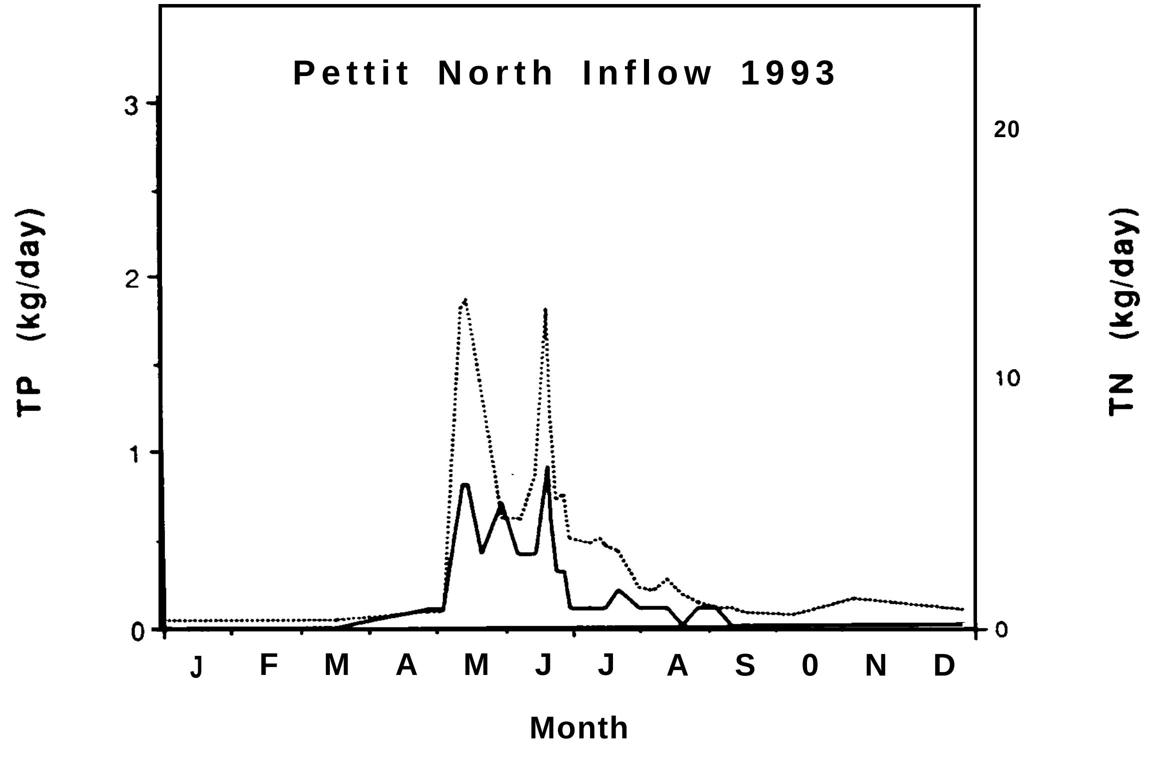
Temporal variation in total nitrogen and total phosphorus loading from tributary streams of the Sawtooth Valley Lakes.


Alturas Inflow 1993
TP
TN
TP (kg/day)
0
10
20
TN (kg/day)
0
50
100
150
J
F
M
A
M
J
J
A
S
O
N
D


Stanley Inflow 1993
TP (kg/day)
0
10
20
TN (kg/day)
0
50
'CO
150
J
F
M
A
M
J
J
A
S
O
N
D
Month

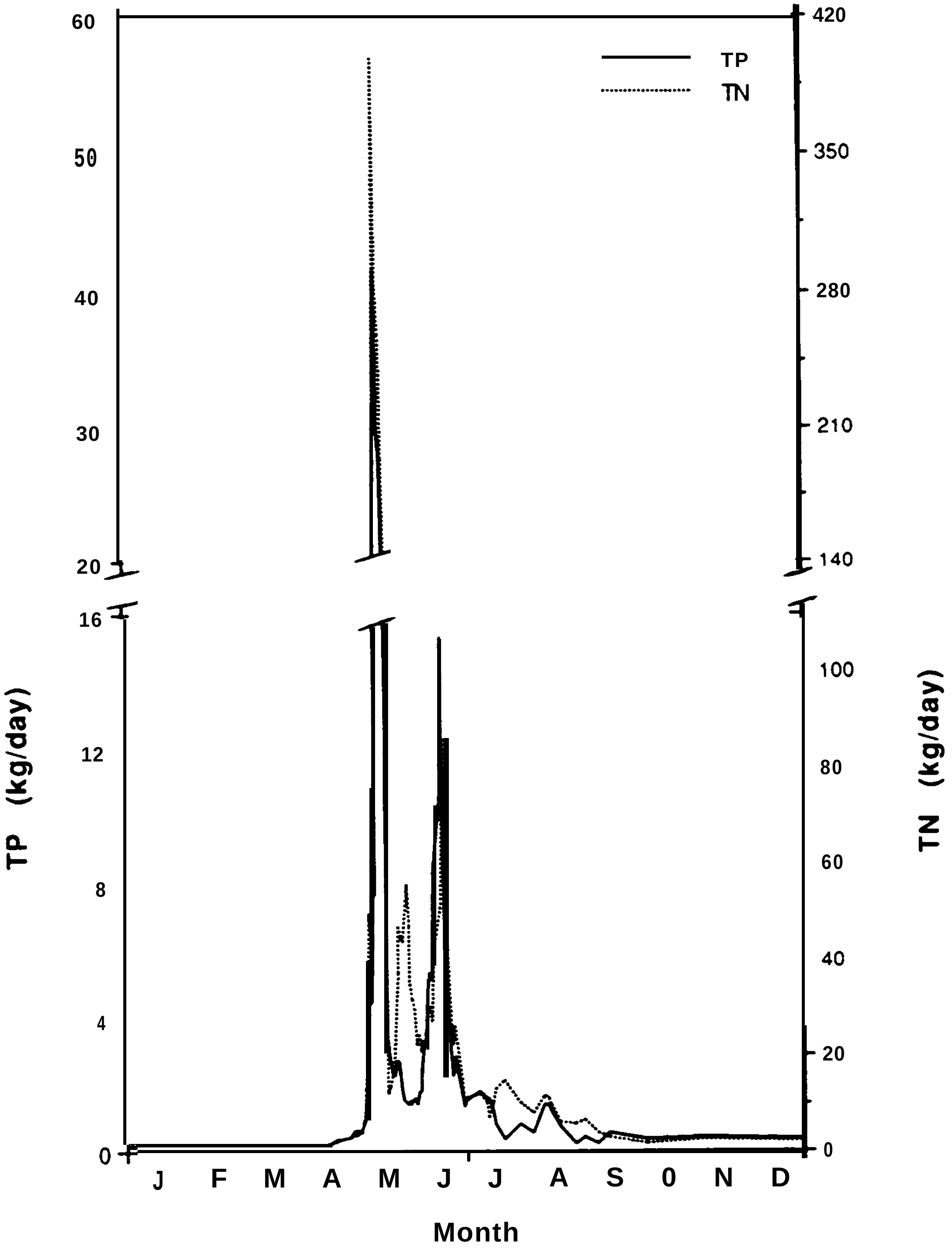


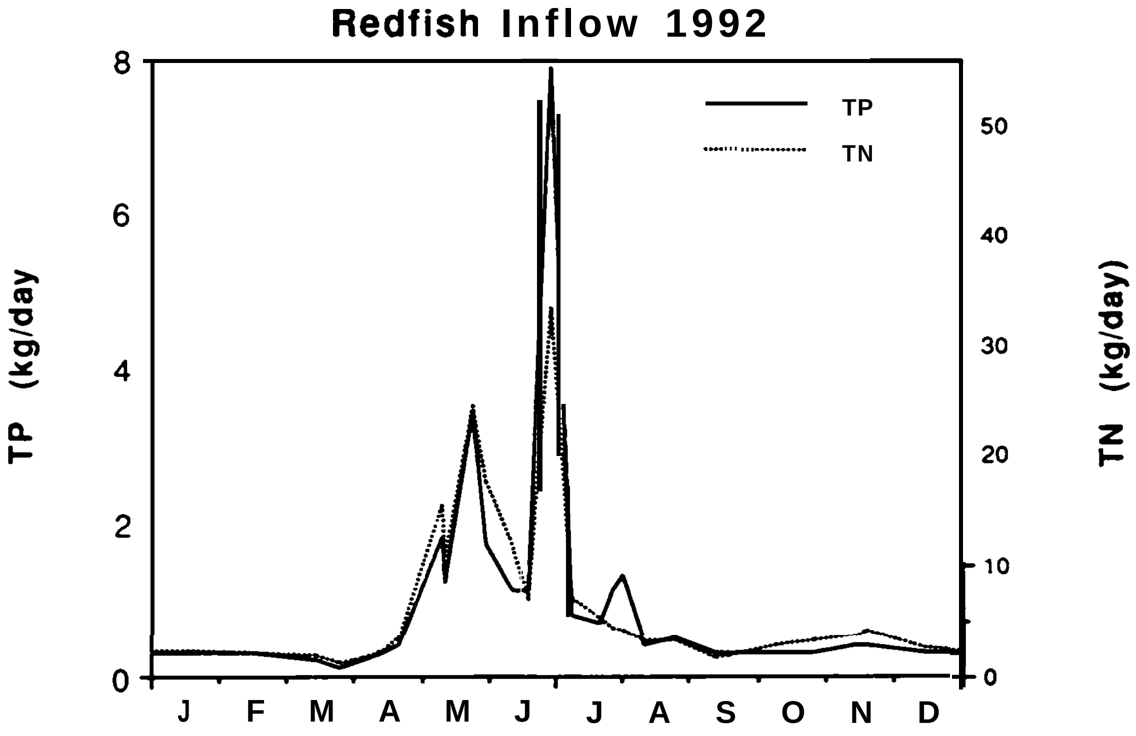
Pettit South Inflow 1993

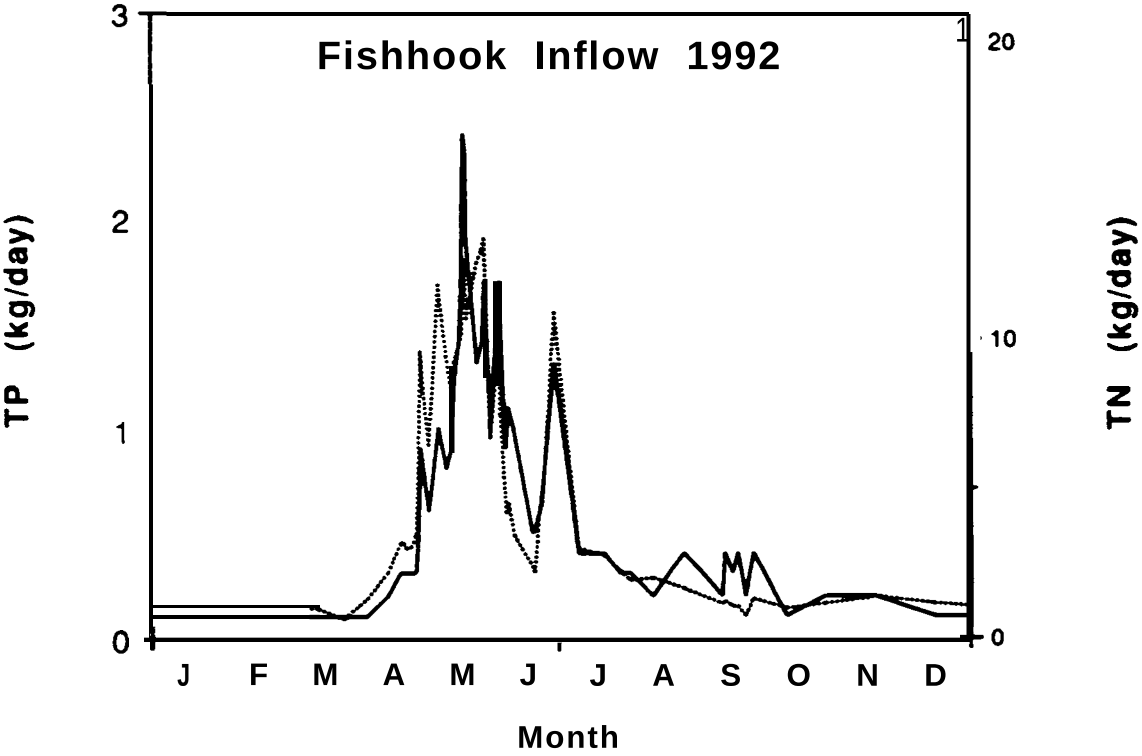


Pettit North Inflow 1993

Fishhook Inflow 1993




Redfish Inflow 1992
TP
TN
TP (kg/day)
TN (kg/day)
0
2
4
6
8
0
10
20
30
40
50
J
F
M
A
M
J
J
A
S
O
N
D


Fishhook Inflow 1992
TP (kg/day)
TN (kg/day)
0
1
2
3
0
10
20
J
F
M
A
M
J
J
A
S
O
N
D
Month

Appendix 2.

Temperature and chlorophyll profiles

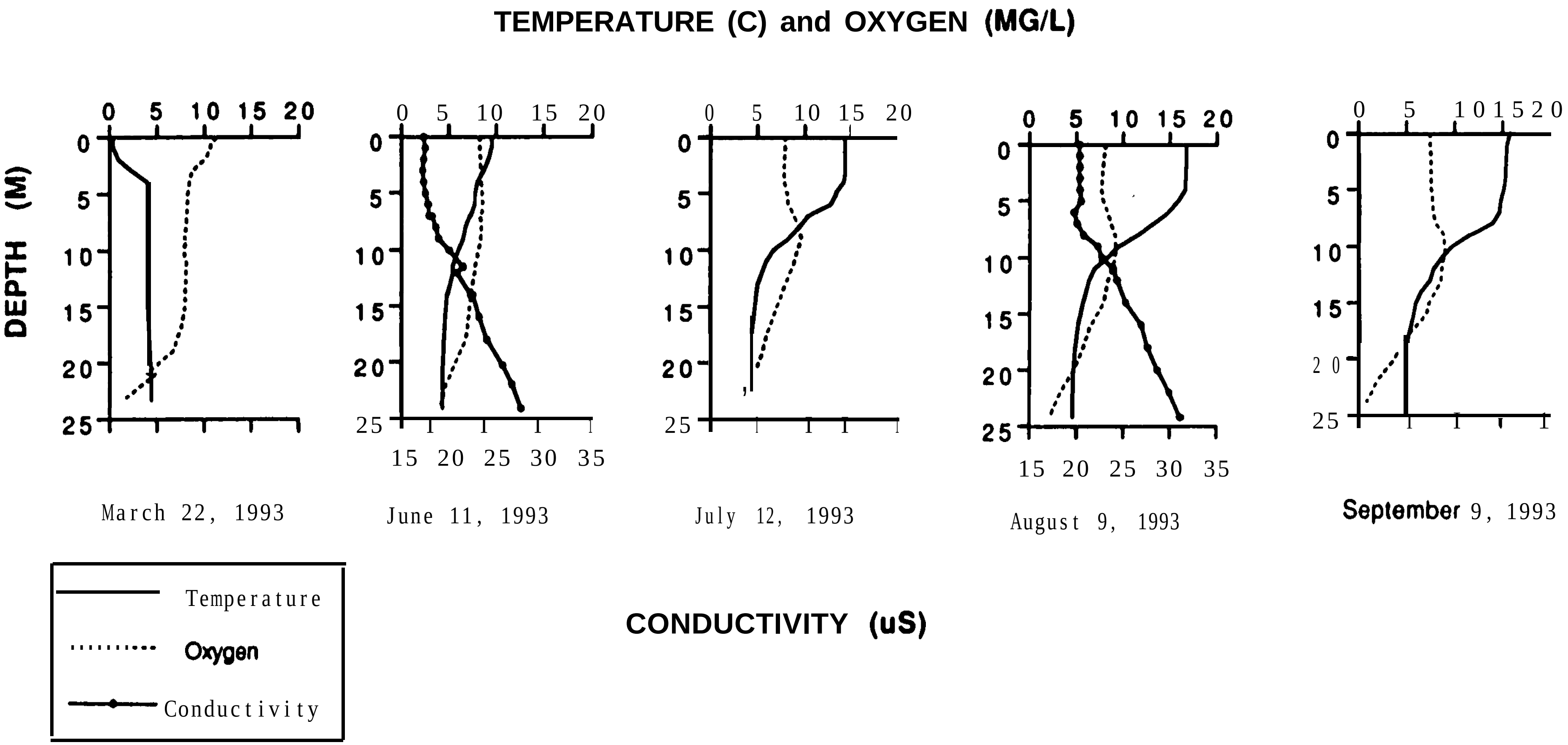


Figure 1. Temperature, oxygen, and conductivity rofiles **for** Yellow Belly Lake in 1993.

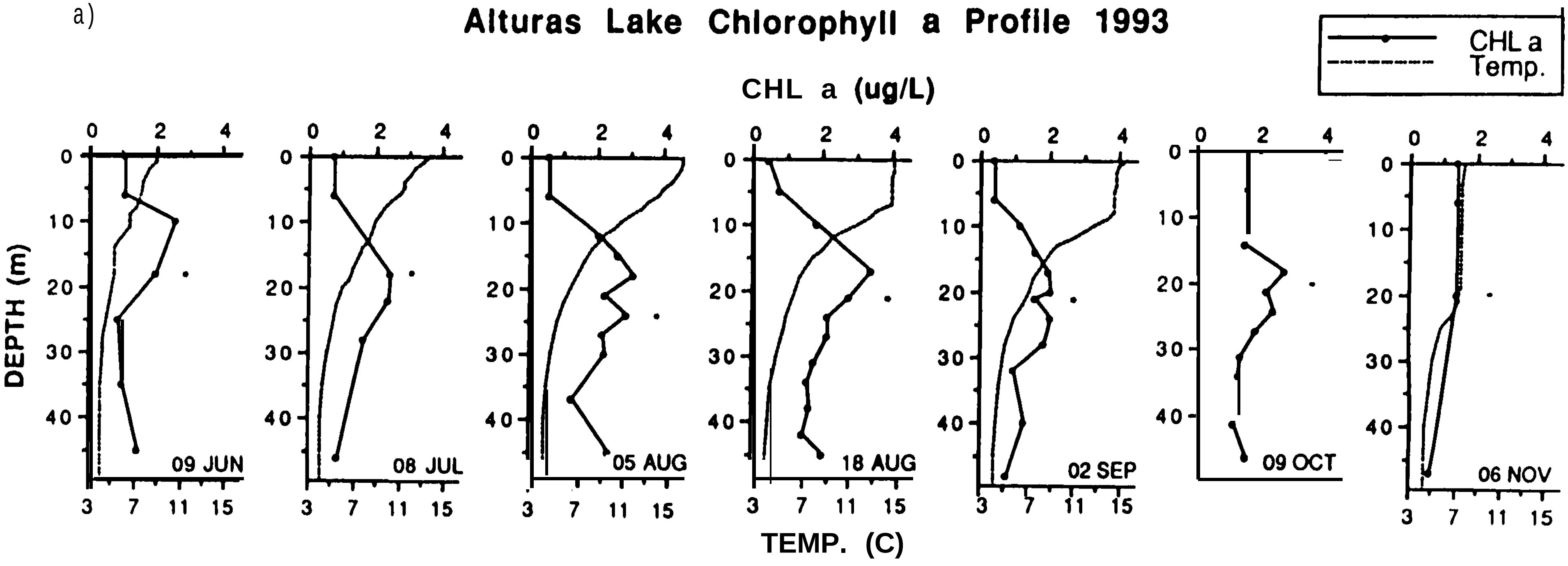


Figure 2. Chlorophyll a and temperature profiles for the Sawtooth Valley Lakes in 1993. **a)** Alturas **b)** Pettit **c)** Stanley **d)** Yellow Belly.

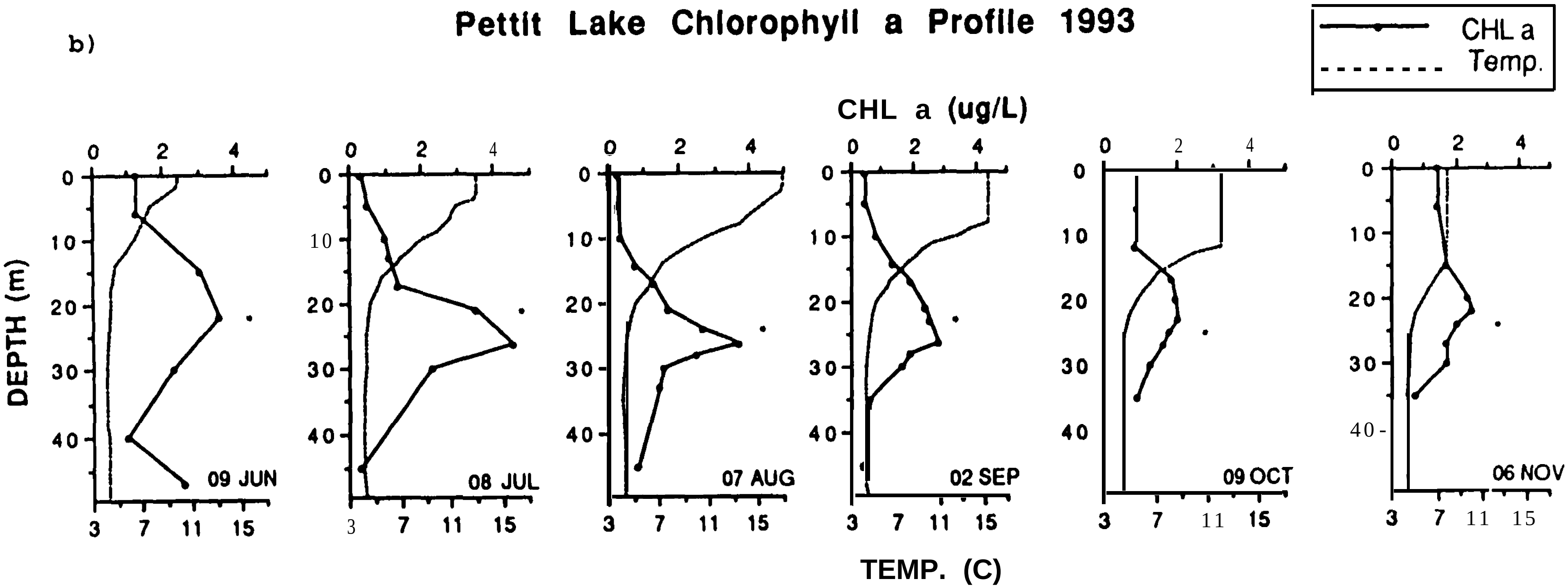


Figure 2. Continued.

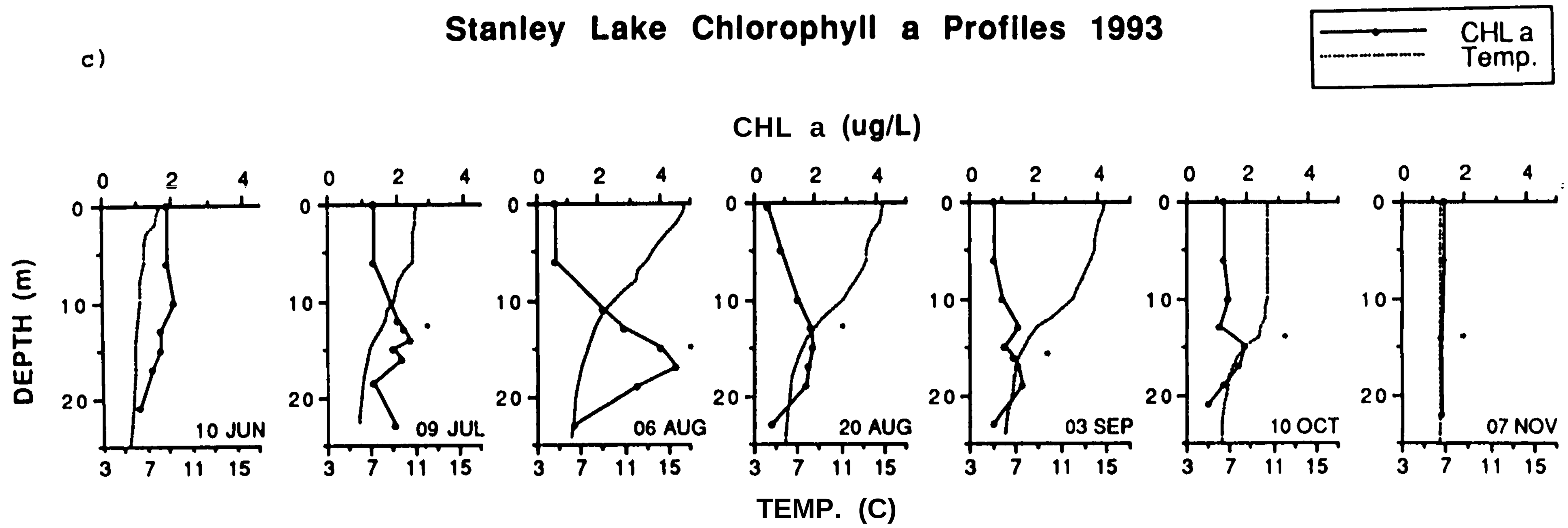


Figure 2. Continued.

d) **Y. Belly Lake Chlorophyll a Profiles 1993**

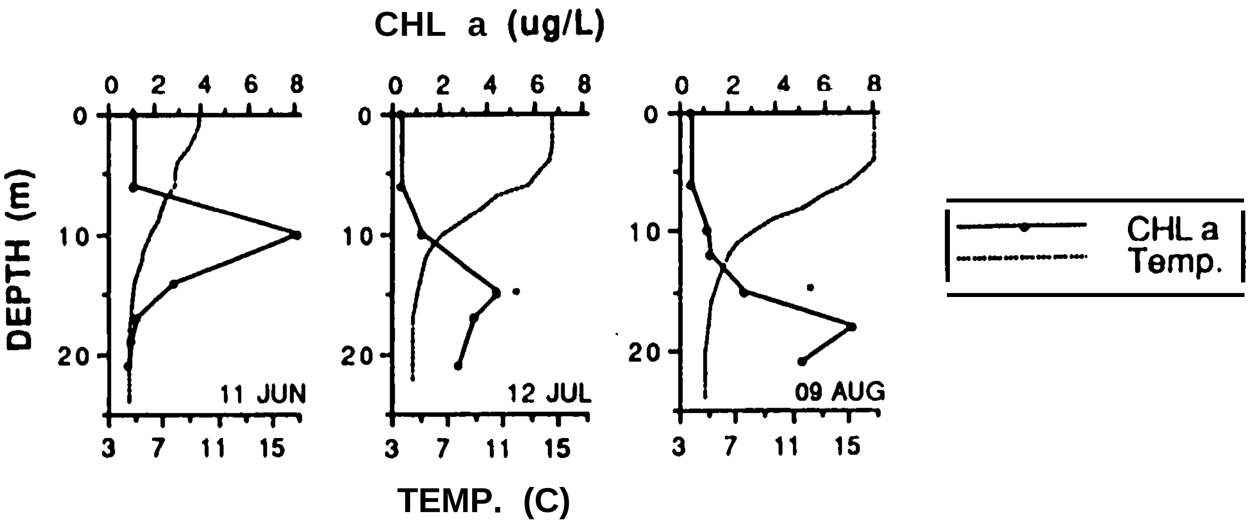
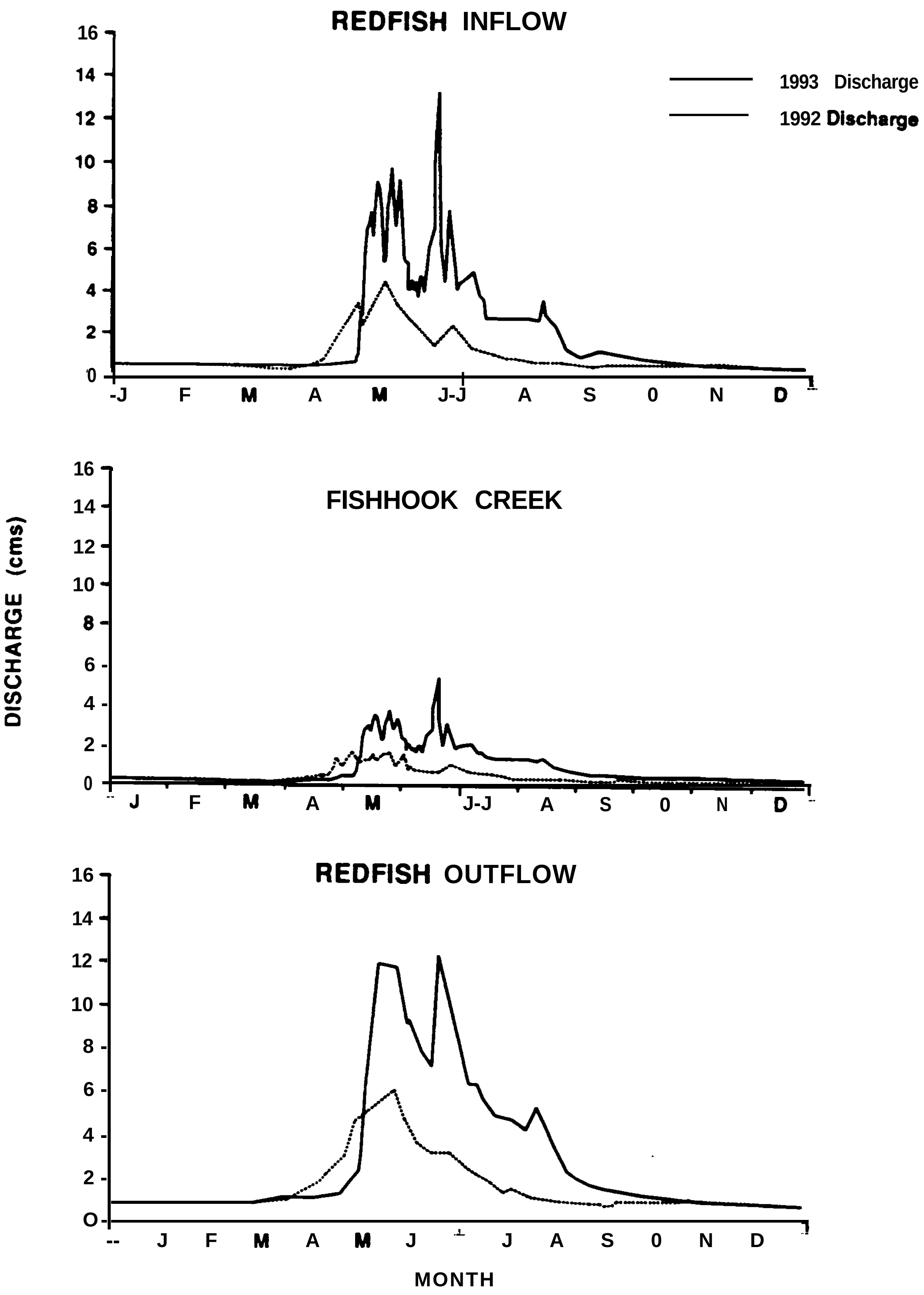
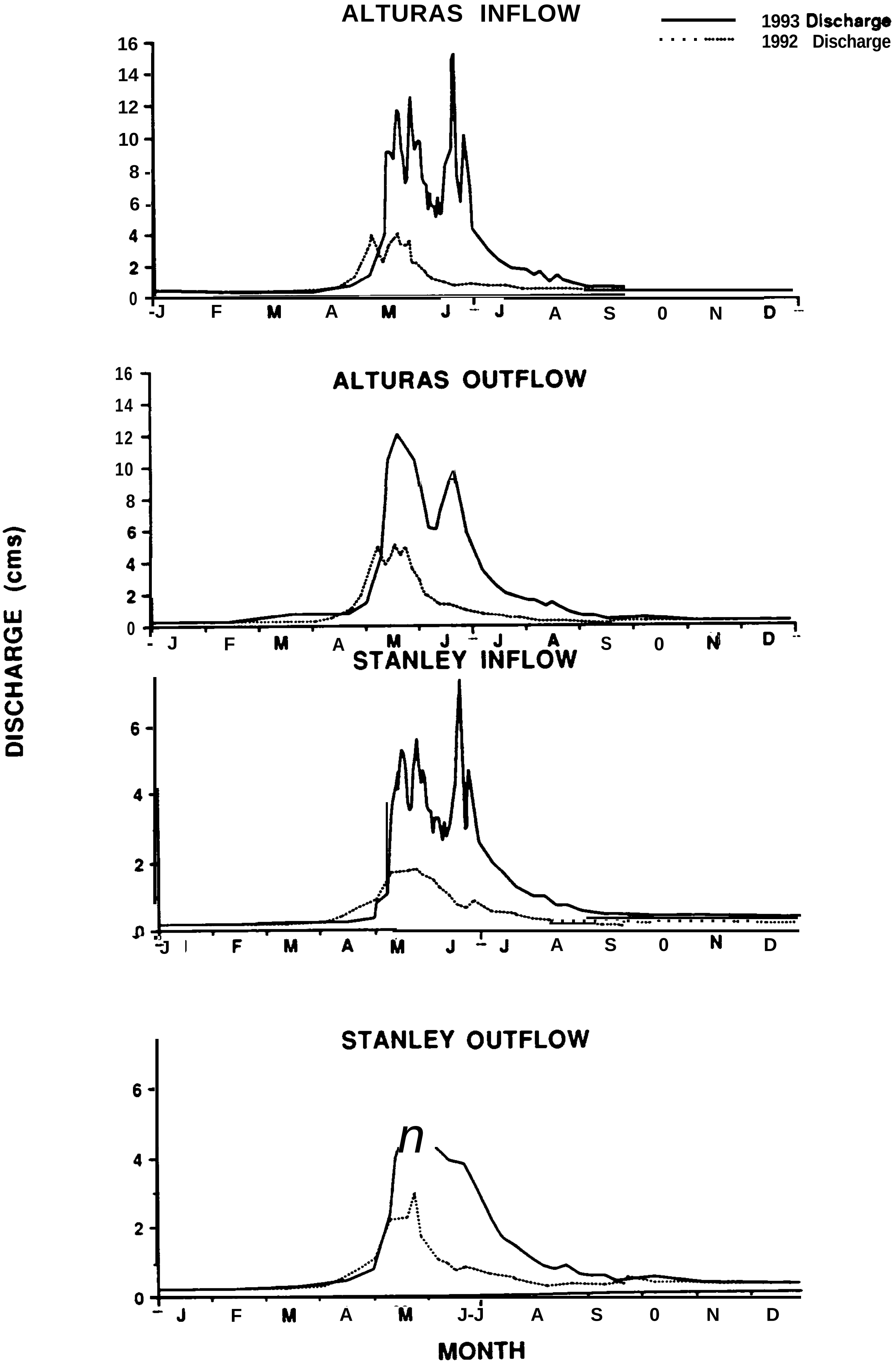


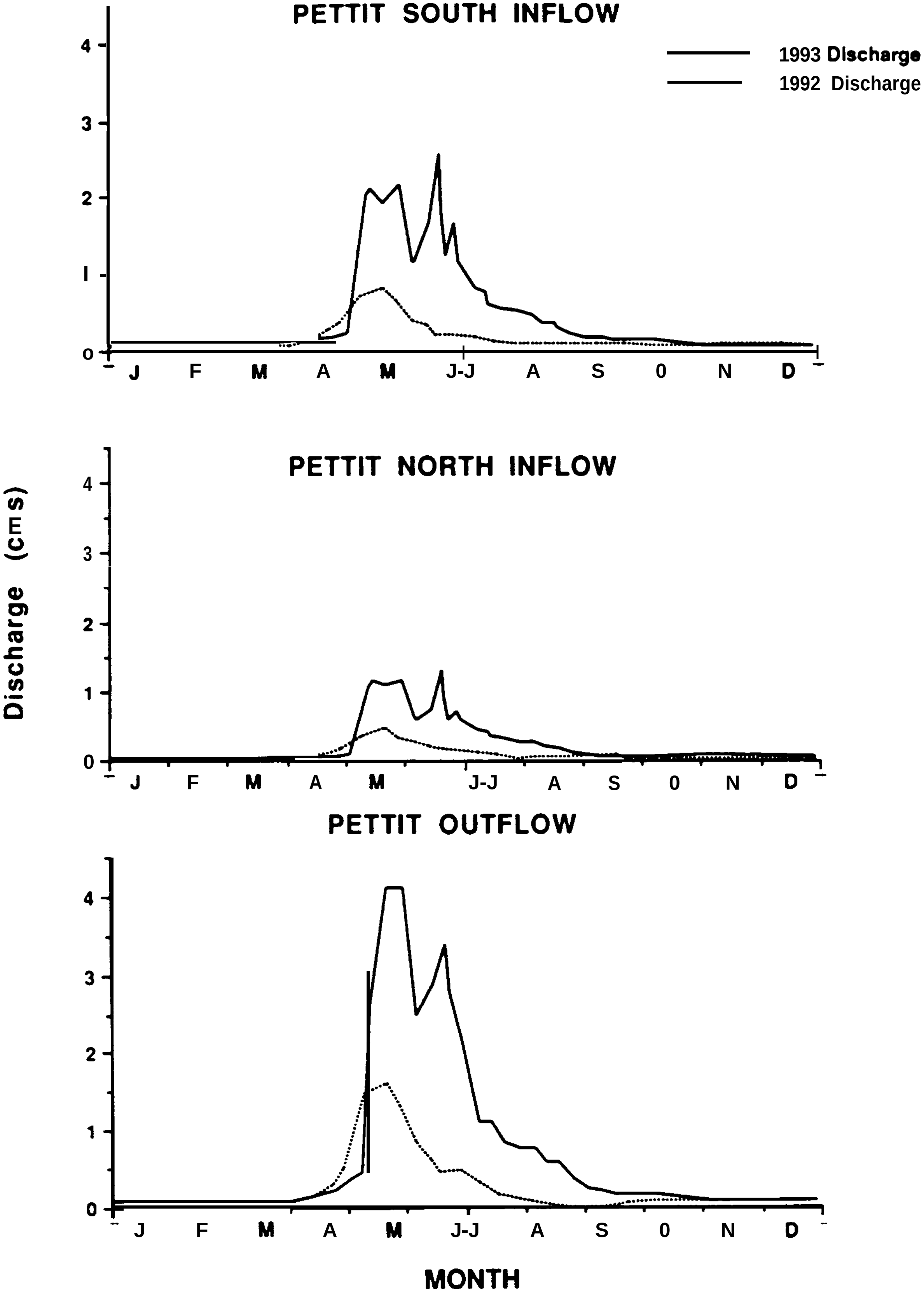
Figure 2. Continued.

Appendix 3.

Discharge of inflow and outflow streams
1992-1993


REDFISH INFLOW
1993 Discharge
1992 Discharge
16
14
12
10
8
6
4
2
0
J
F
M
A
M
J-J
A
S
0
N
D
FISHHOOK CREEK
DISCHARGE (cms)
REDFISH OUTFLOW
J
F
M
A
M
J
J
A
S
0
N
D
MONTH


ALTURAS INFLOW
1993 Discharge
1992 Discharge
16
14
12
10
8
6
4
2
0
J
F
M
A
M
J
J
A
S
0
N
D
ALTURAS OUTFLOW
16
14
12
10
8
6
4
2
0
J
F
M
A
M
J
J
A
S
0
N
D
STANLEY INFLOW
6
4
2
0
J
F
M
A
M
J
J
A
S
0
N
D
STANLEY OUTFLOW
6
4
2
0
J
F
M
A
M
J-J
A
S
0
N
D
DISCHARGE (cms)
MONTH



PETTIT SOUTH INFLOW

PETTIT NORTH INFLOW

PETTIT OUTFLOW

Appendix 4.

Hydroacoustic surveys 1993

	A	B	C	D	E	F	G	H	I	J	K
1	Redfish hydroacoustic survey										
2		16-Sep-93									
3											
4	Wayne collected data with repaired cable										
5	Numbers are in densities (#/ 1000 m3)										
6	Abundances are in thousands, volume is in million cubic meters										
7	Transect 1 is at south end, Transect 11 is near Fishhook creek										
8											
9	Small targets -59 to -51										
10	Transect	0-5	5-10	10-15	15-20	20-25	25-30	30-40	40-50	50-60	60-90
11	1.0000	11.3000	0.0000	0.7770	0.8960	0.0387	0.0851	0.0116			
12	2.0000	0.0000	0.0000	0.9940	0.6800	0.3390	0.0657	0.0000	0.0129	0.0083	
13	3.0000	0.0000	0.0000	0.7020	0.9630	0.0585	0.0740	0.0129	0.0000	0.0000	0.0000
14	4.0000	4.8500	0.4830	0.6930	0.5660	0.2140	0.0251	0.0000	0.0000	0.0000	0.0000
15	5.0000	2.9300	0.5830	1.2000	0.9140	0.3650	0.0645	0.0000	0.0000	0.0000	
16	6.0000	0.0000	0.3880	1.2300	0.7490	0.3110	0.0612	0.0055	0.0000		
17	7.0000	1.7000	0.6840	0.9010	1.9800	0.6310	0.0184	0.0421			
18	8.0000	1.3800	0.1860	1.4800	1.9300	0.3090	0.0000				
19	9.0000	0.0000	0.0000	1.1100	1.4560	0.0405	0.0000	0.0301	0.0000		
20	10.0000	0.0000	0.0000	0.9040	1.3100	0.1940					
21	11.0000	5.3200	0.8720	6.1860	1.7100						
22											
23											
24	mean	1.2067	0.2905	1.4706	1.1958	0.2501	0.0438	0.0128	0.0022	0.0021	0.0000
25	sd	1.6261	0.3101	1.5089	0.4848	0.1741	0.0310	0.0146	0.0048	0.0036	0.0000
26	se	0.5420	0.0935	0.4550	0.1462	0.0580	0.0110	0.0055	0.0020	0.0018	0.0000
27											
28	volume (million	29.00	28.00	23.00	22.00	16.00	28.00	29.00	34.00	22.00	39.00
29	abundance (X	34.99	8.14	33.82	26.31	4.00	1.23	0.37	0.07	0.05	0.00
30	2 se abund	31.44	5.24	20.93	6.43	1.86	0.61	0.32	0.13	0.08	0.00
31											
32	mean of 0-5 depths does not include transects 1 and 11										
33											
34	medium targets -51to -43 targets										
35	Transect	0-5	5-10	10-15	15-20	20-25	25-30	30-40	40-50	50-60	60-90
36	1.0000	0.0000	0.3010	0.3330	1.1900	0.2320	0.1700	0.0000			
37	2.0000	0.0000	0.0000	1.0200	0.7200	0.1460	0.0000	0.0000	0.0043	0.0000	
38	3.0000	0.0000	0.0000	0.5260	0.7790	0.5270	0.1030	0.0259	0.0000	0.0000	0.0000
39	4.0000	0.0000	0.1610	0.9240	1.0700	0.1830	0.1010	0.0000	0.0000	0.0000	0.0000
40	5.0000	0.0000	0.0000	1.2000	1.0600	0.2280	0.0986	0.0119	0.0000	0.0000	
41	6.0000	0.0000	0.1940	1.0900	1.2100	0.3110	0.0153	0.0000	0.0000		
42	7.0000	0.0000	0.0000	0.4920	2.3200	0.2890	0.0554	0.0070			
43	8.0000	0.0000	0.0000	1.0700	1.9300	0.2060					
44	9.0000	0.0000	0.0000	0.3020	1.5100	0.0000	0.0000		0.0277		
45	10.0000	0.0000	1.0600	1.4200	1.2400	0.1450					
46	11.0000	0.0000	0.8720	4.8800	1.9600						
47											
48											
49	mean	0.0000	0.2353	1.2052	1.3626	0.2267	0.0679	0.0064	0.0053	0.0000	0.0000
50	sd	0.0000	0.3605	1.2151	0.4877	0.1297	0.0568	0.0091	0.0101	0.0000	0.0000
51	se	0.0000	0.1087	0.3664	0.1470	0.0410	0.0201	0.0032	0.0041	0.0000	0.0000
52											
53	volume (million	29.00	28.00	23.00	22.00	16.00	28.00	29.00	34.00	22.00	39.00
54	ABUNDANCE >	0.00	6.59	27.72	29.98	3.63	1.90	0.19	0.18	0.00	0.00
55	2 SE ABUN	0.00	6.09	16.85	6.47	1.31	1.13	0.19	0.28	0.00	0.00
56											
57											
58	large targets >-43										
59	Transect	0-5	5-10	10-15	15-20	20-25	25-30	30-40	40-50	50-60	60-90
60	1.0000	0.0000	0.0000	0.3330	0.1790	0.2710	0.1130	0.0000			
61	2.0000	0.0000	0.0000	0.8650	1.0000	0.4120	0.0000	0.0000	0.0043	0.0042	
62	3.0000	0.0000	0.0000	0.5260	0.9630	0.4390	0.0000	0.0065	0.0000	0.0000	0.0000
63	4.0000	0.0000	0.1610	0.5760	0.7740	0.2140	0.0755	0.0000	0.0000	0.0000	0.0000
64	5.0000	0.0000	0.0000	0.4930	0.8040	0.0912	0.0484	0.0060	0.0000	0.0000	
65	6.0000	0.0000	0.0000	0.2740	0.6780	0.1560	0.0153	0.0000	0.0000		
66	7.0000	0.0000	0.0000	0.4920	0.5910	0.1580	0.0000	0.0000			
67	8.0000	0.0000	0.0000	0.0000	0.3090	0.0000					
68	9.0000	0.0000	0.0000	0.0000	0.0539	0.0000	0.0000	0.0000	0.0000		
69	10.0000	0.0000	0.0000	0.1290	0.1380	0.0968					
70	11.0000	0.0000	0.0000	0.3750	0.0000						
71											
72											
73	mean	0.0000	0.0146	0.3694	0.4991	0.1838	0.0315	0.0016	0.0006	0.0010	0.0000
74	sd	0.0000	0.0463	0.2494	0.3557	0.1452	0.0405	0.0027	0.0015	0.0018	0.0000
75	se	0.0000	0.0140	0.0752	0.1072	0.0459	0.0143	0.0010	0.0006	0.0009	0.0000
76											
77	VOLUME	29.00	28.00	23.00	22.00	16.00	28.00	29.00	34.00	22.00	39.00
78	ABUND (X 100	0.0000	0.4098	8.4954	10.9798	2.9408	0.8827	0.0451	0.0209	0.0229	0.0000
79	2SE ABUND	0.0000	0.78	3.46	4.72	1.47	0.80	0.06	0.04	0.04	0.00
80											

ALTURAS SURVEY
15-Sep-93
TRANSECT 1 IS NEAR INFLOW
RAMP IS AT BEGINNING OF TRANSECT 6

SMALL TARGETS

Transect	0-5	5-10	10-15	15-20	20-25	25-30	30-40	40-50
1.0000	0.0000	0.5440	0.5090	0.7160	2.1600	0.9160	0.6640	
2.0000	0.0000	0.6250	0.0758	1.2500	1.6500	1.0400	0.8660	0.0854
3.0000	0.0000	0.2180	0.0785	0.7640	1.4500	0.6570	0.6410	0.1070
4.0000	2.5600	0.0000	0.2730	0.9710	1.1300	0.4910	0.4580	0.1630
5.0000	0.0000	0.3970	0.0741	1.6900	1.6700	0.7730	0.6920	0.1460
6.0000	0.0000	0.0000	0.2630	1.7400	1.6000	0.5510	0.4840	0.1090
7.0000	0.0000	1.0600	0.3140	2.2800	2.9200	1.4600	0.8020	0.1200
8.0000	1.8900	0.0000	0.3670	3.0100	1.8900	1.4100	2.0800	0.3220
9.0000	0.0000	0.0000	0.5750	2.1400	1.9300	1.8400	1.8300	
MEAN	0.4944	0.3160	0.2810	1.6179	1.8222	1.0153	0.9463	0.1503
STD	0.9384	0.3522	0.1742	0.7277	0.4779	0.4374	0.5561	0.0740
SE	0.3128	0.1174	0.0581	0.2426	0.1593	0.1458	0.1854	0.0280
VOLUME MILLI	14.50	13.90	13.40	12.50	12.00	11.30	18.50	12.50
ABUNDANCE X	7.17	4.39	3.77	20.22	21.87	11.47	17.51	1.88
2 SE ABUND	9.07	3.26	1.56	6.06	3.82	3.29	6.86	0.70

MEDIUM TARGETS

Transect	0-5	5-10	10-15	15-20	20-25	25-30	30-40	40-50
1.0000	1.9800	0.0000	0.0000	1.4900	1.8700	0.5200	0.5250	
2.0000	0.0000	0.0000	0.1520	1.2900	1.4200	0.5460	0.9390	0.2050
3.0000	0.0000	0.0000	0.0785	1.9700	1.8200	0.4840	0.9070	0.0949
4.0000	3.8400	0.0000	0.2050	1.5300	1.3400	0.5890	0.5940	0.1190
5.0000	0.0000	0.0000	0.0000	1.6900	1.8500	0.4310	0.7760	0.1120
6.0000	0.0000	0.0000	0.6140	1.8300	1.6000	0.4600	0.4910	0.1270
7.0000	0.0000	0.0000	0.0786	2.4000	2.5500	0.9080	0.8780	0.1630
8.0000	0.0000	0.0000	0.1840	2.1400	2.3500	1.4100	1.8200	0.4030
9.0000	0.0000	0.0000	0.3450	1.4100	1.8300	1.5500	1.8500	
MEAN	0.6467	0.0000	0.1841	1.7500	1.8478	0.7664	0.9756	0.1748
STD	1.2868	0.0000	0.1833	0.3458	0.3726	0.4049	0.4849	0.0992
SE	0.4289	0.0000	0.0611	0.1153	0.1242	0.1350	0.1616	0.0375
VOLUME MILLI	14.50	13.90	13.40	12.50	12.00	11.30	18.50	12.50
ABUND X 100	9.38	0.00	2.47	21.88	22.17	8.66	18.05	2.19
2 SE ABUND	12.44	0.00	1.64	2.88	2.98	3.05	5.98	0.94

LARGE TARGETS

Transect	0-5	5-10	10-15	15-20	20-25	25-30	30-40	40-50
1.0000	0.0000	0.0000	0.0000	0.3310	1.1000	0.3960	0.1550	
2.0000	0.0000	0.0000	0.0785	0.8080	0.8490	0.3520	0.3450	0.0000
3.0000	0.0000	0.0000	0.0000	0.7230	0.5400	0.1560	0.1680	0.0253
4.0000	2.5600	0.0000	0.0683	1.1200	0.4000	0.1290	0.0260	0.0000
5.0000	0.0000	0.0000	0.3710	0.9240	0.8730	0.1490	0.0439	0.0000
6.0000	0.0000	0.0000	0.1750	0.8030	0.5160	0.1290	0.0613	0.0058
7.0000	0.0000	0.0000	0.0000	0.5800	0.7300	0.3030	0.1490	0.0650
8.0000	0.0000	0.0000	0.0918	0.7770	0.3720	0.0771	0.2340	0.2410
9.0000	0.0000	0.0000	0.0000	0.4450	0.4340	0.2950	0.3170	
MEAN	0.2844	0.0000	0.0872	0.7234	0.6460	0.2207	0.1666	0.0482
STD	0.8045	0.0000	0.1150	0.2275	0.2391	0.1091	0.1082	0.0817
SE	0.2682	0.0000	0.0383	0.0758	0.0797	0.0364	0.0361	0.0309
VOLUME MILLI	14.50	13.90	13.40	12.50	12.00	11.30	18.50	12.50
ABUNDANCE X	4.12	0.00	1.17	9.04	7.75	2.49	3.08	0.60
2 SE ABUND	7.78	0.00	1.03	1.90	1.91	0.82	1.33	0.77

	A	B	C	D	E	F	G	H	I
1									
2									
3	PETTIT LAKE ACOUSTIC SURVEY								
4	14-Sep-93								
5	Transect #1 is near outflow								
6	Transect #10 is near inflow								
7	Small targets -59 to -51								
8	Transect	0-5	5-10	10-15	15-20	20-25	25-30	30-40	40-50
9	1.0000	30.0700	6.1500	2.2000	0.1360	0.0412	0.0000		
10	2.0000	16.4000	0.6350	0.9400	0.5540	0.0000	0.0000		
11	3.0000	2.0900	2.7900	1.0100	0.0000	0.0000	0.0000		
12	4.0000	9.9300	3.2100	0.4820	0.0508	0.0000	0.0000	0.0000	
13	5.0000	13.9000	3.7200	0.8040	0.0000	0.0000	0.0000	0.0000	
14	6.0000	11.9000	3.1800	0.5720	0.0587	0.0000	0.0000	0.0000	0.0000
15	7.0000	37.0000	0.6080	0.3030	0.1550	0.0000	0.0000	0.0000	0.0000
16	8.0000	17.2000	5.2500	1.1700	0.0000	0.0000	0.0000	0.0000	0.0000
17	9.0000	11.8000	14.1000	5.3100	0.1170	0.3670	0.0000	0.0343	
18	10.0000	37.6000	11.5000	6.1400	0.0000	0.3270			
19									
20									
21									
22	mean	15.0275	5.1143	1.8931	0.1072	0.0735	0.0000	0.0057	0.0000
23	sd	9.3823	4.2154	1.5307	0.1737	0.1214	0.0000	0.0128	0.0000
24	se	3.3172	1.3330	0.4841	0.0549	0.0384	0.0000	0.0052	0.0000
25									
26	mean of 0-5 depths does not include transects 1 and 10								
27									
28	VOLUME X MILL	6.60	6.30	6.10	6.00	5.50	4.20	7.40	4.90
29	ABUND X 1000	99.18	32.22	11.55	0.64	0.40	0.00	0.04	0.00
30	2 SE ABUND	43.79	16.80	5.91	0.66	0.42	0.00	0.08	0.00
31									
32									
33	MEDIUM TARGETS								
34	Transect	0-5	5-10	10-15	15-20	20-25	25-30	30-40	40-50
35	1.0000	0.0000	1.0800	1.9400	0.0677	0.0412	0.0302		
36	2.0000	0.0000	0.0000	0.4700	0.5540	0.0000	0.0000	0.0000	
37	3.0000	0.0000	1.4000	0.7520	0.0253	0.0000	0.0000	0.0000	
38	4.0000	1.9900	1.0700	0.5790	0.0508	0.0643	0.0000	0.0000	
39	5.0000	7.9100	1.5900	0.9050	0.0530	0.0000	0.0000	0.0000	0.0000
40	6.0000	2.3700	0.9550	0.3430	0.0000	0.0359	0.0000	0.0000	0.0000
41	7.0000	18.5000	0.3040	0.1010	0.0000	0.0000	0.0000	0.0000	
42	8.0000	2.4600	0.3280	0.9380	0.0000	0.0000	0.0000	0.0000	
43	9.0000	14.2000	1.5700	0.6770	0.0000	0.0370	0.0000	0.0000	
44	10.0000	10.6000	3.6700	0.1750	0.0000	0.0000			
45									
46									
47	MEAN	5.8030	1.1967	0.6880	0.0751	0.0176	0.0034	0.0000	0.0000
48	STD	6.3063	0.9759	0.4967	0.1616 .1616	0.0.0230	0.0095	0.0000	0.0000
49	SE	1.9942	0.3086	0.1571	0.0511	0.0073	0.0030	0.0000	0.0000
50									
51	VOLUME X MILL	6.60	6.30	6.10	6.00	5.50	4.20	7.40	4.90
52	ABUND MILLIO	30.30	7.54	4.20	0.45	0.10	0.01	0.00	0.00
53	2 SE ABUND	26.32	3.69.	1.92	0.61	0.08	0.03	0.00	0.00
54									
55									
56									
57	LARGE TARGETS								
58	Transect	0-5	5-10	10-15	15-20	20-25	25-30	30-40	40-50
59	1.0000	0.0000	0.0000	1.4300	0.0000	0.0412	0.3020		
60	2.0000	0.0000	0.0000	0.5880	0.5540	0.0000	0.0000	0.0000	
61	3.0000	0.0000	0.0000	0.3260	0.0000	0.0000	0.0000	0.0000	
62	4.0000	0.0000	0.2670	0.5790	0.0508	0.0000	0.0000	0.0000	
63	5.0000	0.0000	0.0000	0.2010	0.0000	0.0000	0.0000	0.0000	0.0000
64	6.0000	0.0000	0.0000	0.1140	0.0000	0.0000	0.0000	0.0000	0.0000
65	7.0000	0.0000	0.0000	0.3030	0.0000	0.0000	0.0000		
66	8.0000	0.0000	0.0000	0.1170	0.0000	0.0000	0.0000	0.0000	
67	9.0000	0.0000	0.0000	0.0000	0.0000	0.0000			
68	10.0000	0.0000	0.0000	0.0000	0.0000	0.0000			
69									
70	mean	0.0000	0.0267	0.3658	0.0605	0.0041	0.0378	0.0000	0.0000
71	STD	0.0000	0.0801	0.4066	0.1652	0.0124	0.0999	0.0000	0.0000
72	SE	0.0000	0.0253	0.1286	0.0522	0.0039	0.0353	0.0000	0.0000
73									
74	VOLUME X MILL	6.60	6.30	6.10	6.00	5.50	4.20	7.40	4.90
75	ABUND X 1000	0.00	0.17	2.23	0.36	0.02	0.16	0.00	0.00
76	2 SE ABUND	0.00	0.32	1.57	0.63	0.04	0.30	0.00	0.00

	A	B	C	D	E	F
1						
2						
3	STANLEY LAKE SURVEY					
4	17-Sep-93					
5	Transect #1 is near boat ramp					
6	Transect #7 is near outflow					
7						
8						
9						
10	SMALL TARGETS					
11						
12	Transect	0-5	5-10	10-15	15-20	20-25
13	1.0000	11.6000	0.0000	0.1220	0.0745	0.0964
14	2.0000	2.8800	1.8700	0.3430	0.0629	0.0000
15	3.0000	2.5100	2.3500	0.0000	0.0000	0.0000
16	4.0000	4.6000	0.6230	0.2320	0.3230	0.0000
17	5.0000	2.4300	3.2800	0.2480	0.0000	0.0000
18	6.0000	5.1400	0.9380	0.0905	0.0000	0.0000
19	7.0000	32.5000	0.0000	0.0000		
20						
21						
22	MEAN	8.8086	1.2944	0.1479	0.0767	0.0161
23	STD	10.1119	1.1537	0.1212	0.1144	0.0359
24	SE	3.8219	0.4361	0.0458	0.0467	0.0147
25						
26	VOLUME MILLK	3.20	2.70	1.80	1.50	1.20
27	ABUNDANCE X	28.19	3.49	0.27	0.12	0.02
28	2 SE ABUND	24.46	2.35	0.16	0.14	0.04
29						
30						
31	MEDIUM TARGETS					
32						
33	Transect	0-5	5-10	10-15	15-20	20-25
34	1.0000	0.0000	0.0000	0.0774	0.0000	0.0000
35	2.0000	0.0000	1.2500	0.2300	0.0629	0.0000
36	3.0000	0.0000	1.3400	0.0000	0.0000	0.0000
37	4.0000	4.6000	3.7300	0.0000	0.0000	0.0000
38	5.0000	4.8700	0.9850	0.1240	0.1360	0.0000
39	6.0000	0.0000	0.4690	0.0905	0.0000	0.0748
40	7.0000	12.2000	1.6800	0.0000		
41						
42						
43	MEAN	3.0957	1.3506	0.0746	0.0332	0.0125
44	STD	4.2533	1.1029	0.0789	0.0514	0.0279
45	SE	1.6076	0.4169	0.0298	0.0194	0.0114
46						
47	VOLUME MILLK	3.20	2.70	1.80	1.50	1.20
48	ABUND X 1000	9.91	3.65	0.13	0.05	0.01
49	2 SE ABUND	10.29	2.25	0.11	0.06	0.03
50						
51						
52						
53	LARGE TARGETS					
54						
55	Transect	0-5	5-10	10-15	15-20	20-25
56	1.0000	0.0000	0.9600	0.2430	0.0000	0.0000
57	2.0000	0.0000	0.9340	0.2300	0.0000	0.0000
58	3.0000	0.0000	1.3400	0.1240	0.0000	0.0000
59	4.0000	0.0000	0.6230	0.0000	0.0000	0.0000
60	5.0000	0.0000	0.6560	0.1240	0.2710	0.0000
61	6.0000	0.0000	0.2350	0.0000	0.2240	0.0000
62	7.0000	0.0000	0.0000	0.0000		
63						
64						
65	MEAN	0.0000	0.6783	0.1030	0.0825	0.0000
66	STD	0.0000	0.4213	0.0989	0.1175	0.0000
67	SE	0.0000	0.1593	0.0374	0.0480	0.0000
68						
69						
70	VOLUME MILLK	3.20	2.70	1.80	1.50	1.20
71	ABUNDNACE X	0.00	1.83	0.19	0.12	0.00
72	2 SE ABUND	0.00	0.86	0.13	0.14	0.00