Snake River Sockeye Salmon Habitat and Limnological Research

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SNAKE RIVER SOCKEYE SALMON HABITAT AND LIMNOLOGICAL RESEARCH

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EXECUTIVE SUMMARY

In March of 1990, the Shoshone-Bannock Tribes petitioned the National Marine Fisheries Service (NMFS) to list the Snake River sockeye salmon (*Oncorhynchus nerka*) as endangered. As a result of that petition the Snake River sockeye salmon was officially listed as endangered in November 1991 under the Endangered Species Act (56 FR 58619). In 1991 the Snake River Sockeye Salmon Habitat and Limnological Research Program was implemented (Project Number 91-71, Intergovernmental Contract Number DE-BI79-91bp22548). This project is part of an inter-agency effort to save the Redfish Lake stock of *O. nerka* from extinction.

This report summarizes activities conducted by Shoshone-Bannock Tribal Fisheries Department personnel during the calendar year of 1998. Project objectives included; 1) monitor over-winter survival and emigration of juvenile anadromous *O. nerka* released from the captive rearing program into Pettit and Alturas lakes; 2) fertilize Redfish, Pettit, and Alturas lakes; 3) conduct kokanee (non-anadromous *O. nerka*) population surveys; 4) monitor spawning kokanee escapement and estimate fry recruitment on Fishhook, Alturas Lake, and Stanley Lake creeks; 5) control the number of spawning kokanee in Fishhook Creek; 6) evaluate potential competition and predation between stocked juvenile *O. nerka* and a variety of fish species in Redfish, Pettit, and Alturas lakes; 7) monitor limnological parameters of Sawtooth Valley lakes to assess lake productivity. Results by objective are summarized below.

Objective 1. The over-winter survival rate of stocked juvenile *O. nerka* in Pettit Lake was estimated using capture numbers at a weir on Pettit Lake Creek, the lake's outlet stream. During the summer of 1997, 8,634 pre-smolts from the captive rearing program were stocked into Pettit Lake. An estimated 950 smolts out-migrated from the lake during monitoring in the spring of 1998. This results in an estimated 11% over-winter survival rate. Alturas Lake was stocked with 95,000 pre-smolt *O. nerka* from the captive-rearing program during the summer and fall of 1997. An estimated 32,000 smolts emigrated from the lake in the spring of 1998, resulting in an estimated 34% over-winter survival rate. Over-winter survival and out-migration for Redfish Lake was monitored by IDFG.

Objective 2. Lake fertilization activities (addition of liquid nitrogen and phosphorous fertilizer) were continued in 1998 in order to elevate lake carrying capacities for juvenile sockeye. This was the fourth year that Redfish Lake was fertilized and the second year of fertilization for Pettit and Alturas lakes. Redfish Lake received 189.8 kg of total phosphorous (TP) and 3701.7 kg of total nitrogen (TN). Pettit Lake received 25 kg of TP and 465.1 kg of TN and Alturas Lake received 58 kg of TP and 1172.3 kg of TN.

Objective 3. During September 1998 fish densities were assessed using hydroacoustic sampling in Redfish, Pettit, Stanley, and Alturas lakes. Concurrent trawl sampling and density estimates were conducted by IDFG. A hydroacoustic estimate of *O. nerka* densities in the fall of 1998 for Redfish Lake was 107,613±33,615 fish/ha and a biomass of 2.5 kg/ha. Pettit Lake had an estimated 67,206±30,950 fish/ha and a biomass of 13.47 kg/ha. Alturas Lake had a fish density of 101,519±32,605 and a biomass of 2.09 kg/ha.

Objective 4. Stream spawner counts were used to monitor adult kokanee escapement to inlet streams on Redfish, Alturas, and Stanley lakes in 1998. Fishhook Creek, the primary kokanee spawning habitat in Redfish Lake, had an estimated spawning kokanee population of 6,149. This was the lowest escapement observed since monitoring began. Stanley Lake Creek had an estimated 783 kokanee. Alturas Lake Creek had an estimated 15,273 fish, the highest number recorded for any kokanee spawning population since monitoring began. Fry recruitment, calculated from sex ratios, fecundity, and egg to fry survival rates is estimated at 37,500, 7,634, and 218,400 fry for Fishhook, Stanley Lake, and Alturas Lake creeks respectively.

Objective 5. A weir was set up on Fishhook Creek to control kokanee escapement. The weir was only partly effective and failed to contain all fish. No fish were culled during this activity. No further control effort was conducted.

Objective 6. Potential competition and predation interactions between stocked sockeye salmon (anadromous *O. nerka*), rainbow trout (*O. mykiss*), and other fish species were investigated. In an analysis of rainbow trout diets there were no *O. nerka* found in the guts of any of the fish sampled. Diet overlap was 30% for rainbow trout and *O. nerka* consisting of chironomid pupae. Age 0 sockeye salmon, the life stage of primary interest, fed primarily on zooplankton while rainbow trout had a diet dominated by insects. Several potential sockeye/kokanee predators were identified in the lakes including bull trout (*Salvelinus confluentus*), northern pikeminnow (*Ptychocheilus oregonensis*), and brook trout (*S. fontinalis*). No salmonids including kokanee/sockeye were found in the stomachs of any of the northern pikeminnow or brook trout. Piscivory was evident however, with cyprinids found in the diets of both species. Bull trout diet was composed entirely of salmonids, none of which were identified as *O. nerka*. However many were unidentifiable due to the progressed state of digestion.

Objective 7. Limnological parameters including nutrient levels, chlorophyll *a*, secchi depth, primary productivity, phytoplankton, and zooplankton assemblage characteristics (species composition and densities) were monitored. There was a general trend of elevated productivity parameters; demonstrating potential increased lake carrying capacity.

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CHAPTER 1: FISHERIES INVESTIGATIONS ON THE SAWTOOTH VALLEY

Salmon of the Snake River Basin are a valuable cultural resource to the Shoshone-Bannock Tribes. The Shoshone-Bannock Tribes (SBT) traditionally utilized salmon of the Snake River Basin as a subsistence food resource. The Redfish Lake sockeye salmon is the only extant Snake River stock of *O. nerka*. The spawning and rearing habitat of this stock is located in the Sawtooth Valley, a traditional SBT fishing and hunting area. In March of 1990, the SBT petitioned the National Marine Fisheries Service (NMFS) to list the Snake River sockeye salmon as endangered. As a result of that petition the Snake River sockeye salmon was officially listed as endangered in November 1991 under the Endangered Species Act (56 FR 58619). The Endangered Snake River Sockeye Salmon Recovery Program was implemented that same year. The SBT have been actively involved in the sockeye salmon recovery project (BPA Project Number 91-71, Intergovernmental Contract Number DE-BI79-91BP22548) since it's inception.

The Bonneville Power Administration (BPA) provides funding for this inter-agency recovery program through the NPPCFWP. Cooperators in the recovery program include the National Marine Fisheries Service (NMFS), Idaho Department of Fish and Game (IDFG), the University of Idaho (UI), U. S. Forest Service (USFS), and the Shoshone-Bannock Tribe (SBT). The NMFS manages the permitting of activities and the captive rearing program hatchery operations in Manchester, WA. The IDFG monitors a variety of fisheries parameters in the field and is responsible for the captive rearing hatchery operations in Eagle and Stanley, ID. The UI conducts genetic research and management. The U. S. Forest Service participates in permitting activities and habitat improvements. The SBT evaluates a variety of rearing habitat characteristics in nursery lakes that are detailed in this report.

In 1991, only four adult sockeye returned to Redfish Lake. These four fish and emigrating juveniles captured over the next two years formed the initial captive brood stock. The captive brood stock was supplemented with returning adult fish and residual sockeye in subsequent years. Historically, thousands of sockeye returned to the Sawtooth Valley Lakes. Everman (1896) reported that the lakes were 'teeming with redfish'. In 1910 anadromous fish migration was blocked when the Sunbeam Dam was built on the mainstem of the Salmon River approximately 20 miles downstream from the Sawtooth Valley. In 1934 the dam was breached and upstream anadromous fish populations appear to have rebounded. Bjorn (1968) estimated that 4,360 sockeye returned to Redfish Lake in 1955. There has been a steady decline in adult sockeye returns since that time. In the late 1980's, only a small number of fish were returning to Redfish Lake. Sixteen adult sockeye have returned since 1991 including a single male fish in 1998. The recovery program has focused its efforts on restoring anadromous *O. nerka* to Redfish, Alturas, and Pettit lakes, which were designated as critical spawning and rearing habitat under the ESA listing (56 FR 58619).

A variety of activities have been conducted since the inception of the program in the effort to conserve and rebuild the Redfish Lake *O. nerka* ESU. The captive brood stock has served to preserve the unique genome. Fish barriers on Alturas and Pettit lake creeks have been removed to facilitate fish passage. Sockeye from the captive brood stock have been reintroduced into the wild. A variety of stocking strategies have been implemented and evaluated including; adult releases for volitional spawning, in lake egg incubators, smolt releases, net pen rearing with pre-smolt release, and pre-smolt releases during spring, summer, and fall. Lake fertilization has also been implemented in order to increase lake carrying capacities. Kokanee (non-anadromous form of *O. nerka*) control measures have been implemented to reduce intra-specific competition. A variety of fishery and limnological parameters have been monitored in association with these strategies.

The Technical Oversight Committee (TOC) has guided all activities conducted by the SBT in association with the sockeye recovery project. The TOC is composed of representatives of all participating agencies (BPA, NMFS, IDFG, UI, and SBT). The TOC was formed in 1991 to guide new research, coordinate ongoing research, and actively participate in all elements of the Snake River sockeye recovery effort. The project as a whole or in part is subject to further review by the Idaho Department of Environmental Quality (DEQ), the United States Forest Service (USFS), and the NPPCFWP Independent Scientific Review Panel (ISRP).

STUDY AREA

Four lakes in the Sawtooth Valley, Redfish, Alturas, Pettit, and Stanley lakes, are currently the focus of on going SBT habitat and limnological studies. The lakes were glacially formed, range in elevation from 1,985 m to 2,138 m, and are located in central Idaho (Figure 1). Specific features of the sockeye rearing lakes are shown in Table 1.

Table 1. Morphological features of the Sawtooth Valley lakes.

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

All of the Stanley Basin lakes are oligotrophic. Mean summer total phosphorous (TP) concentrations in the epilimnion range from 4.9 to 11.8 *u*g/l. Seasonal mean epilimnetic chlorophyll *a* concentrations range from 0.3 to 2.3 *u*g/l. Mean summer secchi disk transparency range from 9.6 - 15.2 m, excluding Stanley Lake, which ranged from 5.0-8.2 m.

The Sawtooth Valley lakes are approximately 1,440 kilometers from the mouth of the Columbia River. There are 616 kilometers of free flowing river from Redfish Lake to the mouth of the Salmon River (Figure 1) and an additional 835 km with 8 dams on the Snake and Columbia rivers.

Native fish species found in the nursery lake system include sockeye/kokanee salmon (*Oncorhynchus nerka*), steelhead/rainbow trout (*O. mykiss*), chinook salmon (*O. tshawytscha*), cutthroat trout (*O. clarki lewisi*), bull trout (*Salvelinus confluentus*), mountain whitefish (*Prosopium williamsoni*), sucker (*Catastomus sp*.), redside shiner (*Richardsonius balteatus*), dace (*Rhinichthys sp*.), northern pikeminnow (*Ptychocheilus oregonensis*), and sculpin (*Cottus sp*.). Non-native species include brook trout (*S. fontinalis*) and lake trout (*S. namaycush*). The

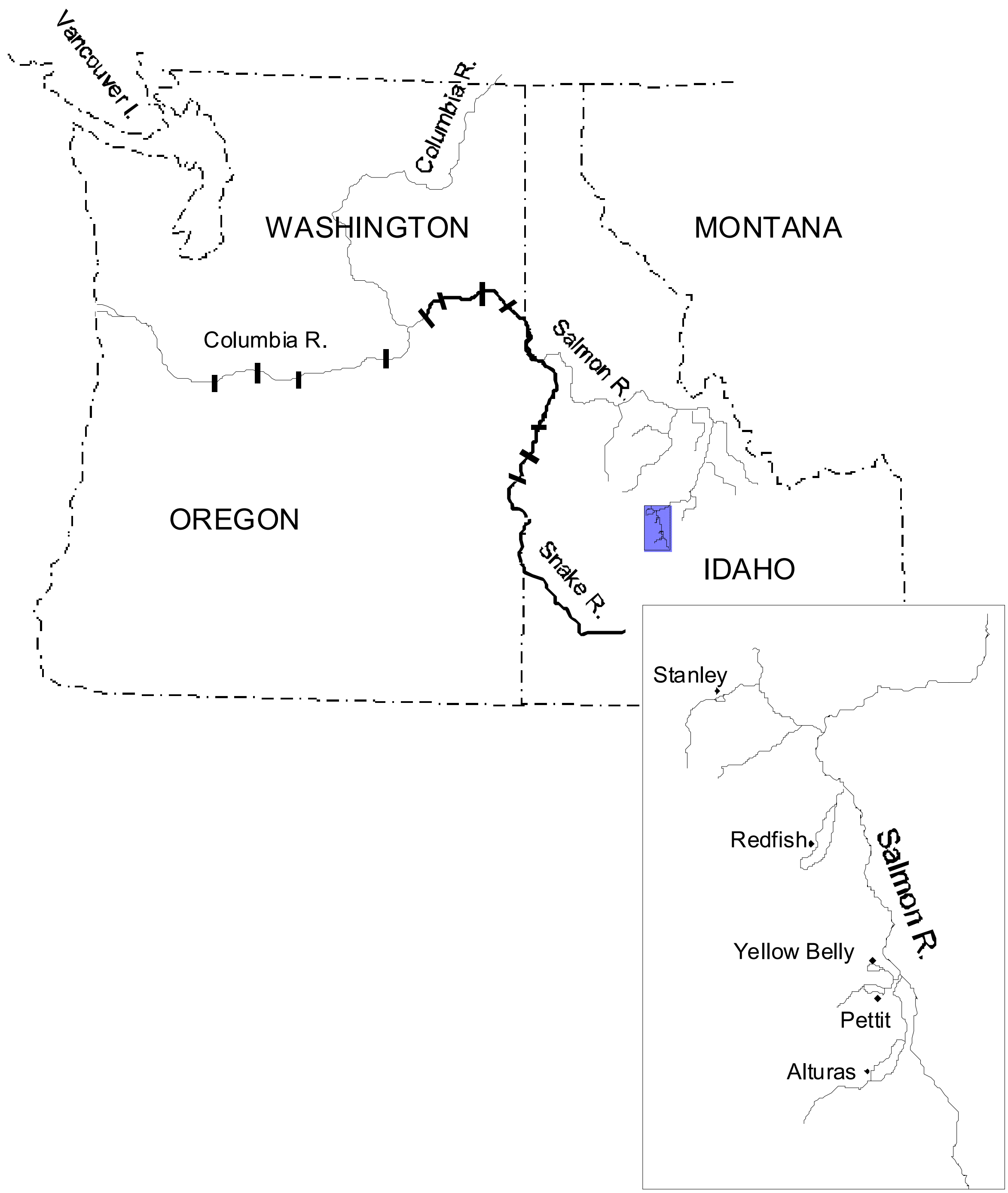

Figure 1. Map of study area.

only pelagic species besides *O. nerka* are redside shiners. The two species are not sympatric because of differing vertical distributions. Hatchery rainbow trout are stocked by IDFG throughout the summer in all lakes except for Redfish Lake. Sport fishing for salmonids is open on all lakes as well as inlet and outlet streams with a restricted season for Redfish Lake and Fishhook creek.

There are several different forms of *O. nerka* that are found in the Sawtooth Valley lakes. Kokanee, a non-anadromous form of *O. nerka*, spends its entire life cycle in the fresh water lakes. *O. nerka* are the primary pelagic zooplanktivore in the Sawtooth Valley Lakes. Kokanee generally spawn at four years of age in the inlet creeks of the lakes during late summer and die afterwards. Kokanee in Redfish, Stanley, and Pettit lakes are from introduced stocks and are genetically different from the anadromous form. Alturas Lake kokanee have similar life history pattern to the historic anadromous *O. nerka*. No Sawtooth Valley kokanee are listed as endangered.

Residuals are another form of *O. nerka* found in only Redfish Lake and listed as part of the ESU. The residual population remains in freshwater for their entire life cycle, yet are genetically similar to the anadromous *O. nerka* form. The residual population spawns at the same time as the anadromous form and, similar to the anadromous form, creates its redds on the lake shore instead of the inlet creeks.

The anadromous form of *O. nerka* spends one or two years in freshwater, emigrating during high spring flows as age 1 or 2 smolts. The anadromous form spends the majority of its life in the Pacific Ocean, generally returning at three-plus years of age to the Sawtooth Valley lakes.

Similar to many species of salmon some anadromous *O. nerka* return as two-plus year olds, which are referred to as a jack or jill, depending on sex. This form is generally a beach spawner, although there are historic accounts of stream spawning and individuals have been sighted in streams during recovery program monitoring. The anadromous and residual forms have been designated as an evolutionarily significant unit (ESU).

MATERIALS AND METHODS

Hydroacoustic population estimates

Data acquisition. - Echo sounding data were collected with a Hydroacoustic Technology, Inc. Model 240 split-beam system. Split-beam echosounders have been shown to have less variability for target strength estimates than dual-beam systems (Traynor and Ehrenberg, 1990) and the target tracking capabilities of the split-beam system further reduce variability of individual targets (Ehrenberg and Torkelson, 1996). We used a 15 degree transducer, and the echo-sounder criteria were set to a pulse width of 0.4 milliseconds, a time varied gain of 40 log(R) + 2 r, and four pings per second for Redfish lakes, and five pings per second for Alturas and Pettit Lake. A minimum of five pings per target was necessary to qualify as a fish target. Data were recorded on a Panasonic SV-3700 digital audio tape recorder.

Identical transects were followed using a global positioning system (GPS) during 1994 (Teuscher and Taki 1995). Waypoints were established to allow for sampling transects to run zigzag across all lakes except Pettit Lake, where we used five parallel and one diagonal transect (Teuscher and

Taki 1995). We sampled twelve and fourteen transects at Alturas and Redfish lakes, respectively.

Surveys were conducted during two moonless nights in October. We began at approximately one and one half-hours after sunset. Boat speed during data collection ranged from 1-1.5 m/s.

Vertical gill netting and trawling (by IDFG) were done concurrently with hydroacoustic sampling. Vertical gill net sampling was used to assist in partitioning targets in Pettit Lake since past trawling efforts have indicated a positive selection for kokanee. Vertical gillnets were used to determine if other fish species were found in the pelagic areas during sampling. Previous gill net sampling conducted in Alturas Lake has not yielded sufficient numbers for partitioning targets and therefore was not used. Due to NMFS Section 10 permit limitations vertical gillnet sampling in Redfish Lake was not conducted.

Data analysis. - Target strengths and fish densities were processed using a Model 340 Digital Echo Processor and plotted with a Model 402 Digital Chart Recorder. Target strengths were used to estimate fish length by the equation

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

developed by Love (1977) where TS = target strength in decibels, L = fork length in centimeters, and F = frequency of transmitted sound (kHz). Fish density estimates were calculated for different size classes for each lake to approximate cohort densities based on 1998 length frequency distributions and age analyses performed by the IDFG from fish captured in the trawl.

Four different size classes were used for all three lakes. Due to overlap in Alturas Lake, we combined the III+ and IV+ kokanee cohorts. Total abundance and vertical distribution were also estimated.

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

The effective beamwidth equation

$$X[ABS\ (M^{RS} - F^{TS})]^{Y} \qquad (2)$$

was used where X = 8.6, ABS = absolute value of the target strength remainder, M^{RS} = minimum system detection (-60), F^{TS} = mean target strength, and Y = 0.47 (P. Nealson, HTI, personal communication).

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

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

and variance was estimated by

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

where D_i = mean density (number/m^2) in stratum i, D_{ij} = mean density for the jth transect in stratum i, L_i = length of transect j, and T_i = number of transects surveyed in stratum i (Gunderson, 1993).

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

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

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

Correlation analysis was used to compare trawl versus hydroacoustic population estimates. Comparisons were made of the previous year's results for total lake populations and cohort estimates.

Smolt Monitoring

Pettit Lake

A weir was operated at the outlet of Pettit Lake, Idaho from 20 April through 26 May 1998. The weir was used to monitor over-winter survival and emigration of Snake River sockeye salmon smolts. In 1997, 8,634 fish were introduced as pre-smolts from the captive broodstock program into Pettit Lake. The weir ran continuously during operation at 100% capture efficiency. Shoshone-Bannock Tribal fisheries personnel checked for fish and cleaned the weir at sunrise and sunset. The trap was checked more frequently when high levels of debris were present. Sampling was discontinued when peak flows associated with spring run-off created logistical problems.

Immediately after removal from the trap, fish were anesthetized using a stock solution of 15 grams of MS222 and 30 grams of sodium bicarbonate per liter of water. All fish that were anesthetized were weighed to the nearest 0.1 grams and measured (fork length) to the nearest millimeter and were held in a live well for five to ten hours after handling and then released. All other fish were counted and immediately released below the weir.

Alturas Lake

A screw trap was operated on the outlet stream of Alturas Lake, Idaho from 21 April through 12 June 1998. The screw trap captured fish to estimate over-winter survival and smolt emigration, and to allow tagging of Snake River sockeye salmon smolts using passive induced transponders (PIT tags). There were 94,746 pre-smolts sockeye introduced from the captive broodstock program into Alturas Lake in 1997. Shoshone-Bannock Tribal fisheries personnel checked for fish and cleaned the screw trap at sunrise and sunset. For one week during peak run-off we checked and cleaned the trap at approximately six-hour intervals during the night to prevent debris accumulation. Trap efficiency was calculated for five periods using a mark recapture method. Trap efficiency was calculated using equation 6.

$$\text{Efficiency} = \text{number marked fish recaptured} \,/\, \text{number marked fish released} \qquad (6)$$

Three different periods were sampled for efficiency based on discharge and two more periods when the trap was relocated to reduce capture of fish. Discharge was not measured at the screw trap but was monitored with a staff gauge approximately 300 meters upstream. A condition factor (K value, $(\text{weight} \times 10^5)/(\text{length})^3$) for each fish was estimated, mean, minimum, and maximum K value are presented in results.

All fish that were anesthetized were weighed to the nearest 0.1 grams and measured (fork length) to the nearest millimeter. Fish that were PIT tagged on site were held in a live well for five to ten hours after handling and then released. PIT tags and needles were sterilized in 70% ethanol. PIT tagged fish used in the mark recapture capture efficiency estimation were held in a live well and released at dusk approximately 300 meters upstream. All other fish were counted and immediately released below the screw trap.

Growth Rates

Growth rates of stocked juvenile sockeye were compared in an effort to evaluate differences in fitness associated with differences between lakes and the various stocking strategies. Length data (in mm) of stocked *O. nerka* were collected at the time of tagging and smolt emigration. Length data were Log_{10} transformed to normalize and create a linear relationship from an exponential growth curve to facilitate growth rate comparisons. Log_{10} transformed lengths at the time of tagging were compared to Log_{10} transformed length data of sockeye collected during smolt emigration monitoring. An initial pair-wise comparison (t-test) was conducted comparing size at release and size at emigration between lakes and release strategy. A second descriptive comparison was generated using a linear regression between the Log_{10} transformed lengths of pre-smolts and smolts for each release group. The slope of the regression line represents the growth rate of each group and was used to compare the growth rates of fish between lakes and release strategies.

The summer release strategy group of fish spends a longer amount of time in the lakes than the fall release strategy group. In order to eliminate potential differences in comparisons associated with this time in lake factor a millimeters of growth per cumulative temperature unit (mm/CTU) was generated for each group. The mm/CTU for each group of fish was calculated using (L_1) - (L_0) / CTU. Where (L_0) is the mean length at the time of tagging, (L_1) is the mean length at time of smolt emigration and, CTU is degrees (C) estimated for each day summed over the time of examination. Temperature data from each lake was used to calculate the mean monthly temperature for the 0-10 meter depth in each lake. The mean monthly temperature was then used as the daily temperature for each month in calculating CTU's. Temperatures for periods during ice-over (January through mid-May) ranged from 4-5 ^{o}C. A temperature of 4.5 ^{o}C was used in the generation of CTU's for periods of ice-over. Temperature data was not collected between late October and January. Temperatures for November and December were estimated by taking the difference between October temperatures and January temperatures divided by three, this number was subtracted from the prior mean monthly temperature.

Gillnet

Horizontal and vertical gillnet sampling was conducted to quantify fish population characteristics including species composition, habitat utilization (pelagic versus littoral), and diet analysis. Horizontal gillnets (30 m long, 1.8 m high) with lead sinking lines composed of five panels 6 m long of graduated mesh size (1.24, 2.54, 3.17, 5.08, and 6.35 cm) were set at selected points along the bank perpendicular to shore on Pettit Lake. Nets were set with the smallest mesh size panel closest to shore and the largest mesh size panel set deeper and further from shore. There were four vertical gillnets of graduated mesh sizes (1.25, 2.54, 3.17, 5.08, and 6.35 cm), each net was 3 m wide and 30m deep and. The four vertical gillnets of graduated mesh size were set in a line at a single station in the pelagic zone in the middle of Pettit and Alturas lakes. Horizontal and vertical gillnet sampling was conducted on 16 June and 29 September 1998 in Pettit Lake. Vertical gillnet sampling was conducted once in Alturas Lake on 27 September 1998. Vertical gill net sampling was also used to assist in partitioning targets in Pettit Lake since past trawling efforts have indicated a selectivity for larger, older age class *O. nerka*. Therefore, we employed vertical gillnets to determine if other *O. nerka* age classes and/or fish species were found in the pelagic areas during sampling. Due to NMFS section 10 permit limitations there were no gillnets set in Redfish Lake.

Stream Spawning

Stream surveys were conducted to estimate spawning kokanee abundance in Fishhook Creek (primary kokanee spawning habitat to Redfish Lake), Stanley, and Alturas lake tributary streams. Pettit Lake has no identified stream spawning kokanee population. Fish were counted from the bank by one or two observers equipped with polarized sunglasses. The number of fish in the stream, on days when counts were missed, was interpolated by dividing the difference between the actual counts by the number of days between the counts. Total escapement estimates were calculated by summing daily counts of kokanee and dividing by average stream life as described by English et al. (1992).

Beach Spawning

Sockeye Beach, located near the boat ramp, and a small section of the southeast corner of Redfish Lake are spawning grounds for residual and adult sockeye. Night snorkel surveys were conducted to estimate the relative abundance of residual spawners and adult anadromous sockeye stocked from the captive-rearing program in both locations. Snorkel surveys in Redfish Lake were conducted weekly on 4 nights from 6 October to 3 November 1998. At least three observers, equipped with waterproof flashlights, snorkeled parallel to shore 10 m apart, at depths ranging from 0.5 to 5 m. At Sockeye Beach, estimates of residual spawner abundance were conducted within the boundary (600 m) of Sockeye Beach as delineated by USFS signs. Spawning ground surveys in the south end of the lake were conducted in the 200 m shoal area section near the two southeast inlet streams.

Diet Analysis

Fish stomachs collected from gillnet, trawl, screw trap, and weir operations were examined to determine diet composition. Stomach samples from rainbow trout, bull trout, brook trout, northern pike minnow, and kokanee were collected. Fish were measured (fork length to the nearest millimeter) and weighed (to the nearest 0.1 gram) after which stomachs were removed and placed in 70% ethanol. Prey were identified, enumerated, blotted dry, and weighed to the nearest 0.01 g. Zooplankton were enumerated and lengths were derived from zooplankton tows performed on same sampling months. Zooplankton lengths were converted to dry weight using the length-weight relationship reported in McCauley (1984). Aggregate percent of diet by dry weight for all species of fish sampled was calculated (Swanson et al. 1974). Aggregate percent by dry weight totals were used to determine diet overlap and electivity indices. Diet overlap indices for *O. nerka* and other species captured were calculated using equations described by Koenings et al. (1987). Electivity indices (Ivlev 1961) describing calculations for prey preferences were used for *O. nerka*.

RESULTS

Hydroacoustics

Hydroacoustic population estimates of *O. nerka* during October of 1998 ranged from 107,613 to 67,206 fish in Redfish and Pettit lakes, respectively (Table 2). The greatest annual change in the three lakes occurred in Alturas Lake where the population increased over 300% from 30,795 in 1997 to 107,613 in 1998. This coincided with the largest observed kokanee escapement in Alturas Lake Creek. In 1997 an estimated 8,500 kokanee spawned in Alturas Lake Creek which resulted in an estimated 92,700 kokanee fry recruiting to the lake in the spring of 1998 (Taki et al. 1999). Although Redfish Lake had the largest whole lake population of *O. nerka*, it had the lowest density of

Table 2. *O. nerka* abundance and density estimates for three Sawtooth Valley lakes from 1992 through 1998. A subcontractor with different sampling equipment took 1992 and 1993 estimates.

Lake	Year	Population Estimate	Density (fish/hectare)
Redfish	1998	107,613 ± 33,615	175.0 ± 54.6
Redfish	1997	131,513 ± 32,319	213.8 ± 52.5
Redfish	1996	66,325 ± 24,000	107.8 ± 39.0
Redfish	1995	103,570 ± 24,500	168.4 ± 39.8
Redfish	1994	133,360	216.80
Redfish	1993	203,500	
Redfish	1992	188,000	
Pettit	1998	67,206 ± 30,950	414.9 ± 191.1
Pettit	1997	63,195 ± 29,581	390.1 ± 182.6
Pettit	1996	77,680 ± 15,850	479.5 ± 97.8
Pettit	1995	77,765 ± 46,900	480.0 ± 289.5
Pettit	1994	12,265 ± 8.360	75.7 ± 51.6
Pettit	1993	20,400	
Pettit	1992	19,000	
Alturas	1998	101,519 ± 32,605	300.4 ± 96.4
Alturas	1997	30,795 ± 5,869	91.1 ± 17.4
Alturas	1996	20,620 ± 4,140	61.0 ± 12.3
Alturas	1995	32,260 ± 5,090	95.4 ± 15.1
Alturas	1994	10,980 ± 1,090	32.5 ± 3.2
Alturas	1993	200,700	
Alturas	1992	144,000	

175 ± 55 fish/ha. Pettit Lake remained similar to 1997 with 415 ± 191 fish/hectare, and Alturas Lake increased from 91 to 300 ± 96 fish/ha from 1997 (Table 2).

In 1998 we also surveyed Stanley Lake for the first time in three years. Total population estimate for Stanley Lake was 12,857 kokanee 158 fish/ha. Kokanee escapement in Stanley Lake Creek in 1997 was similar to 1994 (Taki et al. 1999).

Redfish Lake- The total *O. nerka* population in Redfish Lake was 82% of the 1997 estimate. The decrease was seen in every cohort except for the YOY cohort which increased (Table 3). In 1997 there were eighty adult sockeye released into the lake for volitional spawning and 85,378 fertilized eggs were put into in lake incubation boxes. The progeny from these activities may account for the increase in YOY cohort.

Table 3. Hydroacoustic population estimates by age class for Redfish Lake, 1994 – 1998.

Cohort	1994	1995	1996	1997	1998
0+	76,600 ± 19,560	22,360 ± 6,410	12,680 ± 5,030	37,234 ± 14,449	46,747 ± 19,155
I+	36,000 ± 8,240	49,120 ± 12,400	34,950 ± 21,040	51,681 ± 14,533	27,767 ± 10,955
II+	20,760 ± 7,470	31,070 ± 12,340	18,700 ± 4,570	30,623 ± 6,599	12,450 ± 5,215
III+				11,973 ± 3,104	9,926 ± 5,629

Because we are not allowed to set vertical gill nets in Redfish Lake we assume every fish tracked in the pelagic zone is an *O. nerka,* and that no *O. nerka* are in the littoral zone when we sample. Under that assumption, annual survival rates of the one-year-old cohort to their second year (24%) was poor when compared to previous years (Table 3). Caution should be used when comparing survival rates based on the hydroacoustic population estimate because of the large confidence intervals in age class estimates. Annual survival rates of the two-year-old cohort to the following year cannot be made because of the unknown proportion of three-year-olds that spawn before sampling is conducted. The same can be said for survival from YOY to yearling because of the unknown contribution of stocked sockeye to the YOY population.

Pettit Lake- The total *O. nerka* population in Pettit Lake rose slightly compared to 1997 although the large confidence intervals suggest the population remained stable (Tables 2 and 4).

Table 4. Hydroacoustic population estimates by age class for Pettit Lake, 1995– 1998.

Cohort	1995	1996	1997	1998
0+	2,880 ± 1,270	4,740 ± 3,020	4,471 ± 2,705	16,593 ± 6,548
I+	15,600 ± 9,330	17,890 ± 3,020	14,061 ± 6,010	17,027 ± 9,963
II+	37,270 ± 23,570	31,800 ± 5,820	23,635 ± 11,485	29,974 ± 15,704
III+	19,667 ± 13,930	23,247 ± 5,100	21,027 ± 11,502	7,895 ± 4,567

Results from hydroacoustic sampling in Pettit Lake have always given confounding results when each cohort is examined. YOY and yearlings are underrepresented. Concurrent trawling results are similar. No YOY have been caught in the trawl during the years 1995-1998 and yearlings were captured in 1995 and 1996 but not in the last two years. In 1996 only one yearling was captured (personal communication, J. Pravecek, IDFG).

During previous surveys we partitioned rainbow trout from the population estimate based on the size of the largest kokanee captured in the trawl. During 1997 IDFG stocked rainbow trout that overlapped in size with the largest kokanee. Because of this size overlap, in 1998 we did not include any hydroacoustic targets in the littoral that were within four meters of the bottom and greater than 185 mm in length. This depth was chosen based on SCUBA observations and gill net data related to rainbow trout habitat utilization in Pettit Lake during 1993 and 1994. Elimination of these targets reduced the estimate of the three-year-old cohort compared to 1997 (Table 4).

Alturas Lake- Whole lake *O. nerka* population estimates increased more than three times the 1997 estimate (Table 2). The YOY contribution to this estimate is higher than the whole lake population of all age classes combined has been since 1994 (Table 5). This large cohort is a result of the high kokanee escapement observed in 1997 (8,500 fish) in Alturas Lake Creek (Taki et al. 1999). Prior to this project implementation, Alturas Lake had a total lake

Table 5. Hydroacoustic population estimates by age class for Alturas Lake, 1996– 1998.

Cohort	1996	1997	1998
0+	3,255 ± 1,490	4,330 ± 979	73,176 ± 27,411
I+	7,670 ± 3,175	11,859 ± 3,071	20,106 ± 5,372
II+	4,665 ± 635	4,304 ± 1,149	6,399 ± 1,734
III+	3,702 ± 1,300	10,775 ± 2,920	1,838 ± 1,297
IV+	1,260 ± 785	0	0

population of 126,644 kokanee base on a trawl survey (Kline and Lamansky 1997) in 1990. Since that year there has been a zooplankton collapse (Griswold 1997) that led to a low population estimate of 10, 980 kokanee in 1994 (Taki and Mikkelsen 1997). As the zooplankton have rebounded (Taki et al. 1999) so has the kokanee population. Continued high numbers of kokanee in this lake represent potential competition and could adversely affect rearing conditions for sockeye pre-smolts introduced from the broodstock program.

Hydroacoustic/trawl comparisons

Hydroacoustic population estimates were greater in all three lakes than the trawl estimates during 1998 (Figure 2). Hydroacoustic estimates have been consistently higher than trawl estimates since this project began (Taki et al. 1999, Taki and Mikkelsen 1997). The hydroacoustic versus trawl ratio in Redfish Lake was much

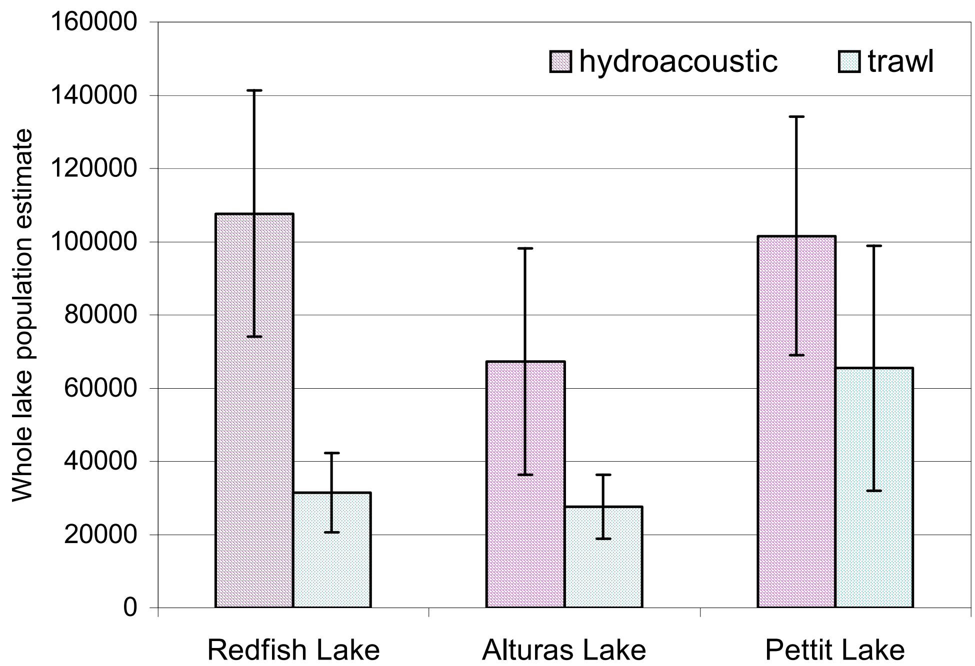

Figure 2. 1998 hydroacoustic and trawl *O. nerka* population estimates for three Sawtooth Valley lakes.

higher in 1998 at 3.41:1 than the mean over the previous four years (1.95:1). Redfish Lake was the only lake where the 95% confidence intervals did not overlap. Alturas Lake also had a higher hydroacoustic/trawl ratio in 1998 (2.43:1) than the previous four year mean (1.77), although the 95% confidence intervals slightly overlapped. Pettit Lake did not follow the same trend as the other two lakes as its ratio was lower than the previous years mean (1998 = 1.55:1, four year mean = 2.08). Redfish Lake was the only lake that did not fall within the bounds of other comparisons of hydroacoustic and trawl population estimates (Parkinson et al. 1994).

Comparing the total lake *O. nerka* fish/m^2 correlation between hydroacoustic and trawl estimates reveals Alturas Lake is more closely related than Pettit Lake, with Redfish Lake intermediate (Figure 3). The inability of the trawl to capture younger cohorts in Pettit Lake may contribute to this.

We combined lakes to look for a relationship between cohorts. The YOY cohort has an R-value of 0.83 (Figure 4), even though the trawl does not often capture fish from this cohort. There is also no significant difference between the means for this cohort ($P = 0.22$). The yearling cohort showed no relationship at all. Because of the low R value (0.07) and a showing of significant difference between the means ($P < 0.001$) it was not graphically illustrated. We found the closest relationship between hydroacoustic and trawl population estimates for the II+ cohorts (R = 0.98) (Figure 5). This was the only cohort that the trawl has captured every year. Hydroacoustic and trawl estimates for age II *O. nerka* were not significantly different (P > 0.99).

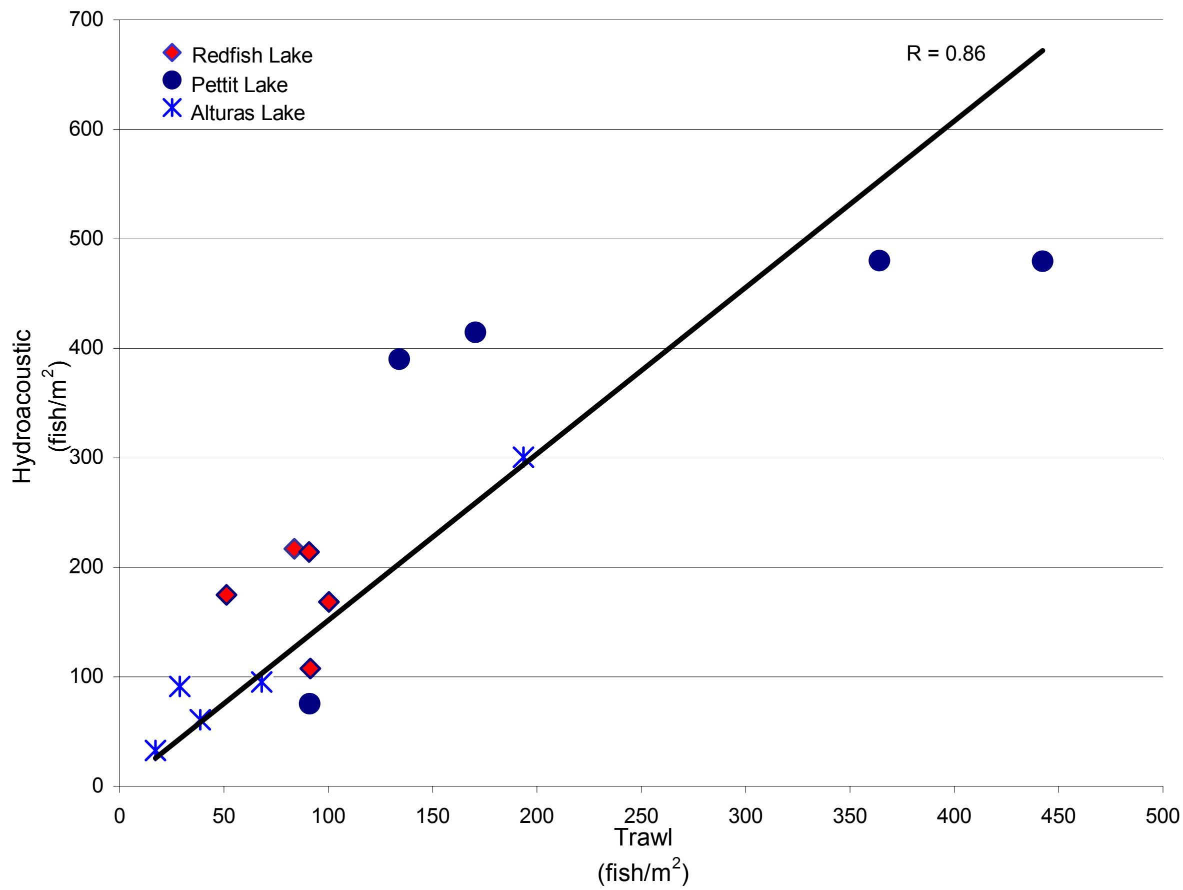

Figure 3. Correlation of total lake *O. nerka* population estimates between hydroacoustic and trawl surveys.

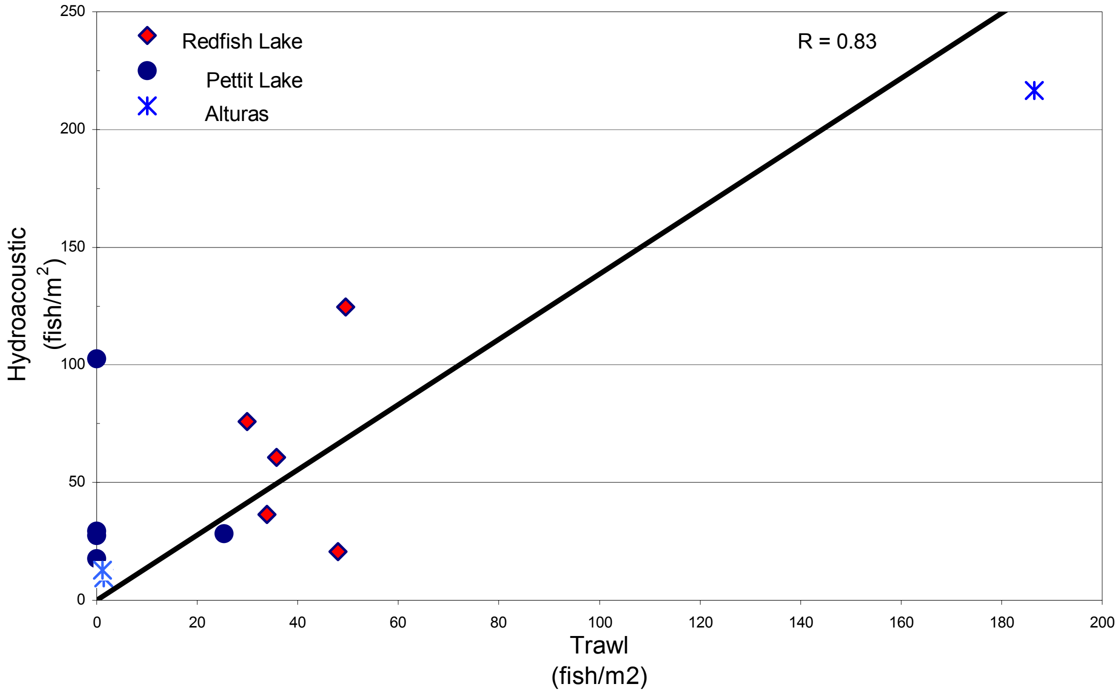

Figure 4. Correlation of YOY *O. nerka* population estimates between hydroacoustic and trawl.

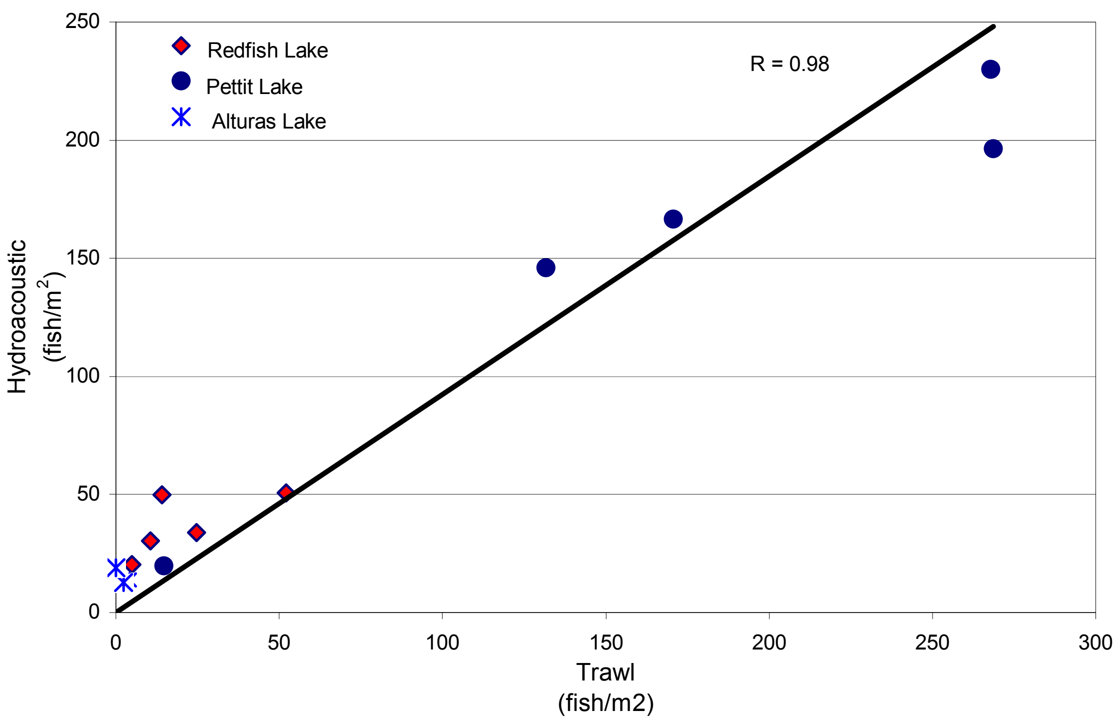

Figure 5. Correlation of II+ *O. nerka* population estimates between hydroacoustic and trawl population samples.

Smolt Monitoring

Pettit Lake Creek

There were 8,634 Snake River sockeye pre-smolts stocked into Pettit Lake in 1997 of which 15.4% (1,336 fish) were PIT tagged. Forty-one of these PIT tagged fish were interrogated at down stream dams. Dam interrogation had a 56% capture efficiency (J. Pravecek IDFG personal communication) resulting in an estimated 73 PIT tagged fish from Pettit Lake which survived to the dam and an estimated 155 PIT tagged fish that emigrated from Pettit Lake (back-calculated from a 47% lake to dam survival rate). One hundred fifty-five fish represents an 11.6% over-winter survival of the original 1,336 PIT tagged pre-smolts. At that survival rate there were approximately 1000 total sockeye smolts that emigrated from Pettit Lake.

The mean fork length of *O. nerka* captured at the weir (Figure 6) was 129 mm (range 106-165 mm), mean weight 18.71 g (range 11.6- 25.96 g), and mean K value (condition factor) of 0.87 (range 0.38-1.27). During weir operation there were three indirect mortalities of unknown cause of listed fish. Although no chinook salmon redds have been documented above the weir during the last several years, ten wild chinook salmon parr with an average length of 95.5 mm were captured at the weir. Other fish species captured included redside shiners (*Richardsonius balteatus*), mountain whitefish, brook trout, bull trout, and northern pikeminnow.

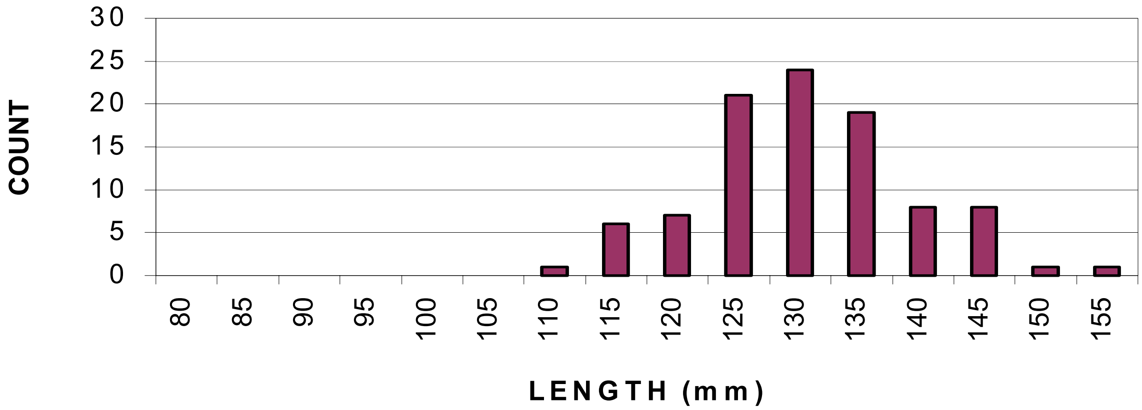

Figure 6. Pettit Lake *O. nerka* smolt length frequency.

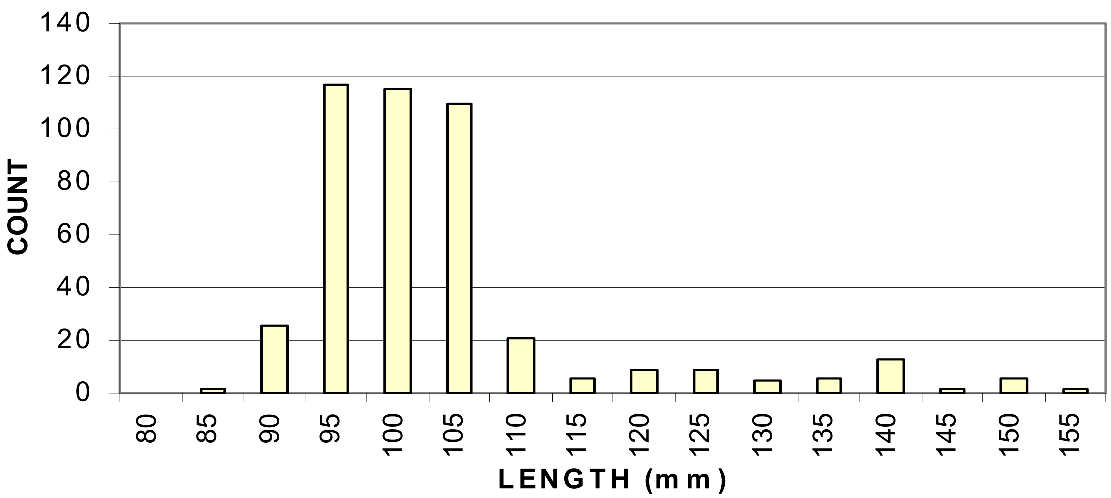

Figure 7. Alturas Lake *O. nerka* smolt length frequency.

All sockeye salmon smolts were captured from 2 May through 26 May 1998. Down river PIT tag interrogations showed that the majority of smolt emigration from Pettit Lake occurred after the weir had been removed on 26 May 1998. The weir removal was necessary due to high flows, which threatened to over-top the weir. Water temperatures at the weir ranged from 1.5° to 12° C and discharge from less than .85 to greater than 4.24 m^3/sec.

Alturas Lake Creek

Screw trap efficiencies ranged from 11% to 60%, with a season mean of 25.8%. Using the number of captured sockeye smolts integrated over the five calculated efficiency periods there were an estimated total of 32,174 sockeye smolts emigrating from Alturas Lake. This results in an estimated 34% over-winter survival/migration rate for the Snake River sockeye pre-smolts introduced from the captive broodstock to Alturas Lake in 1997. The actual number is potentially higher due to additional emigration that occurred after the trap was pulled. Eighty sockeye smolts were PIT tagged at the screw trap and released downstream. Twenty-five of these fish were interrogated at downstream dams with a 56% capture efficiency resulting in an estimated 45 PIT tagged fish that reached the lower Snake River dams. This tag recovery rate results in an estimated 18,017 smolts reaching the the lower Snake River dams from Alturas Lake (56% lake to dam detection survival rate).

The mean fork length of sockeye smolts captured at the trap (Figure 7) was 103 mm (range 85-151 mm), mean weight 9.39 g (range 5.18-29.21 g), and a mean K value (condition factor) 0.821 (range 0.71-1.16). Additionally 393 chinook salmon smolts where caught in the trap during 1998. There was an indirect mortality of five sockeye and no chinook. We caught all of the species listed above for the Pettit Lake Creek weir operation as well as eight steelhead with no fin clips. Water temperatures at the screw trap ranged from 1° to 11.2° C.

All listed fish captured were handled according to the protocol described in the Section 10 Permit request and no mortalities were attributed to handling or PIT tagging. Hence, the negative effects due to the handling of ESA listed fish were minimal. Sockeye captured at the trap during the morning were held until evening twilight of that same day for release. All the fish released were in good condition. At random dates some fish were held for observation. Although some sockeye were slightly descaled they all appeared to be in good condition. All chinook salmon captured were released immediately and were in good condition with no evidence of descaling.

Growth Rates

At the time of tagging, pre-smolts stocked into Pettit Lake were significantly smaller than pre-smolts stocked into Redfish and Alturas lakes (t-test Log_{10} transformed length measurements) ($P<0.0001$, alpha=0.05) (Figure 8). At the time of smolt emigration the Pettit Lake fish were significantly larger (t-test Log_{10} transformed length measurements) ($p<0.0001$, alpha = 0.05) than fish emigrating from Redfish and Alturas lakes (Figure 8). A comparison of the Redfish Lake group of fish at the time of tagging were significantly larger (t-test Log_{10} transformed length measurements) ($p<0.0001$, alpha = 0.05) than the Alturas Lake group of fish. The same Redfish Lake group were significantly smaller fish at the time of smolt emigration (t-test Log_{10} transformed length measurements) ($p<0.0001$, alpha = 0.05) than the Alturas Lake group of fish (Figure 8).

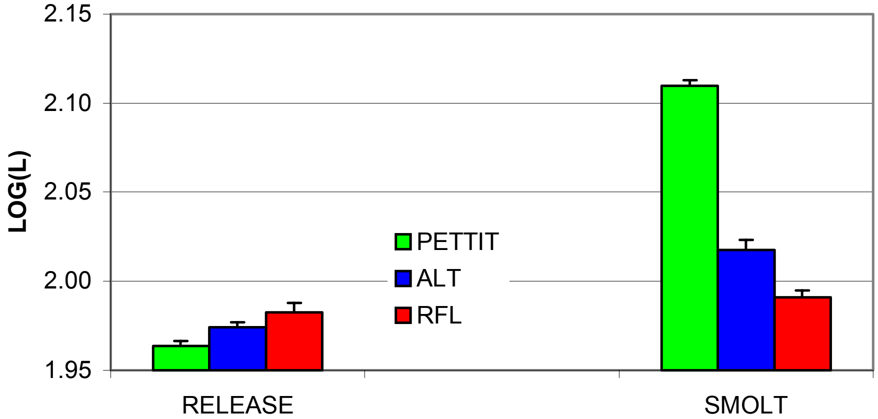

Figure 8. Log transformed length comparisons between lakes at time of tagging and smolt emigration, error bars are one standard error

Linear regression of the Log_{10} transformed length data at the time of tagging and smolt emigration for each group of fish by lake produced a slope of 0.0004 for Pettit Lake, 0.0001 for Alturas Lake, and 0.00005 for Redfish Lake (Fig. 9).

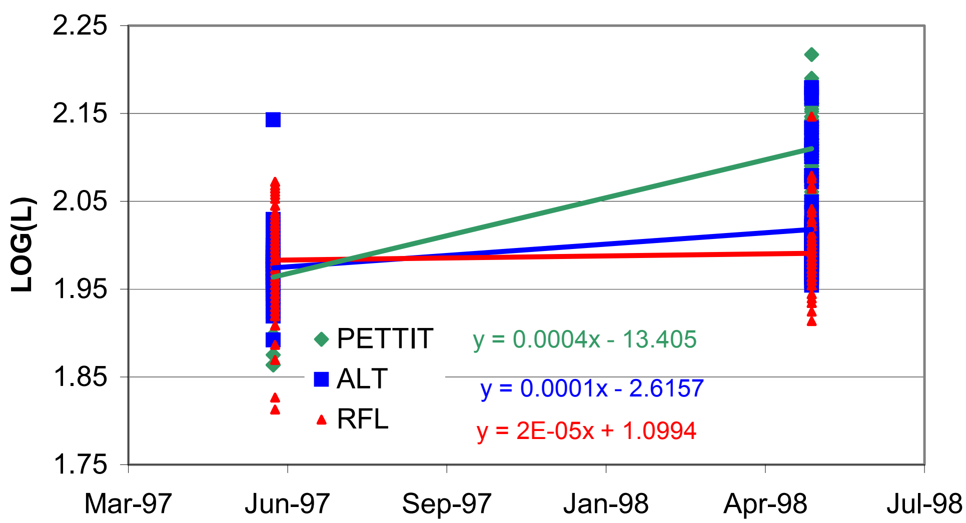

Figure 9. A linear regression of Log_{10} transformed lengths for each group of fish by lake.

The millimeter growth per cumulative temperature unit (mm/CTU) calculations generated a growth rate of 0.0143 mm/CTU for Pettit Lake, 0.0051 mm/CTU for Alturas Lake, and .0029 mm/CTU for Redfish Lake.

Gillnet

Vertical gillnetting on 27 September 1998 in Alturas Lake yielded a total of four *O. nerka* (three had a mean fork length of 97 mm and one with a fork length of 211 mm) and one pit tagged stocked sockeye salmon 100 mm long and 9.05 g. The catch per unit effort (CPUE) for kokanee was 0.24 fish/hour and 0.06 fish/hour for sockeye. Vertical gillnetting on 16 June 1998 in Pettit Lake yielded no fish. On 3 August 1998 vertical gillnetting yielded two northern pikeminnows (mean fork length 137 mm, mean weight 38 g) resulting in a CPUE of 0.15. Horizontal gillnet sampling on 16 June 1998 in Pettit Lake yielded five rainbow trout (CPUE 0.164, mean fork length of 283 mm), four bull trout (CPUE 0.131, mean fork length 285 mm), and eight brook trout (CPUE 0.294, mean fork length 268 mm). Horizontal gillnet sampling on 29 September 1998 in Pettit Lake yielded 28 rainbow trout (CPUE 0.92, mean fork length 224.96, mean weight 139.22), two bull trout (CPUE 0.07, mean fork length 429, mean weight 1200), two brook trout (CPUE, mean fork length 305), and 28 northern pikeminnow (CPUE 0.92, mean fork length 200.39mm, mean weight 120 g) (Table 6).

Table 6. Results of Pettit Lake horizontal gillnet samples 1998

Date	(n)CPUE	Mean Length	Mean Weight	Gillnet Hours
Rainbow Trout				
June 16, 1998	(5) 0.16	283 mm	na	31
Sept. 29, 1998	(28) 0.92	224.9 mm	139 g	22
Bull Trout				
June 16, 1998	(4) 0.13	285 mm	na	31
Sept. 29, 1998	(2) 0.07	429 mm	1200 g	22
Brook Trout				
June 16, 1998	(8) 0.29	268 mm	na	31
Sept. 29, 1998	(2) 0.09	305mm	na	22
Northern Pikeminnow				
June 16, 1998	(0) 0	na	na	31
Sept. 29, 1998	(28) 0.92	200.4 mm	120 g	22

Stream Spawning

Using a modified area under the curve (AUC) method, kokanee escapement for 1998 was estimated for Fishhook Creek (6,149), Alturas Lake Creek (15,237), and Stanley Lake Creek (783) (Figure 10, Table 7).

Alturas Lake Creek

Since 1992, spawning population levels have fluctuated in Alturas Lake Creek ranging from 200 in 1993 to a high in 1998 of 15,237. The 1998 Alturas Lake Creek spawning kokanee population is the highest of any stream surveyed throughout the history of this project. It occurred four years after a large escapement of spawners (3,200) in 1994 (Figure 10). Escapement estimates for Alturas Lake Creek were calculated from field counts made approximately every 4 days from 3 August through 12 September 1998. Assuming equal sex ratios

Table 7. Fry recruitment, egg-to-fry survival and adult escapement in Fishhook, Alturas, and Stanley Lake Creeks.

Location	Brood Year	Adult Escapement	Mean # Eggs	male:female Ratio	Egg-Fry Survival	Fry Recruits
Fishhook	1998	6,149	233	4.6:1	12.3%	38,059
Fishhook	1997	8,572	233	1.4:1	12.3%	70,186
Fishhook	1996	10,662	286	3:1	13.1%	133,036
Fishhook	1995	7,000	230	1:1	12.3%	99,015
Fishhook	1994	9,200	330	1:1	13.6%	143,888
Fishhook	1993	10,800	300	1:1	11.0%	142,000
Fishhook	1992	9,600	300	1:1	12.0%	166,000
Fishhook	1991	7,200	300	1:1	3.0%	36,000
Alturas	1998	15,273	220	1:1	13.0%	218,390
Alturas	1997	8,492	168	1:1	13.0%	92,733
Alturas	1996	744	150	1:1	13.0%	51,677
Alturas	1995	1,600	150	1:1	13.0%	15,600
Alturas	1994	3,200	150	1:1	13.0%	30,000
Alturas	1993	200	-	1:1	13.0%	2,000
Stanley	1998	783	270	1:1	7.0%	7,399
Stanley	1997	629	270	1:1	7.0%	5,935
Stanley	1996	825	270	1:1	7.0%	3,431
Stanley	1995	90	270	1:1	7.0%	850
Stanley	1994	600	270	1:1	7.0%	5,000
Stanley	1993	1,900	-	1:1	7.0%	19,000

there was an estimated 7,636 female fish escapement in 1998. At 220 eggs per female there were 1,142,850 eggs deposited in Alturas Lake Creek (Table 7). Using a 13% egg-to-fry survival rate (Teuscher and Taki 1995) there were an estimated 218,390 fry produced for 1999.

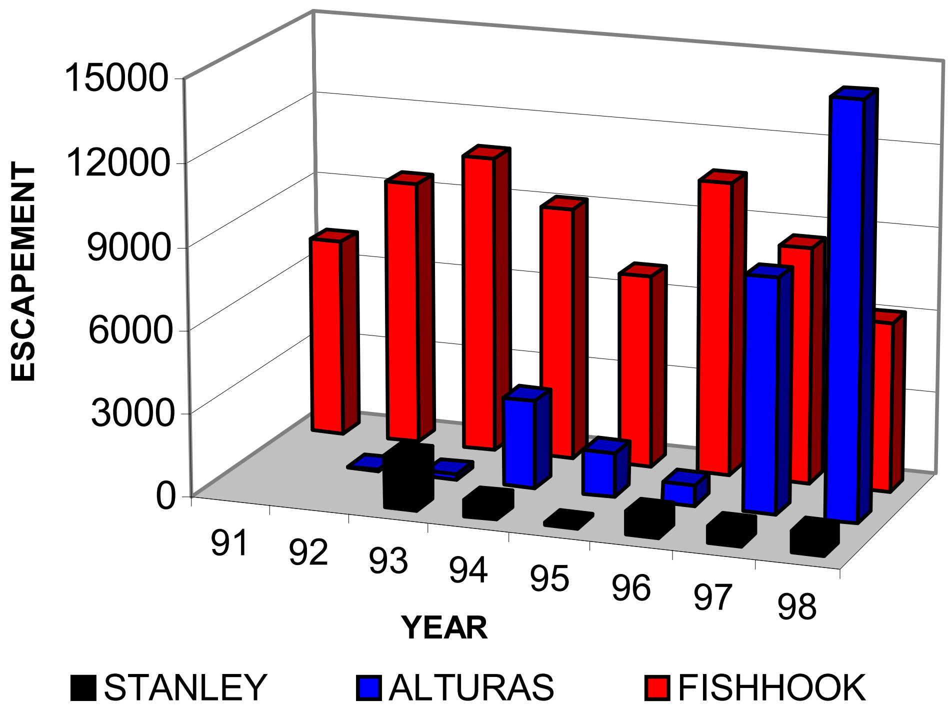

Figure 10. Kokanee escapement estimates for Stanley, Alturas, and Fishhook creeks.

Stanley Lake Creek

The Stanley Lake Creek escapement remained relatively constant (783) despite the low (90) escapement numbers three years ago (Figure 10). Escapement estimates for Stanley Lake Creek was calculated from field counts made every 3 days from 12 August to 9 September. Assuming equal sex ratios the escapement estimate for kokanee females in 1998 was 392 fish. At 270 eggs per female (Teuscher and Taki 1995) there were 105,840 eggs deposited in Stanley Lake Creek. There would be an estimated 7,399 fry produced for 1999 at a 7% egg-to-fry survival rate (Teuscher and Taki 1995).

Fishhook Creek

The Fishhook Creek kokanee escapement estimate for 1998 (6,149 fish) is the lowest recorded in that stream since monitoring started in 1991. However, they remain relatively close to prior population estimates (Figure 10). The sex ratio of kokanee spawners collected at Fishhook Creek weir was 4.63 males to 1 female (Teuscher and Taki 1995). Fishhook Creek escapement estimates should be viewed with caution as they were calculated using only 4 actual counts from 5 August to 11 September 1998. Escapement was intended to be controlled using a weir, which never worked as well as anticipated. More fish were seen in the creek past the weir than counts at the weir accounted for. Using the above sex ratio and estimated escapement there were 1,328 females spawning in the creek with 309,424 eggs (233 eggs per female) (Teuscher and Taki 1995) deposited. A 12.3% egg-to-fry survival rate (Teuscher and Taki 1995) should produce an estimated 38,059 fry in the spring of 1999.

Beach Spawning

All sockeye observed during these surveys were from hatchery adult releases. Snorkel surveys of the southeast shore Redfish Lake recorded one sockeye and two bull trout on 6 October 1998, one sockeye and three bull trout on 13 October 1998, one sockeye, three kokanee, and one bull trout on 27 October 1998, and two sockeye, two residuals, and nine bull trout on 3 November 1998. Snorkel surveys of Sockeye Beach detected three sockeye on 6 October 1998, one sockeye and three bull trout on 13 October 1998, three residuals and twelve adipose clipped net pen reared sockeye in a school on 20 October 1998, three kokanee, and one sockeye on 27 October 1998, and one sockeye on 3 November 1998. There were also many mountain white fish present at both sites and a few westslope cutthroat trout on 6 October 1998. On 13 October 1998, 10 suckers, 24 white fish, and two sculpin were recorded. Trends for residual spawner densities are presented in Figure 11.

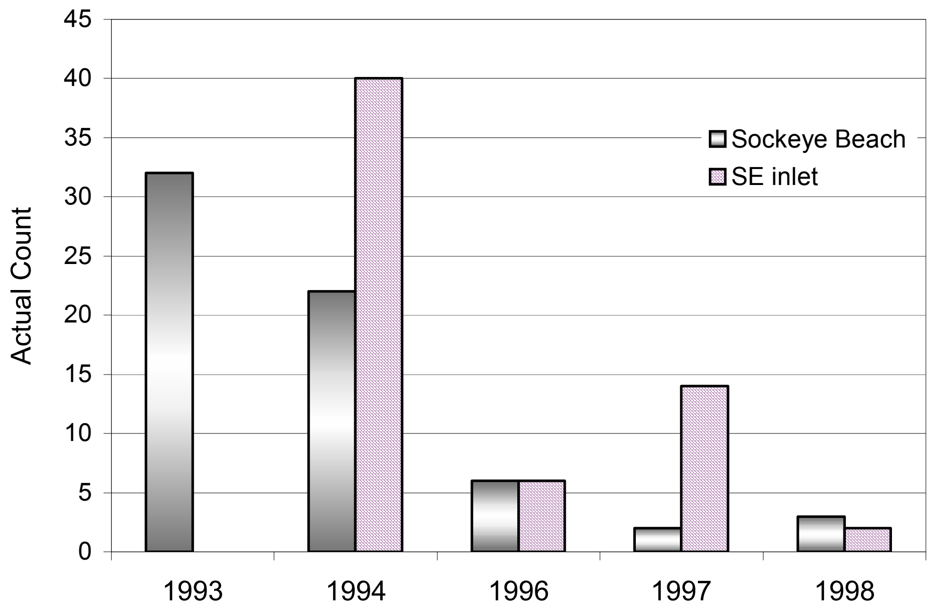

Figure 11. Redfish Lake *O. Nerka* residual counts from snorkel surveys, 1995 missing.

Diet Analysis

The stomachs of 33 rainbow trout (RBT) caught during Pettit Lake gill net efforts (5 in June and 28 in September) were analyzed for diet comparison. No *O. nerka* were found in the stomachs of any of the 33 rainbow trout. Piscivory was evident however with approximately 3% by dry weight of the average RBT diet represented by cyprinids. In June 1998 the diet of RBT (n=5) were dominated by 48% odonates and 30% mollusca followed by 11% chironomid pupae by dry weight (Figure 12). The majority of RBT diet (n=28) caught in September 1998 consisted of 30% chironomid pupae, 16 % terrestrial insects, and 24% plant material by dry weight (Figure 13).

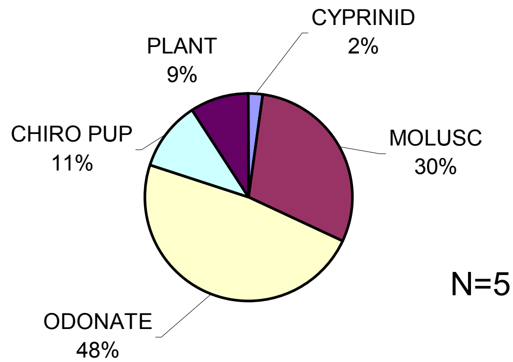

Figure 12. Mean June 1998 rainbow trout diet by percent dry weight.

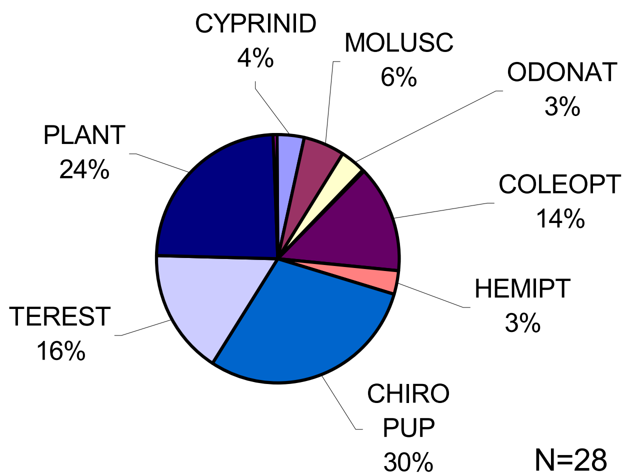

Figure 13. Mean September 1998 rainbow trout diet by percent dry weight.

Stomach content analysis was conducted on 16 Pettit Lake kokanee that were collected by IDFG trawling in September 1998. Kokanee had a mean fork length of 147 mm (range 104-171 mm) and an average weight of 29.9 g (range 11.8–40.7 g). The diet of Pettit Lake kokanee was dominated by 31% chironomid pupae, 27% *Bosmina*, and 24% *Daphnia* by dry weight (approximately 56% zooplankton) (Figure 14, Table 8). The diet of 23 kokanee from the Redfish Lake IDFG trawl sample was composed of 21% chironomid pupae, 31%

daphnia, and 42% cyclopoid by dry weight (approximately 79% zooplankton) (Figure 15, Table 8). There was an average diet overlap of 30% for kokanee and rainbow trout in Pettit Lake. This overlap is almost entirely attributed to chironomid pupae. Past diet surveys (Teuscher and Taki 1995) found that chironomid pupae dominated kokanee diets in early summer and shifted to zooplankton in late summer. In 1998 there was no early summer kokanee diet data available to compare and contrast changes in prey selection.

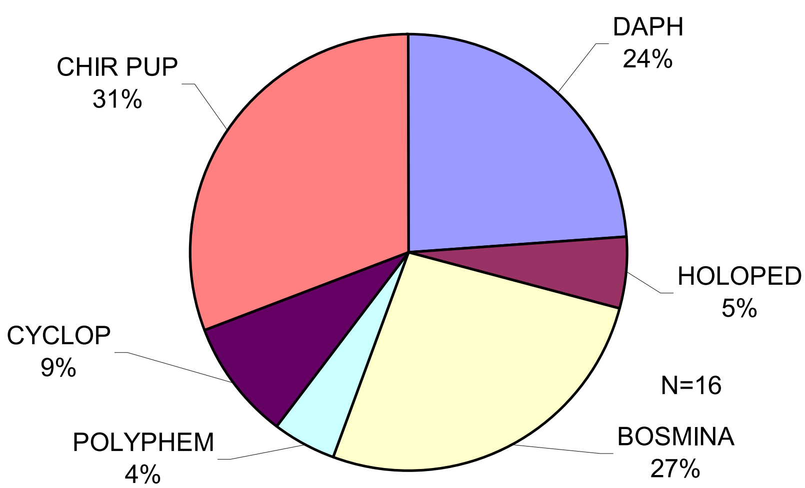

Figure 14. Diet by percent dry weight of Pettit Lake kokanee from IDFG trawl sample.

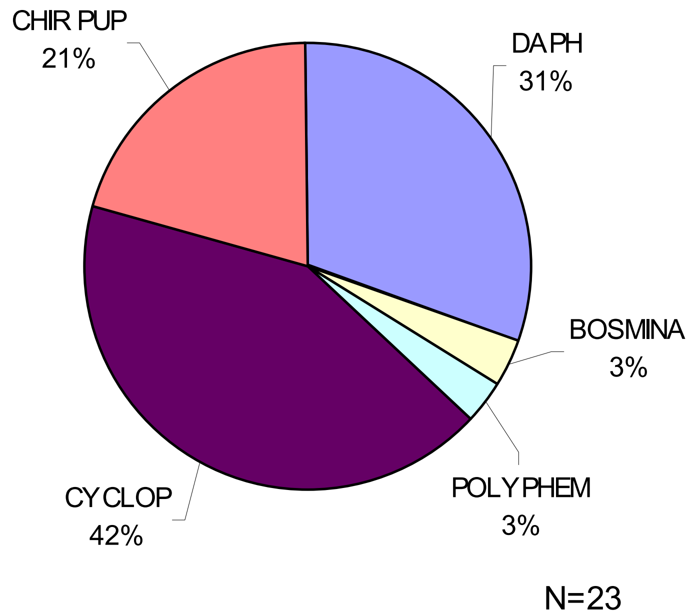

Figure 15. Diet by percent dry weight of Redfish Lake kokanee from IDFG trawl sample.

Table 8. Mean percent dry weight of kokanee diets in Redfish (N=24) and Pettit (N=16) lakes.

	Daph	**Holoped**	**Bosmina**	**Polyphem**	**Cyclop**	**Chiro Pup**
Redfish	30.6	0.0	3.2	3.1	42.3	20.8
Pettit	24.0	5.2	26.5	4.5	8.9	30.9

Additional diet analysis was conducted on fish from Pettit and Alturas lakes that were identified as potential *O. nerka* predators. An analysis of bull trout diet (n=4 June, n=2 September) found no *O. Nerka*. However their September diet was composed of 100% unidentified salmonids (Figure 16). Due to the advanced state of digestion these salmonid prey were unidentifiable and may have been *O. nerka*. The average diet of northern pikeminnow captured in September (n=7) was composed of 77% cyprinids and 23% Odonates (Figure 17). The diet of brook trout captured in June (n=9) was dominated

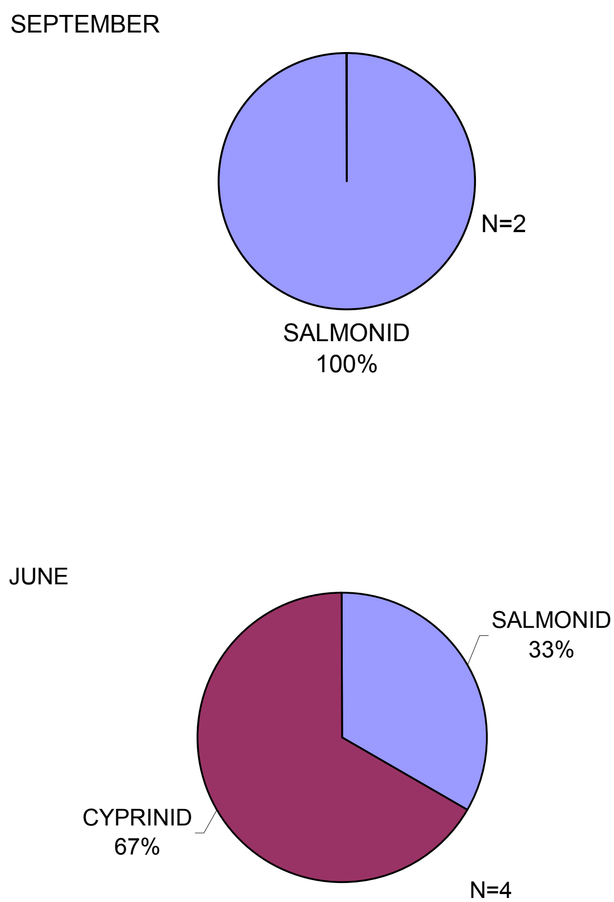

Figure 16. Bull trout diet for June and September by percent dry weight.

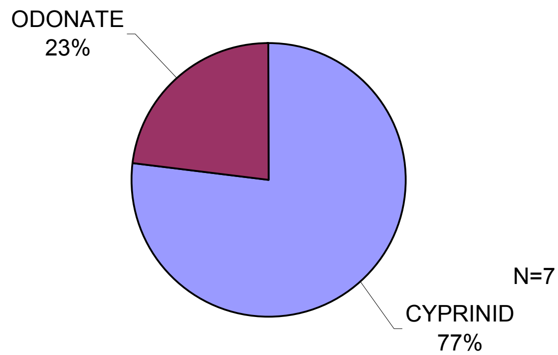

Figure 17. September diet of northern pikeminnow by percent dry weight.

by 79% Odonates followed by 12% cyprinids and 8% insects (Figure 18). Brook trout captured in September had a diet composed of 100% plant material demonstrating a possible diet shift although the sample numbers were too low (n=2) to establish any kind of trend.

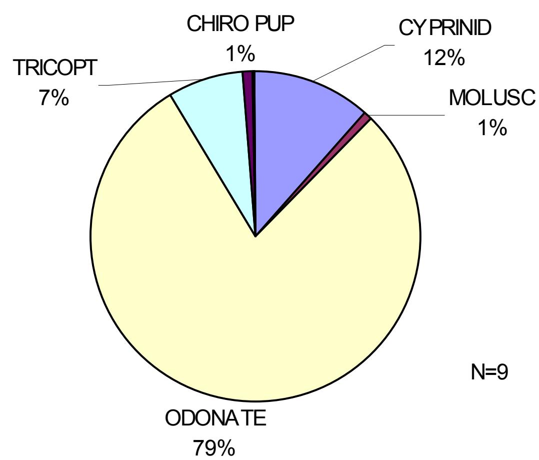

Figure 18. June 1998 brook trout diet by percent dry weight.

Electivity indices calculated for kokanee from Redfish and Pettit lakes are presented in Table 9. Bosmina had the only positive index for Pettit Lake kokanee while Redfish Lake kokanee had positive indices for both Holopedium and Bosmina.

Table 9. Electivity indices for kokanee sampled on September 1, 1998 from Redfish and Pettit lakes.

	Daphnia	Holopedium	Bosmina	Cyclops	Polyphemus
Redfish	-0.77	1.00	0.93	-0.68	-0.81
Pettit	-0.71	-0.59	0.53	-0.06	-0.87

Rainbow Trout Over Winter Survival

Three of the five rainbow trout caught in June had fin clips on their left ventral fins that represent over-winter survival from 1997 stocked fish (left ventral fin clip) into 1998. The other two fish did not have fin clips. Fin clip data was not collected on the 28 RBT collected in Sept. 1998. Thus due to the small sample size (five fish) over-winter survival cannot be calculated.

DISCUSSION

Smolt monitoring

The emigrating fish trapped on Alturas Lake Creek had a mean length of 105 mm and a mean weight of 10.02 g. It is assumed, based on size of emigrating fish and stocking percentages, that these fish were reared at the Sawtooth Hatchery. These fish had an in-river survival rate of 56% from the trap to Lower Snake River dams. Pettit Lake was stocked with Eagle Hatchery reared fish and had a mean length of 129 mm and a mean weight of 18.71 g at the time of emigration. These fish had a 47% lake to dam interrogation survival rate. Pettit Lake smolts captured at the weir had a higher K value of 0.87 compared to 0.82 for smolts from Alturas Lake screw trap. There was a 9% increase in survival rate from lake to dam interrogation for fish from Alturas Lake (derived from Sawtooth Hatchery stock) compared to the fish emigrating from Pettit Lake (derived from Eagle Hatchery stock). This provides initial supporting evidence to the hypothesis that fish raised at the Sawtooth Hatchery may have higher fitness than those raised at the Eagle Hatchery despite the apparent size and K value advantages of the Eagle Hatchery fish.

We encountered no problems during the 1998-trapping season. Downstream dam PIT tag interrogations indicate that these fish migrated faster than previous years (approximately seven days) and that no adverse affect were detected from the time spent in the live boxes.

There is a trend for smolts from Pettit Lake to emigrate later than the other sockeye rearing lakes in the Sawtooth Valley. This pushes their emigration timing further into the period of peak spring flows. The weir on the Pettit Lake outlet was designed for substantially lower peak flows than we encountered in 1998. Since the weir could not be operated during peak run-off a significant portion of the smolt emigration may be unaccounted for. Trapping operations were suspended in 1998 due to high flows while smolts were still being captured. Therefore the 11% over-winter survival/emigration rate is conservative.

Growth rates

Growth rates of stocked *O. nerka* from the captive rearing program may provide further insight into potential differences into hatchery origin and between lake rearing conditions. However, these results should be viewed with caution due to the historic organization of the stocking strategy discussed in the stocking strategy section above. Despite the confounding variables, fish grew significantly faster in Pettit Lake compared to fish released into Redfish and Alturas lakes during 1997-1998. Pettit Lake has an intermediate total zooplankton biomass between Redfish (lower) and Alturas (higher). Yet stocked *O. nerka* are growing significantly faster in Pettit Lake compared to both lakes. In 1997 Pettit Lake received one stocking of fish on 1 July. Redfish and Alturas lakes received two fish stockings with the majority of the fish stocked in October. The differences in growth may be associated with release timing or hatchery origin but that cannot be determined from the current data set. Further possible explanations include factors such as foraging efficiency, zooplankton community dynamics, and fish community dynamics, which could be affecting growth rates of these fish. There are between lake differences in species composition of the zooplankton community. Alturas Lake kokanee population was three times higher in 1998 than in 1997 which has implications for the growth and survival of stocked juvenile *O. nerka*.

Kokanee spawning

Alturas Lake Creek (the inlet to the lake) had the largest kokanee escapement (estimated >15,000) of any creek or spawning event since monitoring began. This high number has implications on the recovery program in several areas. In 1997 Alturas Lake was fertilized to increase carrying capacity. The increased number of kokanee may be evidence of a non-target fish population benefiting from recovery activities (fertilization and associated increased carrying capacity). The enhancement of a non-target fish population was one of the original concerns associated with the fertilization program. The kokanee population in Alturas Lake has fluctuated as much as 1,200% from 1990 to 1997 (based on trawl) when no fertilization activities were performed.

The large number of kokanee fry produced by this spawning event (an estimated 218,390) represent intra-specific competition for the stocked anadromous form of *O. nerka*, the target fish population of the recovery effort. Diet data collected on this project (Taki et al. 1999) found that juvenile kokanee and juvenile sockeye diets were dominated by zooplankton and had a high degree of overlap.

The large cohort associated with this spawning event represents a significant increase in grazing pressure on the zooplankton community. The increased grazing pressure on the zooplankton community may have a negative affect on zooplankton density and species composition. It has been shown that increased zooplankton grazing can decrease densities, alter size distribution, and alter species composition (Soranno et al. 1993, Leavitt et al. 1993). Furthermore, there is a demonstrated shift in species composition from larger sized, high food value species, such as *Daphnia*, to species that are smaller and of lesser food value such as *Bosmina* (Leavitt et al 1993). This potential change in zooplankton due to the effects of this cohort may overwhelm effects on zooplankton community expected from fertilization.

There have been natural fluctuations in both the kokanee population and zooplankton community, which cannot be attributed entirely to the fertilization program. Alturas Lake experienced a crash in the zooplankton density and biomass prior to the implementation of fertilization activities. Similarly, kokanee escapement has shown fluctuations from 200 in 1993 to the current high of 15,237 that can not be attributed solely to the two years of fertilization. Pettit Lake has also experienced large natural fluctuations in kokanee abundance and zooplankton biomass and species composition providing further evidence that alterations may not be a result of fertilization.

There are several steps that can be taken to address the implications of an increased kokanee biomass on stocked sockeye. One strategy is to do nothing, allowing the kokanee population to fluctuate naturally with the trophic dynamics of the lake. Another strategy would be similar to that initiated in Fishhook Creek, using a weir to control kokanee escapement. Attempts to use a weir in Fishhook Creek have thus far has been unsuccessful due to problems with the weir. The application of this strategy to Alturas Lake Creek which is larger than Fishhook Creek may be problematic and costly. Another option would be to disturb kokanee redds using a rake or other implement before fry emergence. Culling fry caught in fry traps after emergence could also control the kokanee population. The discontinuation of fertilization or a different fertilization application schedule should also be considered as a possible strategy. These options will have to be discussed and evaluated by the TOC before any activities are implemented.

Fishhook Creek, the primary kokanee spawning habitat for Redfish Lake, continues to be monitored for kokanee escapement. A target of 2,000 female kokanee was set by the TOC with up to 4,000 males. Control efforts to date have met with limited success yet that target was met in 1998 (6,149 fish past the weir). Further attempts to mitigate the potential competition with the stocked anadromous form of *O. nerka* may be warranted depending on future spawner densities. Stanley Lake kokanee spawn numbers remain low and stable and are not of concern but should have continued monitoring to track population trends through time.

Diet

Northern pikeminnow are known to predate on juvenile salmon and are the subject of control efforts in the main stem of the Columbia River. Northern pikeminnow are one of the more abundant species found in the sockeye rearing/nursery lakes of the Sawtooth Valley. There has been concern expressed about their potential predation on stocked juvenile sockeye. Diet analysis has found that while piscivorous (77% of diet in 1998 sample composed of cyprinids, Figure 17) there has been no evidence of predation on *O. nerka* by northern pikeminnow. During gillnet sampling, the majority of northern pikeminnow are caught in the littoral zone of the lakes. *O. nerka* are primarily a pelagic species. The low degree of habitat utilization overlap may limit the opportunity for northern pikeminnow to predate on *O. nerka*. Predation by northern pikeminnow is not currently considered a problem. Ongoing monitoring of the northern pikeminnow populations and diets is warranted in order to detect any potential changes.

Stocked juvenile sockeye from the captive rearing program were found in the stomachs of stocked rainbow trout (*O. mykiss*) in Pettit Lake during 1995, the first year that sockeye were stocked into Pettit Lake (Teuscher and Taki 1996). The sockeye were released at the boat ramp in the littoral zone. After detection of *O. nerka* in *O. mykiss* stomachs, the stocking strategy was modified to a pelagic release using a barge. Since the pelagic release was implemented there has been no predation of *O. nerka* detected by *O. mykiss* in Pettit Lake.

Bull trout are the top piscivorous predator of the fish community in the Sawtooth Lakes. The diets of bull trout in this program have been shown to be primarily composed of fish prey

(Taki et al. 1999). However, no *O. nerka* have been detected in any of the samples. Salmonids, too far digested to be identified, were found in some of the samples and some may have been *O. nerka*. Bull trout have been listed as a threatened species in 1998 under the Endangered Species Act and, as the top predator, are an important component of fish community dynamics in the Sawtooth Lakes and upper Salmon River. Any predation by this species on *O. nerka* is considered a natural process and no control measures will be implemented. Continued incidental takes during gill net sampling are anticipated. This will allow for monitoring of bull trout population dynamics. Otoliths and scale samples are collected from all bull trout and will be used to develop population age structure and age at length relationships.

Beach spawning

The main reason for conducting the snorkel surveys on Sockeye Beach and the small section of spawning habitat at the south end of Redfish Lake is to monitor residual sockeye spawning activity. Residuals are genetically similar to the listed anadromous form of Redfish Lake *O. nerka* and are listed as an Evolutionarily Significant Unit. They are behaviorally different in that they are non-anadromous and spend their entire life cycle in the lake. Otherwise they have a similar life history to the anadromous form of sockeye living for four years, spawning in the same areas, and then dying. In 1998 there were a total of five residuals sighted during the four night snorkel surveys (three on Sockeye Beach and two on the south end). There has been a steady decline in the numbers of residuals detected since monitoring began in 1993 (Figure 11). There is no known cause associated with this decline. However the decline may indicate a linkage with anadromous fish. There have been very limited numbers of adult anadromous fish returning for the past ten years. Adult returns are known to provide a significant amount of potential production in many aquatic systems through the import of marine derived nutrients. There has been a concurrent decline in residual sockeye, which may similarly be dependent on the allochthonous input supplied by returning adult anadromous fish.

REFERENCES

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

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

Ehrenberg, and Torkelson. 1996. Application of dual-beam and split-beam target tracking in fisheries acoustics. ICES Journal of Marine Science. 53:329-334.

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

Gunderson, D. R. 1993. Surveys of fisheries resources. John Wiley and Sons, Inc. New York, New York. 248 pp.

Ivlev, V. S. 1961. Experimental ecology of the feeding of fishes. Yale University Press, New Haven, CT.

Koenings, J.P., J.A. Edmundson, G.B. Kyle, and J.M. Edmundson. 1987 Limnology field and laboratory manual: methods for assessing aquatic production. Alaska Department of Fish and Game, Division of Fisheries Rehabilitation, Enhancement and Development. Juneau, Alaska.

Leavitt, P. R., P. R. Sanford, S. R. Carpenter, J. F. Kitchell, and D. Benkowski. 1993. Annual fossil records of foodweb manipulation. In Eds. Carpenter and Kitchel, The trophic cascade in lakes. Cambridge University Press, Cambridge UK.

McCauley, E. 1984. The estimation of the abundance and biomass of zooplankton in samples. In Eds. J. A. Downing and F. Ringler. Secondary production in freshwater, Second edition. Blackwell Scientific Publishing, Oxford, UK.

Soranno, A. S., S. R. Carpenter, and X. He. 1993. Zooplankton biomass and body size. In Eds. Carpenter and Kitchel, The trophic cascade in lakes. Cambridge University Press, Cambridge UK.

Swanson, G. A., G. L. Krapu, J. C. Bartonek, J. R. Serie, and D. H. Johnson. 1974. Advantages of mathematically weighted waterfowl food habitat data. Journal of wildlife management. 38:302-307.

Taki, D. B. Lewis, and R. Griswold. 1999. Snake River sockeye salmon habitat and limnological research. U.S. Department of Energy, Bonneville Power Administration, Portland, OR. Project number 91-71. Pp.1-50.

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

Traynor and Ehrenberg. 1990. Fish and standard-sphere target-strength measurements obtained with a dual-beam and split-beam echosounding system. Rapp. P.-v. Reun Cons. Int. Explor. Mer. 189:325-335.

CHAPTER 2: LIMNOLOGY OF THE SAWTOOTH VALLEY LAKES, 1998

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INTRODUCTION

In December 1991, the National Marine Fisheries Service listed Snake River sockeye salmon O*ncorhynchus nerka* as endangered under the Endangered Species Act (Waples et al. 1991). As a result, the Sawtooth Valley Project was initiated to conserve and rebuild sockeye salmon populations in the Sawtooth Valley lakes (Redfish, Pettit, Alturas and Stanley lakes). The recovery strategy involved increasing the number of juveniles rearing in the nursery lakes using hatchery broodstocks (Flagg and McAuley 1994, Johnson and Pravecek 1995), improving growth and survival of juvenile sockeye salmon and increasing lake carrying capacities via lake fertilization (Stockner and MacIsaak 1996). In Redfish Lake, kokanee O*ncorhynchus nerka* escapement was controlled to limit kokanee recruitment to the lake and reduce intra-specific competition between kokanee and sockeye salmon (Taki and Mikkelsen 1997).

Lake fertilization has been successfully used to stimulated primary production in oligotrophic lakes and, through trophic transfer, increase macrozooplankton production thus improving rearing habitat for young planktivorous sockeye salmon (LeBrasseur et al. 1978, Hyatt and Stockner 1985, Stockner and Shortreed 1985, Stockner 1987, Kyle 1994). Lake fertilization programs replace nutrients formerly derived from decaying salmon carcasses with liquid fertilizer in nutrient limited systems with depressed adult escapement (LeBrasseur et al. 1979, Stockner and MacIsaak 1996). In Alaska and British Columbia, lake fertilization has been associated with increased survival and growth of young sockeye salmon (LeBrasseur et al. 1978, Robinson and Baraclough 1978, Hyatt and Stockner 1985, Kyle 1994) and elevated adult escapement (LeBrasseur et al. 1979, Stockner and MacIsaac 1996). The relationship between *O. nerka* population abundance and available forage in a nursery lake (carrying capacity) can be manipulated with nutrient applications resulting in higher lake carrying capacities (Stockner and MacIsaak 1996). While success of fertilization programs are often determined by increases in zooplankton abundance and biomass (Kyle 1994), a successful program could also result in stable zooplankton populations under increased grazing pressure from expanding *O. nerka* populations.

In 1998, limnological monitoring was conducted to assess productivity and identify changes in physical and chemical characteristics of the Sawtooth Valley lakes. The information was used to identify inter-annual variation of physical and chemical characteristics, evaluate fertilizer treatments, and determine *O. nerka* carrying capacities of the Sawtooth Valley lakes. Methodologies and sampling designs were developed by Utah State University (USU) during the initial phase of this project (Budy et al. 1993, Steinhart et al. 1994, Budy et al. 1995, Luecke et al. 1996) and modified by Griswold (1997). The variables measured in 1998 include, water temperature, dissolved oxygen, conductivity, water transparency, light penetration, nutrient concentrations, phytoplankton species composition, abundance and bio-volume, chlorophyll *a* concentrations, primary productivity, and zooplankton density and biomass.

STUDY AREA

The Sawtooth Valley lakes are located in South Central Idaho near the town of Stanley. The watersheds are located in the Sawtooth Mountains, mostly within the Sawtooth Wilderness Area and administered by the US Forest Service, Sawtooth National Recreation Area (SNRA). The Sawtooth Mountains are part of the Idaho batholith, comprised of granite-like rock, consisting of granodiorite, quartz diorite and quartz monzonite (Emmett 1975). The lakes are at a relatively high elevation (1985-2157 m), generally ice covered from January to May and classified as oligotrophic. The ratio of drainage area to lake surface area (Table 1)(Gross et al. 1993) is 48.6 for Stanley Lake,

Table 1. Sawtooth Valley lakes physical characteristics.

Lake	Area (km^2)	Volume (m^3x10^6)	Elevation (m)	Mean Depth (m)	Maximum Depth (m)	Drainage Area (km^2)	Drainage area/ lake surface area	Water residence time in years (Gross, 1993)
Redfish	6.15	269.9	1996	44	91	108.1	17.6	3.0
Pettit	1.62	45.0	2132	28	52	27.4	16.9	2.2
Alturas	3.38	108.2	2138	32	53	75.7	22.4	1.8
Stanley	0.81	10.4	1985	13	26	39.4	48.6	0.3

22.4 for Alturas, 17.6 for Redfish and 16.9 for Pettit Lake. Morphometric maps of the lakes and descriptions of the lakes and their watersheds are reported in Budy et al. (1993), a map of the study area is in Chapter I of this report. Limnological sampling was conducted at three stations in each lake. The stations were positioned along the longitudinal axes of the lakes, with the main station near mid-lake, the south station near the south or west end (inlet), and the north station nearest the north or east end (outlet).

METHODS

Lakes were sampled from January to November 1998. Redfish, Pettit and Alturas lakes were sampled once per month in January, March, May, October, and November and twice per month from June through September. In 1998, these three lakes were stocked with juvenile sockeye salmon from the Redfish Lake captive broodstock and were enriched with liquid fertilizer. Stanley Lake was not stocked with sockeye salmon and did not receive nutrient applications. Stanley Lake was sampled once in March, June, August, September and October to help distinguish the effects of natural annual variation versus fertilization treatments. Utah State University, contracted by the Shoshone-Bannock Tribe, studied these lakes extensively from 1991 to 1995. Data collected, compiled, and reported by USU have been used throughout this project (Spaulding 1993, Teuscher et al. 1994, Teuscher et al. 1995, Teuscher and Taki 1996). When lakes were stratified, water for nutrient analysis was collected from the epilimnion, metalimnion and hypolimnion. Chlorophyll *a* and phytoplankton samples were collected from the epilimnion, metalimnion, and 1% light level. Three discrete samples were collected from each stratum with a three L Van Dorn bottle and mixed in a churn splitter. When lake strata could not be delineated, surface water was collected from 0-6 m with a 25 mm diameter, 6 m long lexan® tube and discrete samples were collected from mid-depth (Redfish = 45 m, Pettit and Alturas = 25 m, and Stanley = 12 m) and 1-2 m above the bottom.

Lake Fertilization

In 1998, the Shoshone-Bannock Tribes (SBT), operating under a consent order issued by the Idaho Division of Environmental Quality (DEQ), added supplemental nutrients to Redfish, Pettit, and Alturas lakes. The consent order requires measurement of water transparency once per week and estimates of epilimnetic and metalimnetic chlorophyll *a* and nutrient concentrations every two weeks. The consent order specifies that nutrient enhancement activities may continue as long as 1) water transparencies are greater than 8, 6 and 4 m in Redfish, Pettit and Alturas lakes, respectively, prior to 15 July and greater than 8 m in all three lakes after 15 July, 2) epilimnetic chlorophyll *a* is less than 3 *u*g/l, 3) metalimnetic chlorophyll *a* remains less than 6 *u*g/l, and 4) total phosphorus concentrations remain below 15 *u*g/l in the epilimnion and metalimnion of all three lakes.

Liquid ammonium phosphate (20-5-0) and ammonium nitrate (28-0-0-0) fertilizer was applied weekly from 2 June to 30 September 1998. Stockner (1997) developed fertilization prescriptions for each lake. Nutrients were applied at a ratio of approximately 20:1 N:P by mass (45:1 molar) and were purposely skewed toward high nitrogen loads to avoid stimulation of nitrogen fixing Cyanophytes. The quantity of nutrients applied each week was variable, intended to simulate the natural hydrograph. Thus, initial applications were relatively small, rapidly increased to a peak in mid to late July, and then gradually declined until late September. However, the intended pattern was interrupted in Pettit and Alturas lakes during June and July 1998 because DEQ criteria were exceeded. The applications were made from a 6.7 m boat equipped with a portable plastic tank and electric pump. The fertilizer was loaded into the tanks off-site then pumped into the boat's wake while traveling over the surface of the lake. Twenty predetermined transect lines were followed at Redfish Lake, 12 at Alturas Lake and 8 at Pettit Lake, using GPS, compass, and local landmarks to evenly disperse the nutrients over the surface of the lake.

Profile Data

Temperature (^{o}C), dissolved oxygen (mg/l), and conductivity (*u*S/cm) profiles were collected at the main station of each lake using a Hydrolab® Surveyor3™ equipped with a Hydrolab H20® submersible data transmitter or a Yellow Springs Instrument Model 58 dissolved oxygen meter. The instrument was calibrated each day prior to sampling. Dissolved oxygen was calibrated using barometric pressure estimated from elevation.

Standards obtained from the Myron L Company were used to calibrate conductivity. Temperature, dissolved oxygen and conductivity were recorded at 1 m intervals from the surface to 10 m, 1-2 m intervals from 10 m to the thermocline, then at 2-10 m intervals to the bottom. Mean water temperatures from 0-10 m were used to calculate seasonal mean (May-October) surface water temperatures.

Water transparency and light penetration

Water transparency was measured at the main station of each lake with a 20 cm secchi disk. The disk was lowered into the water until it disappeared from sight and the depth was noted. The depth at which the disk reappeared when raised was also noted and an average of the two values was recorded as water transparency depth (Koenings et al. 1987).

Light attenuation was measured at the main station of each lake with a LiCorr® Li-1000 data logger equipped with a Li-190SA quantum sensor deck cell and a LI-193SA spherical sea cell. Photosynthetically active radiation (400-700 nm) was measured at two-meter intervals from surface to 2-4 m below the 1% light level. Deck and sea cell readings were made simultaneously to correct for changes in ambient light. Depth of the 1% light level was determined by linear regression of the natural log of percent surface light at each depth versus depth (Wetzel and Likens 1991).

Water chemistry

Water was collected for nutrient analysis once per month in January, March and May-October. Water was transferred to nalgene bottles that had been rinsed in 0.1 N HCL and sample water. Bottles were stored at 4 ^{o}C while in the field. Water for ammonia, nitrate, and orthophosphorus assays were filtered through 0.45 *u*m acetate filters at 130 mm Hg vacuum in the laboratory. Water samples were frozen and shipped to the UC Davis Limnology Laboratory for analysis. Ammonia was assayed with the indophenol method, nitrate with the hydrazine method, organic nitrogen using kjeldahl nitrogen, the calorimetric method was used to determine orthophosphorus and total phosphorus was assayed by persulfate digestion (Hunter et al. unpublished). Method detection levels for each assay are shown in Table 2.

Table 2. Nutrient assay methods, minimum detection levels, and confidence intervals.

Assay	Method	MDL (*u*g/l)	99% C.I.
Ammonia	Indophenol	3	± 0.3
Nitrate	Hydrazine	2	± 0.3
Organic nitrogen	Kjeldahl	35	± 16.0
Orthophosphorus	Calorimetric	1	± 6.6
Total phosphorous	persulfate digestion	2	± 0.5

Chlorophyll a and phytoplankton

Water was collected for chlorophyll *a* each sample period and for phytoplankton once per month in May and October and twice per month from June through September from the epilimnion, metalimnion, and the 1% light level. Water samples were stored at 4 ^{o}C in the field and then filtered onto 0.45 *u*m cellulose acetate membrane filters with 130 mm Hg vacuum pressure. Filters were placed in centrifuge tubes and frozen (-25 ^{o}C). The filters were then placed in methanol for 12-24 hours to extract the chlorophyll pigments. Florescence was measured with a Turner model 10-AU fluorometer calibrated with chlorophyll standards obtained from Sigma Chemical Company. Samples were run before and after acidification to correct for phaeophytin. (Holm-Hansen and Rieman 1978). Phytoplankton samples were fixed in Lugol's solution and shipped to Eco-Logic, Inc. for identification and analysis. Phytoplankton was identified and counted using an inverted fluorescent microscope and abundance and bio-volume were determined (Stockner 1998).

Primary Productivity

State of Washington Water Research Center personnel estimated primary productivity in the Sawtooth Lakes during the 1998-growing season. Primary productivity estimates were obtained on 27-30 June, 16-19 July, 22-25 August, and 17-20 September 1998 at two stations in Redfish Lake and one station each in Pettit, Alturas and Stanley lakes. Primary productivity was evaluated within the photic zone, which was delineated by the depth of the 1-% light level. Discrete primary productivity estimates were made at eight depths in Redfish, Pettit and Alturas lakes and from six depths in Stanley Lake. Field and laboratory procedures used to make discrete productivity estimates are summarized in Wierenga et al. (1999). Discrete primary productivity (mg C/m^3/hour) estimates were plotted and integrated using plainimetry to determine hourly rates of primary productivity based on surface area (mg C/m^2/hour). Hourly productivity estimates were expanded to daily productivity (mg C/m^2/day) using solar irradiance data and the methodology described by Vollenweider (1965) and Britton and Greeson (1987). Wierenga et al. (1999) provides a complete description of methods used to determine primary productivity in 1998.

Zooplankton

Zooplankton was sampled one to two times per month. Vertical hauls were made with a 0.35 m diameter, 1.58 m long, 80 *u*m mesh conical net, with a removable bucket. The net was equipped with a release mechanism that allowed sampling at discrete depth intervals. A General Oceanics flow meter modified with an anti-reverse bearing was mounted in the mouth of the net. The flow meter was used to correct for net efficiency (clogging). The net was retrieved by hand at a rate of one meter per second from 10-0 m, 30-10 m, 60-30 m and 2 m above bottom to 60 m at the deep station in Redfish Lake. The shallow stations in Redfish and all stations in Pettit and Alturas lakes were sampled from 10-0 m, 30-10 m, and bottom-30 m. Stanley Lake was sampled at 10-0 m and bottom to 10 m. Samples were preserved in 10% buffered sugar formalin. Techniques used to subsample, count, and measure zooplankton were adopted from Utah State University (Steinhart et al. 1994) using techniques and length-weight relationships developed by McCauley (1984) and Koenings et al. (1987).

RESULTS

In 1998, mean annual discharge of the Salmon River at Salmon, Idaho (USGS gage 13302500) was 62.1 m^3/s, slightly above the 1913-1998 average of 55.4 m^3/s (Figure 1). This represents a continuation of the pattern of below average annual discharge prior to 1995 and above average discharge since 1995.

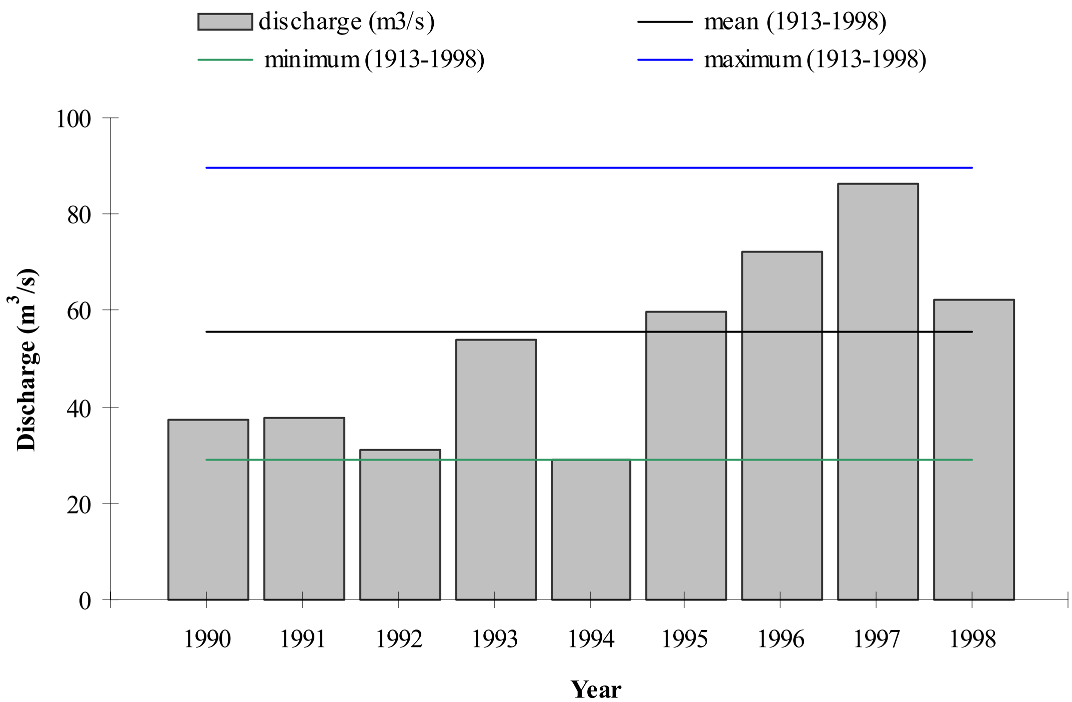

Figure 1. Mean annual discharge for the Salmon River at Salmon, Idaho, 1990 through 1998. Mean, minimum and maximum are for period of record, 1913 to 1998.

Lake Fertilization

Between 2 June and 30 September 1998, Redfish Lake was fertilized with 190 kg phosphorus (P) and 3702 kg nitrogen (N). This was the forth-consecutive year nutrients were added to Redfish Lake. Applications in 1998 were 27% less than those in 1995, nearly four times the 1996 applications and similar to the 1997 additions (Table 3). Nutrient supplementation was initiated in 1997 in Pettit and Alturas lakes and continued in 1998. Pettit Lake was fertilized with 25 kg P and 465 kg N between 2 June and 19 August and Alturas Lake received 58 kg P and 1172 kg N between 4 June and 29 September. Total P applications were 22% and 9% less than the 1997 additions in Pettit and Alturas lakes, respectively. Total nutrient additions for Pettit and Alturas lakes, based

Table 3. Supplemental nutrient additions to Redfish, Pettit, and Alturas lakes, Idaho during 1995 to 1998.

Lake	Year	P (kg)	N (kg)	mg P/m^2	mg N/m^2	TN:TP
Redfish	1995	261	4,623	42	752	18
	1996	51	934	8	152	18
	1997	190	3,695	31	601	19
	1998	190	3,702	31	602	20
Pettit	1995	0	0	0	0	-
	1996	0	0	0	0	-
	1997	32	632	20	390	20
	1998	25	465	15	287	19
Alturas	1995	0	0	0	0	-
	1996	0	0	0	0	-
	1997	64	1,339	19	396	21
	1998	58	1,172	17	347	20

on lake surface area, were approximately 50% of applications to Redfish Lake in 1998. Areal loading rates were 30.9, 15.4, and 17.2 mg P/m^2 for Redfish, Pettit and Alturas lakes, respectively. TN:TP ratios of fertilizer additions were 19.5, 18.6, and 20.2 in Redfish, Pettit and Alturas lakes, respectively.

Lower nutrient supplementation rates in Pettit and Alturas lakes resulted partly from shutdowns caused by declines in water transparencies and increases in chlorophyll *a* concentrations. In Pettit Lake surface chlorophyll *a* was 2.0 *u*g/l and water transparency was 9.3 m on 21 June 1998. The following week water transparency had declined to 7.0 m (Shoshone-Bannock Tribes, unpublished data), which was still within DEQ guidelines, but nutrient additions were voluntarily suspended for two weeks (Figure 2). Nutrient supplementation resumed on 8 July when water transparency increased to 11.4 m. On 24 August, water transparency was 7.6 m, less than the 8.0 m transparency required by DEQ during late summer. Nutrient applications were discontinued in Pettit Lake for the remainder of the year.

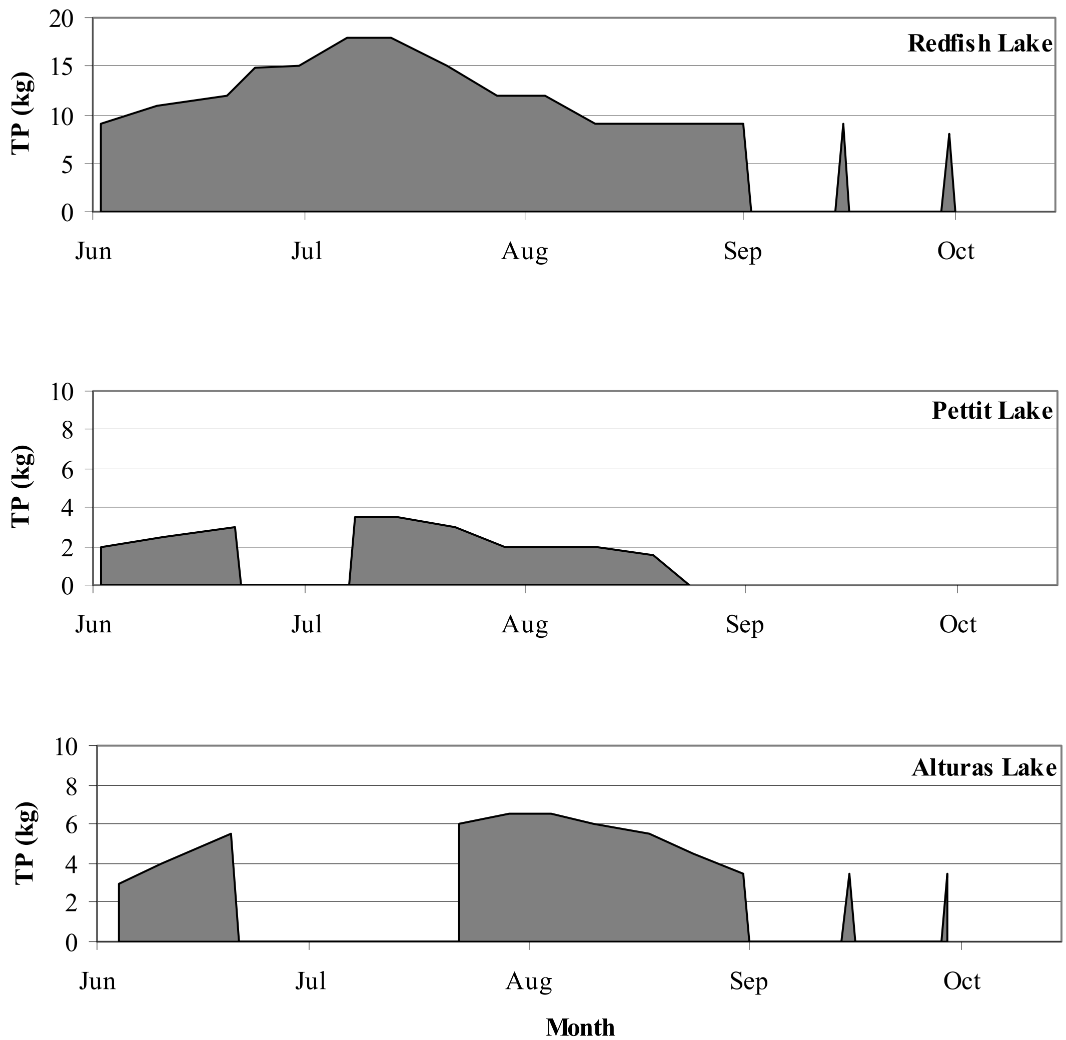

Figure 2. Supplemental total phosphorus additions (kg) to Redfish, Pettit, and Alturas lakes, Idaho during June through October 1998.

Alturas Lake surface chlorophyll *a* concentrations were 6.3 *u*g/l on 9 June and 3.7 *u*g/l on 20 June 1998, which exceeded the early summer DEQ criterion of 3 *u*g/l of chlorophyll *a*. The SBT suspended fertilization efforts until 22 July when surface chlorophyll *a* declined to 0.5 *u*g/l and water transparency increased to 13.0 m. During September nutrient applications were interrupted two times in Alturas and Redfish lakes because of manpower constraints.

Profile Data

The Sawtooth Valley lakes were inversely stratified and ice covered from December 1997 to early May 1998. On 9 May 1998 Alturas Lake remained ice covered while Redfish, Pettit and Stanley lakes were ice-free. Thermoclines were well developed from July through late October (Figures 3a, 3b, 3c, and 3d). The lakes were nearly isothermic by November.

Dissolved oxygen concentrations in the Sawtooth lakes were generally greater than 5 mg/l, the minimum level that will support growth and survival of salmonids (Lagler 1956). However, hypoxic conditions in the deeper areas were more extensive than previously observed in all four lakes. In late May, dissolved oxygen concentrations were less than 5 mg/l in the bottom 4 to 6 m of Redfish, Alturas and Stanley lakes. From early June until early August or September these lakes had dissolved oxygen concentrations greater than 5 mg/l throughout the entire water column. Oxygen concentrations in the deeper areas gradually declined from August or September until November when the bottom 3, 9 and 7 m dropped below 5 mg/l in Redfish, Alturas and Stanley Lakes, respectively. Pettit Lake, which is meromictic, had dissolved oxygen concentrations less than 5 mg/l below 27-33 m depth.

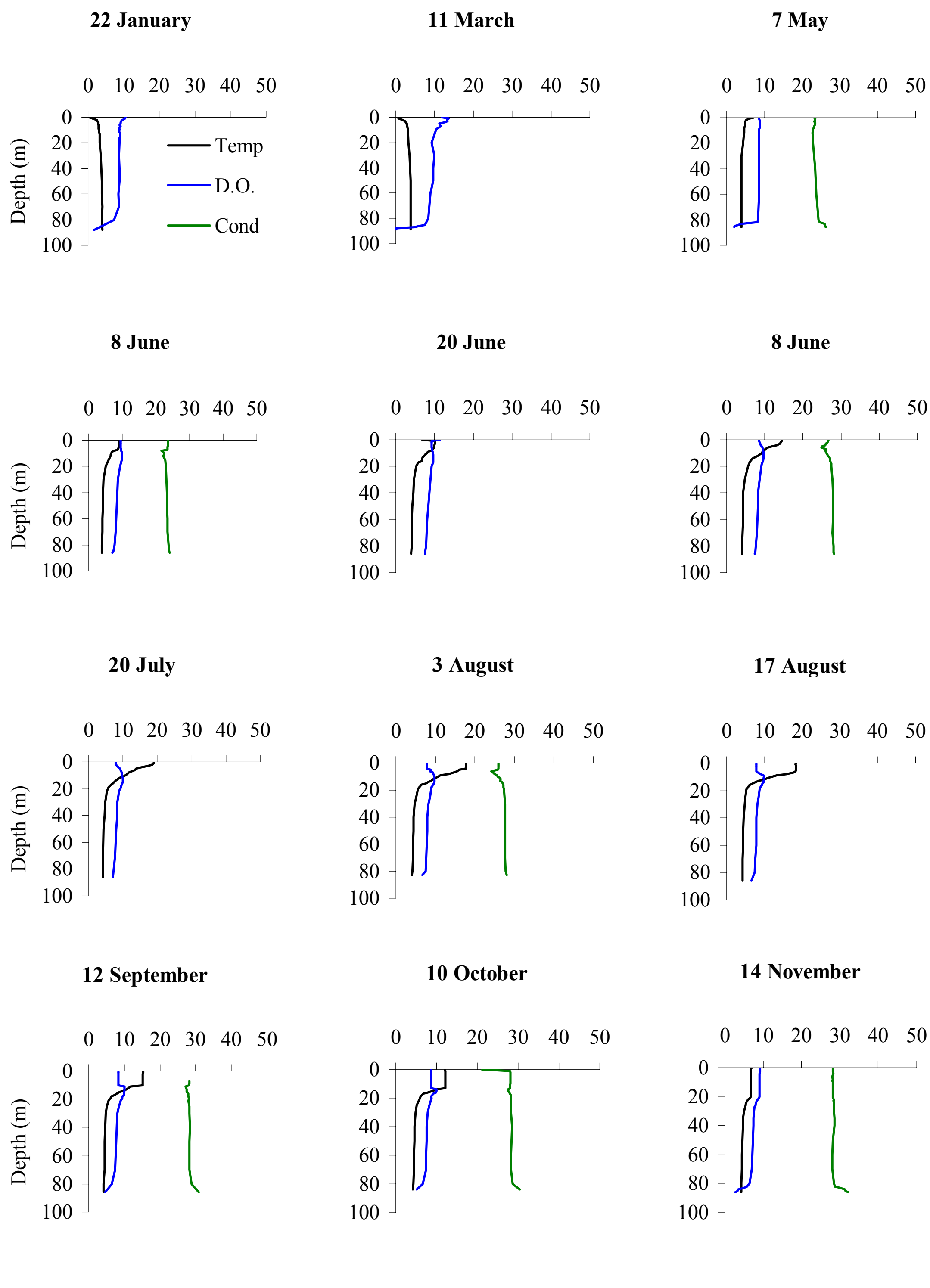

Figure 3a. Temperature (^oC), dissolved oxygen (mg/l), and conductivity (*u*S/cm) profiles for Redfish Lake, January through November 1998.

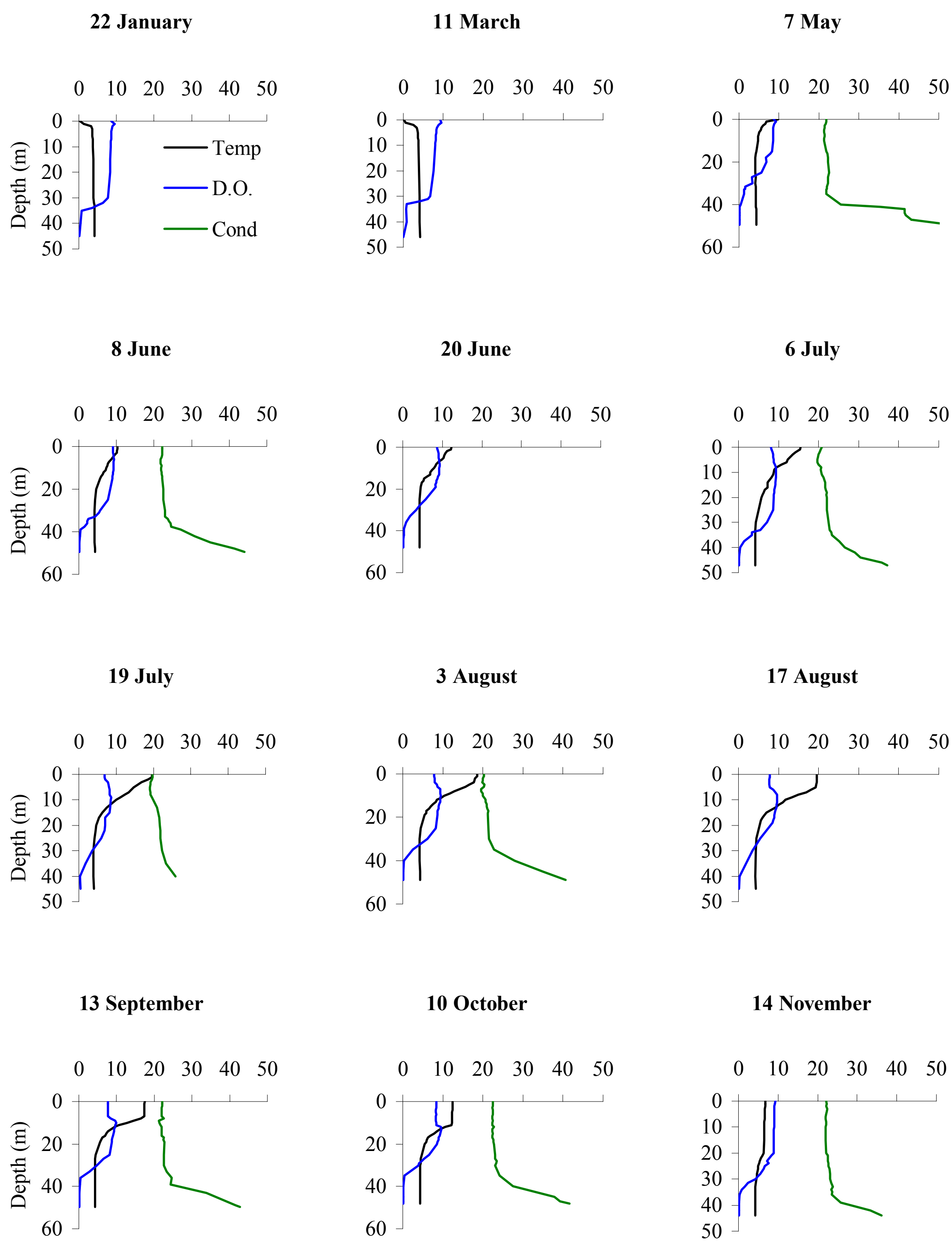

Figure 3b. Temperature (°C), dissolved oxygen (mg/l), and conductivity (*u*S/cm) profiles for Pettit Lake, January through November 1998.

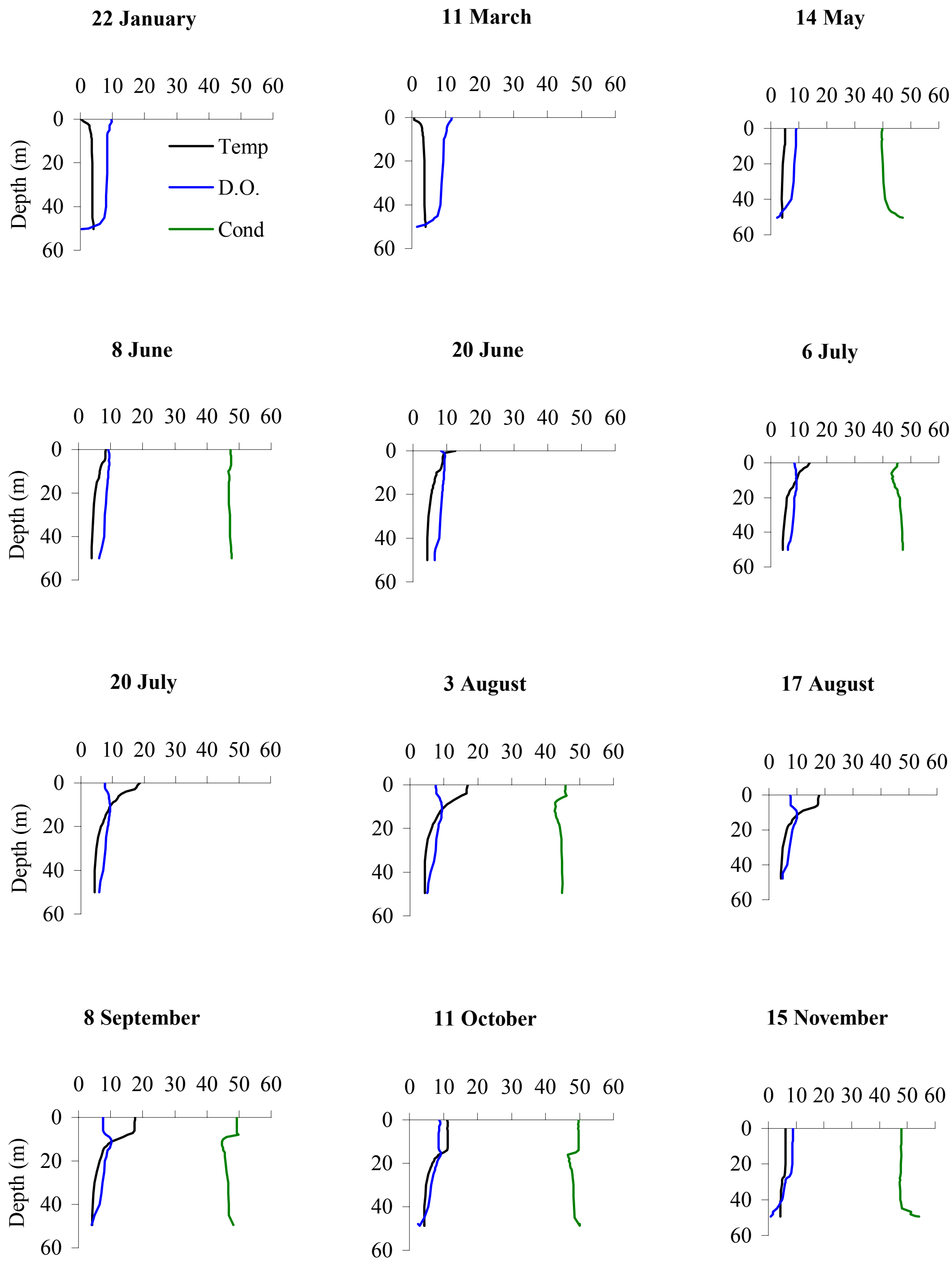

Figure 3c. Temperature (^{o}C), dissolved oxygen (mg/l), and conductivity (uS/cm) profiles for Alturas Lake, January through November 1998.

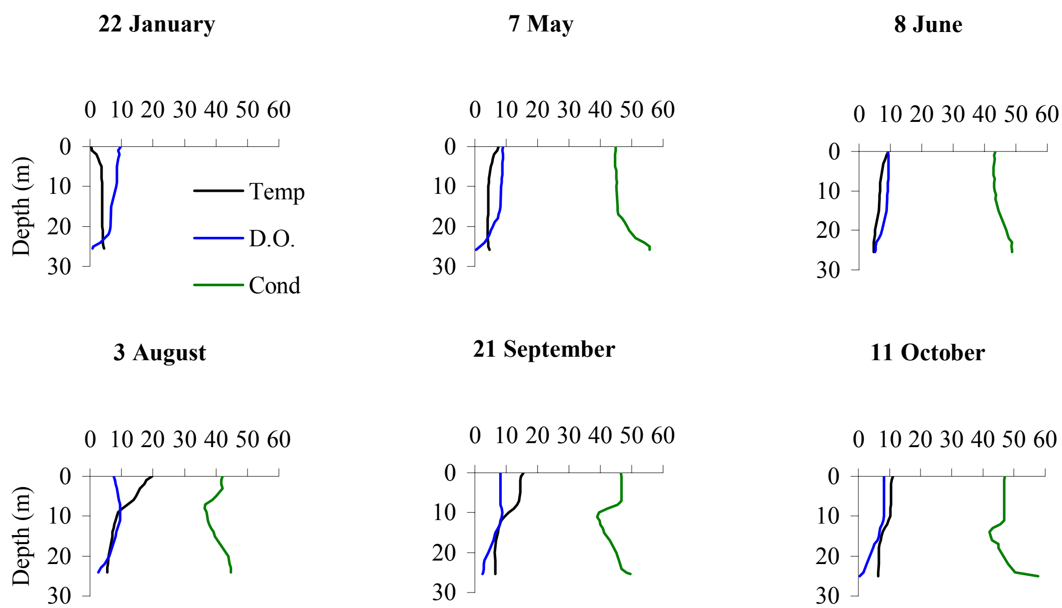

Figure 3d. Temperature (°C), dissolved oxygen (mg/l), and conductivity (*u*S/cm) profiles for Stanley Lake, January through November 1998.

Conductivities were relatively consistent throughout the year and were similar to those reported by Luecke et al. (1996), Griswold (1997) and Taki et al. (1999). Conductivities ranged from 19-23 *u*S/cm above 30 m depth and 21-43 *u*S/cm at depths greater than 30 m in Pettit, 22-32 *u*S/cm in Redfish, 40-50 *u*S/cm in Alturas and 35-57 *u*S/cm in Stanley lakes.

Seasonal mean epilimnetic water temperatures were 11.9, 12.3, 11.3, and 10.8 °C in Redfish, Pettit, Alturas, and Stanley lakes, respectively. Seasonal mean surface water temperatures were higher than in 1996 and 1997 but less than the 7 year average in Redfish, Pettit, and Alturas lakes. Stanley Lake was warmer than in 1996 and similar to 1997 and less than the 7 year average (Table 4).

Table 4. Seasonal mean (May-October) surface water temperature (^oC), water transparency (m), depth of 1% light level (m), epilimnetic chlorophyll *a* (*u*g/l), and whole-lake total zooplankton biomass (*u*g/l), 1992-1998.

Lake	Year	Surface Temperature (^oC) 0-10 m	Water Transparency (m)	Depth of 1% light level (m)	Epilimnetic chl a (ug/l)	Whole-lake zooplankton biomass (ug/l)
Redfish	**1992**	14.4	13.6	32.5	0.5	4.7
	1993	12.2	13.6	26.3	0.7	6.9
	1994	14.0	15.0	31.0	0.4	10.3
	1995	12.9	12.3	28.4	0.4	11.8
	1996	11.0	13.7	22.6	0.8	7.8
	1997	11.4	11.3	20.0	1.5	8.2
	1998	11.9	12.7	23.5	1.4	10.4
	mean	**12.6**	**13.2**	**26.3**	**0.8**	**8.6**
Pettit	**1992**	14.9	15.2	29.3	0.4	30.7
	1993	12.7	14.3	23.3	0.6	23.3
	1994	14.5	14.1	30.5	0.3	33.9
	1995	12.7	12.6	23.8	0.5	3.9
	1996	11.1	11.1	20.6	1.0	9.1
	1997	11.5	10.9	19.0	1.4	11.2
	1998	12.3	10.6	23.1	1.4	9.3
	mean	**12.8**	**12.7**	**24.2**	**0.8**	**17.4**
Alturas	**1992**	14.3	13.5	26.2	0.6	4.7
	1993	11.8	-	20.6	1.0	0.5
	1994	13.4	14.7	23.5	0.5	3.9
	1995	12.0	9.6	17.6	0.4	2.7
	1996	10.4	10.3	16.1	1.1	5.7
	1997	10.5	10.2	15.5	1.2	11.0
	1998	11.3	10.3	17.2	2.3	10.9
	mean	**11.9**	**11.4**	**19.5**	**1.0**	**5.6**
Stanley	**1992**	14.2	8.2	18.6	0.8	32.1
	1993	11.1	7.5	15.4	1.4	18.8
	1994	14.1	7.8	15.5	0.5	24.6
	1995	11.4	5.6	11.4	0.9	20.8
	1996	10.0	7.0	12.2	1.3	21.8
	1997	10.7	7.0	12.7	1.4	19.9
	1998	10.8	5.0	11.5	1.3	26.2
	mean	**11.8**	**6.9**	**13.9**	**1.1**	**23.5**

Water transparency and light penetration

Water transparencies in Redfish, Pettit and Alturas Lakes were lowest during January when lakes were ice covered and in June after spring mixing stimulated phytoplankton production, similar to past years (Budy et al.1996, Griswold 1997, Taki et al. 1999) (Figure 4). Transparencies in Redfish and Alturas lakes increased throughout the

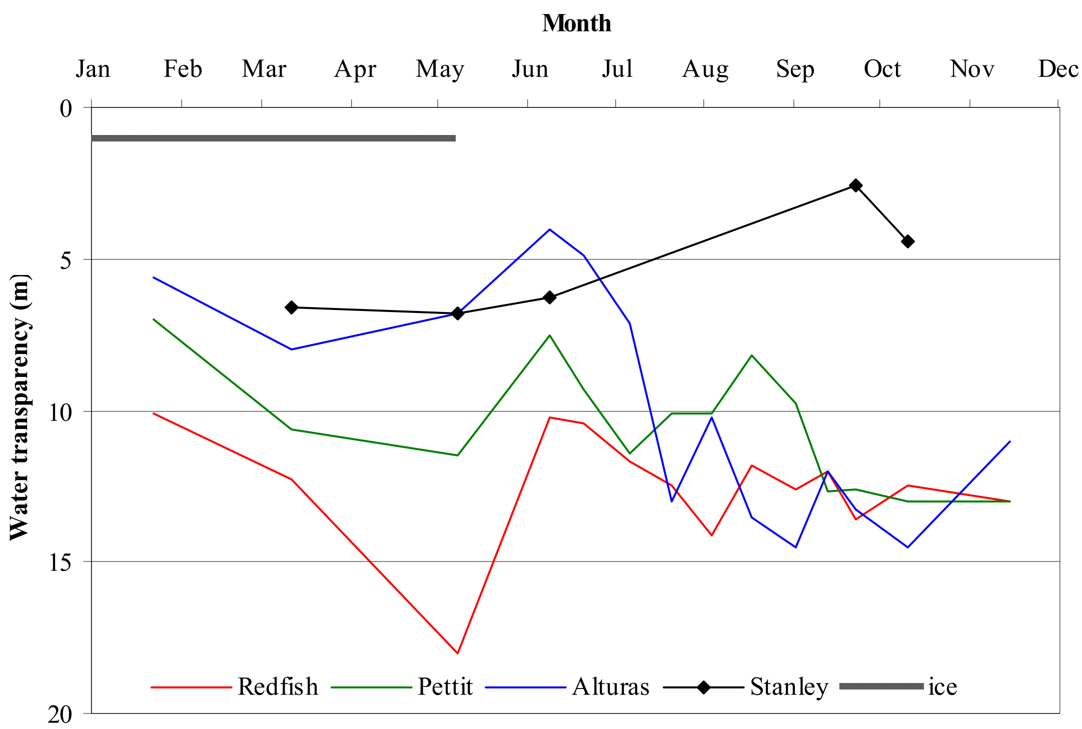

Figure 4. Water transparencies (secchi depth) in meters for Redfish, Pettit, Alturas and Stanley lakes, January through November 1998. Shaded line indicates ice cover.

summer and fall until the lakes mixed in the fall. In Pettit Lake this trend was interrupted during August when a green algae bloom reduced water transparency to 7.6 m. In Stanley Lake, water transparencies were highest in May, declined slightly in June and were very shallow by the time the lake was sampled again in late September. The extremely low transparency measurement in September (2.6 m) resulted from a storm event that introduced fine sediment into the lake that remained in suspension. The lake took on the appearance of a glacial system. Water transparency in Stanley Lake increased to 4.4 m in October, still well below October transparencies observed during 1992-1997.

Seasonal mean water transparencies (May-October) were 12.7, 10.6 and 10.3 m in Redfish, Pettit and Alturas lakes, respectively (Table 4). These values were less than the respective 7 year averages but similar to the seasonal mean water transparencies observed since 1995 when discharge increased and lake fertilization began. In Stanley Lake seasonal mean water transparency was 5.0 m, which was the lowest observed since sampling began in 1992. The lakes showed similar ranking to past years with Redfish Lake having the deepest water transparency, followed by Pettit, Alturas and Stanley lakes.

Depth of the one-percent light level followed similar trends as water transparency. Photic zones in Redfish, Pettit and Alturas Lakes were lowest during January and March when lakes were ice covered and in June or early July. Light penetration gradually increased throughout the summer except in Stanley Lake where light penetration was reduced to 9.0 m after the storm event in September (Figure 5). Seasonal mean light penetration in 1998

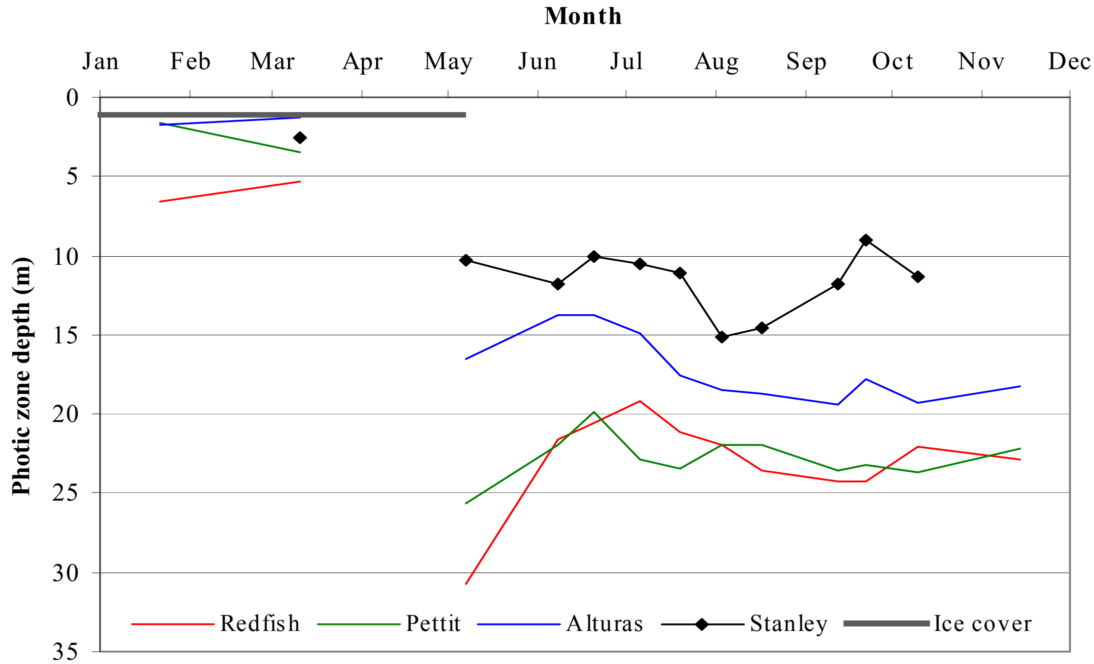

Figure 5. Photic zone depth defined by the 1% light level (in meters) for Redfish, Pettit, Alturas and Stanley lakes, January through November 1998. Shaded line indicates ice cover.

was lower than mean light penetration observed during pre-fertilized years and similar to mean light penetration in recent years with higher discharge and nutrient supplementation (Table 4). Declines in light penetration appear to be related to climatic conditions rather than nutrient supplementation. Since 1994, light penetration in Redfish Lake has declined by 21% compared to a 28% decline in unfertilized Stanley Lake. Pettit and Alturas lakes have experienced 17% and 22% reductions in light penetration since fertilization began in 1997 during which time the photic zone in Stanley declined by 17%. Seasonal mean light penetration began to decline in Pettit and Alturas lakes in 1995, two years prior to initiation of nutrient supplementation, when precipitation and discharge increased.

Water chemistry

During spring turnover (May 1998) depth integrated total phosphorus (TP) concentrations were higher than previously observed in all four lakes. TP concentrations were the highest in Alturas Lake (18.1 *u*g/l), followed by Stanley Lake (15.1 *u*g/l), Redfish Lake (11.0 *u*g/l) and Pettit Lake (10.2 *u*g/l). Although concentrations were nearly double those observed in past years, the relative rankings of the lakes were similar to previous years (Table 5).

Depth integrated total nitrogen (TN) concentrations at spring turnover were below the 7 year average in Pettit and Stanley Lakes, similar to the average in Alturas Lake and higher than the 7 year average in Redfish Lake. As a result of the elevated TP concentrations, TN:TP ratios were much lower than observed in previous years. TN:TP ratios were 7.7 in Redfish Lake, 4.8 in Pettit Lake, 3.8 in Alturas Lake and 5.1 in Stanley Lake.

Depth integrated nitrate concentrations during spring turnover were also high during spring 1998 except in Pettit Lake (Table 5). In Redfish, Alturas and Stanley lakes nitrate concentrations were approximately double the values observed in previous years. Nitrate concentrations just after spring turnover ranged from 3-37 *u*g/l and the ranking from highest to lowest was Redfish, Stanley, Alturas and Pettit. In Pettit Lake the deep sample was omitted because it was mistakenly pulled from one meter above the bottom. Because Pettit Lake does not mix completely the sample should have been pulled from one meter above the monimolimnion (perennially isolated deep-water mass). As a result the spring nutrient values were underestimated. In the other three lakes spring nutrient values were

Table 5. Nutrient concentrations (*u*g/l) and TN/TP ratio during May (after spring mixing) 1992-1998 in Redfish, Pettit, Alturas and Stanley lakes, Idaho. Concentrations are averages of three discrete depths.

Lake	Year	TP	TN	Nitrate	Ammonia	Orthophosphorus	TN/TP
Redfish	**1992**	6.5	61.0	5.5	-	1.0	9.5
	1993	8.6	52.7	6.7	-	-	6.2
	1994	5.6	-	-	-	-	-
	1995	5.0	74.2	3.8	2.0	1.0	14.8
	1996	4.8	77.0	12.7	2.3	-	16.1
	1997	6.0	-	17.0	-	-	-
	1998	11.0	82.6	37.0	5.2	1.0	7.7
	Mean	**6.8**	**69.5**	**13.8**	**3.2**	**1.0**	**10.9**
Pettit	**1992**	6.4	94.5	7.0	-	1.0	18.3
	1993	5.8	94.0	4.0	-	-	-
	1994	6.6	-	-	-	-	-
	1995	4.8	88.8	12.0	3.5	1.0	18.4
	1996	5.3	64.3	13.0	7.1	-	11.6
	1997	5.5	-	16.5	-	-	-
	1998	10.2	48.0	3.1	0.7	0.9	4.8
	Mean	**6.4**	**77.9**	**9.3**	**3.8**	**1.0**	**13.3**
Alturas	**1992**	10.0	74.0	2.0	-	2.8	7.4
	1993	9.4	72.5	3.3	-	-	8.2
	1994	13.9	-	-	-	-	-
	1995	8.2	66.4	5.8	3.5	1.4	7.7
	1996	6.0	74.6	11.6	2.2	-	12.4
	1997	10.0	-	14.7	-	-	-
	1998	18.1	69.6	17.8	3.1	1.9	3.8
	Mean	**10.8**	**71.4**	**9.2**	**2.9**	**2.0**	**7.9**
Stanley	**1992**	10.5	93.5	5.0	4.0	1.0	8.9
	1993	11.4	129.8	8.5	-	-	12.7
	1994	11.3	-	-	-	-	-
	1995	7.0	103.0	9.2	18.0	1.2	14.8
	1996	6.5	-	-	-	-	-
	1997	7.2	-	9.7	-	-	-
	1998	15.1	78.5	19.0	11.6	1.2	5.1
	Mean	**9.9**	**101.2**	**10.3**	**11.2**	**1.1**	**10.4**

calculated as the mean of all depths sampled during May, which included a surface tube sample (0-6 m), one sample from mid-water column and a sample from one to two meters above the bottom.

Nutrient concentrations in surface waters of the Sawtooth Valley lakes were variable during May-October 1998. TP concentrations ranged from 3-13 *u*g/l in Redfish, 4-9 *u*g/l in Pettit, 6-15 *u*g/l in Alturas and 6-14 *u*g/l in Stanley Lake (Figure 6). TP concentrations were highest during May and gradually declined through October 1998. Seasonal TP peak concentrations were higher than observed in 1997 and mean concentrations were equal to or slightly above the 7 year average (Table 6).

TN concentrations in surface waters were relatively stable during 1998. TN concentrations in Redfish Lake appeared to be out of synch with the other lakes. TN concentrations in Redfish were relatively high in May and August while the other three lakes exhibited peaks in June and October (Figure 6). TN concentrations ranged from 47-100 *u*g/l in the four lakes during May-October. Seasonal mean TN in Redfish and Pettit lakes were similar to the 7 year averages and to values observed during 1997. Alturas Lakes seasonal mean TN concentration was similar to the 1997 average and 12% lower than the 7 year average. Stanley Lake seasonal mean TN concentration was approximately 20% lower than the 7 year average (Table 6).

TN:TP ratios were very low in May, a result of elevated TP concentrations. As TP concentrations declined TN:TP ratios increased (Figure 6). By October TN:TP ratios were 13.4 in Redfish and Stanley Lakes, 18.4 in Pettit Lake and 14.3 in Alturas Lake. Seasonal mean TN:TP concentrations were low in all four lakes (Table 6).

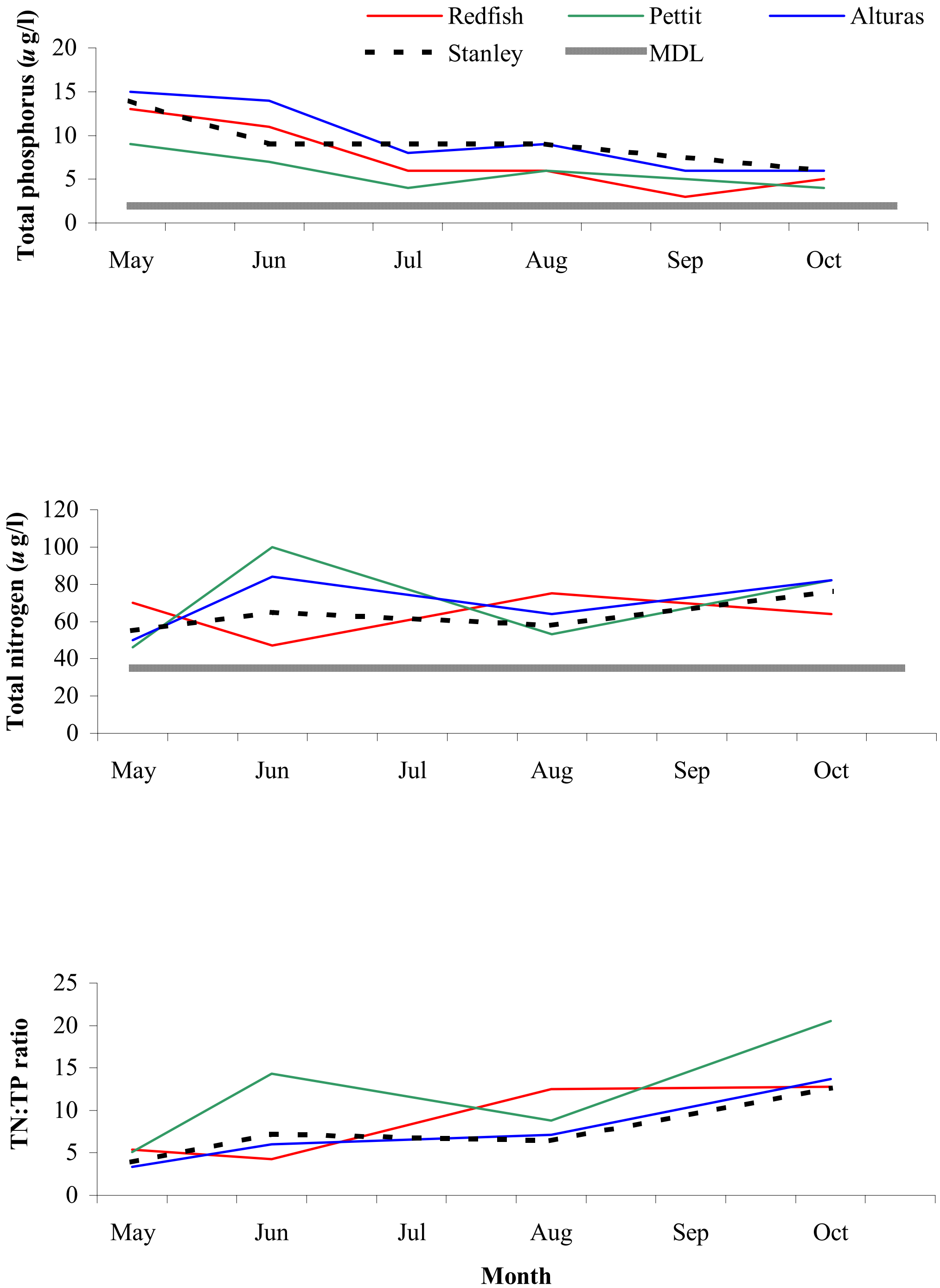

Figure 6. Epilimnetic total phosphorus (TP) and total nitrogen (TN) concentrations (*u*g/l), and the TN/TP ratio in the epilimnetic waters of Redfish, Pettit, Alturas and Stanley lakes during May through October 1998. Grey line denotes method detection level.

Table 6. Seasonal mean (May-October) epilimnetic nutrient concentrations (*u*g/l) and TN/TP ratio in Redfish, Pettit, Alturas and Stanley lakes during 1992-1998.

Lake	Year	TP	TN	Nitrate	Ammonia	Orthophosphorus	TN/TP
Redfish	**1992**	8.3	50.1	6.5	-	1.7	6.7
	1993	6.8	65.1	2.4	3.2	1.6	10.0
	1994	8.5	-	-	-	2.0	-
	1995	7.1	85.5	3.7	6.4	1.7	14.7
	1996	4.9	48.1	1.9	1.3	0.9	10.9
	1997	5.6	67.0	5.8	3.5	0.0	16.0
	1998	7.2	63.9	9.2	2.9	-	8.9
	mean	**6.9**	**63.3**	**4.9**	**3.5**	**1.3**	**11.2**
Pettit	**1992**	5.9	87.4	4.6	-	1.9	16.4
	1993	6.5	75.0	2.1	3.0	1.7	13.4
	1994	6.4	-	-	-	1.0	-
	1995	5.5	82.4	1.0	3.3	1.4	16.5
	1996	5.8	40.6	0.6	1.3	0.9	7.7
	1997	5.4	71.6	2.0	2.6	0.0	17.9
	1998	5.9	76.3	1.4	1.8	-	12.7
	mean	**5.9**	**72.2**	**2.0**	**2.4**	**1.1**	**14.1**
Alturas	**1992**	7.8	82.4	4.0	-	1.3	9.9
	1993	8.6	87.6	3.4	2.6	1.2	12.7
	1994	11.8	-	-	-	2.4	-
	1995	8.4	109.8	2.2	7.0	1.8	14.9
	1996	7.8	61.7	2.1	1.7	1.0	8.6
	1997	8.2	66.6	1.7	1.8	0.3	11.6
	1998	9.4	69.9	1.0	1.9	-	7.8
	mean	**8.9**	**79.7**	**2.4**	**3.0**	**1.3**	**10.9**
Stanley	**1992**	8.0	90.9	3.9	4.0	1.8	11.4
	1993	7.0	98.1	4.7	11.6	1.6	14.8
	1994	9.9	-	-	-	2.7	-
	1995	7.9	90.6	2.3	7.3	1.8	11.7
	1996	7.1	-	-	-	-	-
	1997	4.9	57.3	3.0	3.3	0.0	13.7
	1998	9.3	63.6	1.2	1.9	-	7.9
	mean	**7.7**	**80.1**	**3.0**	**5.6**	**1.6**	**11.9**

Nitrate concentrations were extremely low (below method detection levels) during May-October 1998 in Pettit, Alturas and Stanley lakes (Figure 7). Redfish Lake had high nitrate concentrations especially during May (19 *u*g/l) and August (15 *u*g/l). By October nitrate concentrations in Redfish Lake dropped below the method detection level (2.0 *u*g/l). Ammonia concentrations remained below methods detection levels during May-October 1998, except during August in Redfish, Pettit and Alturas lakes (Figure 7).

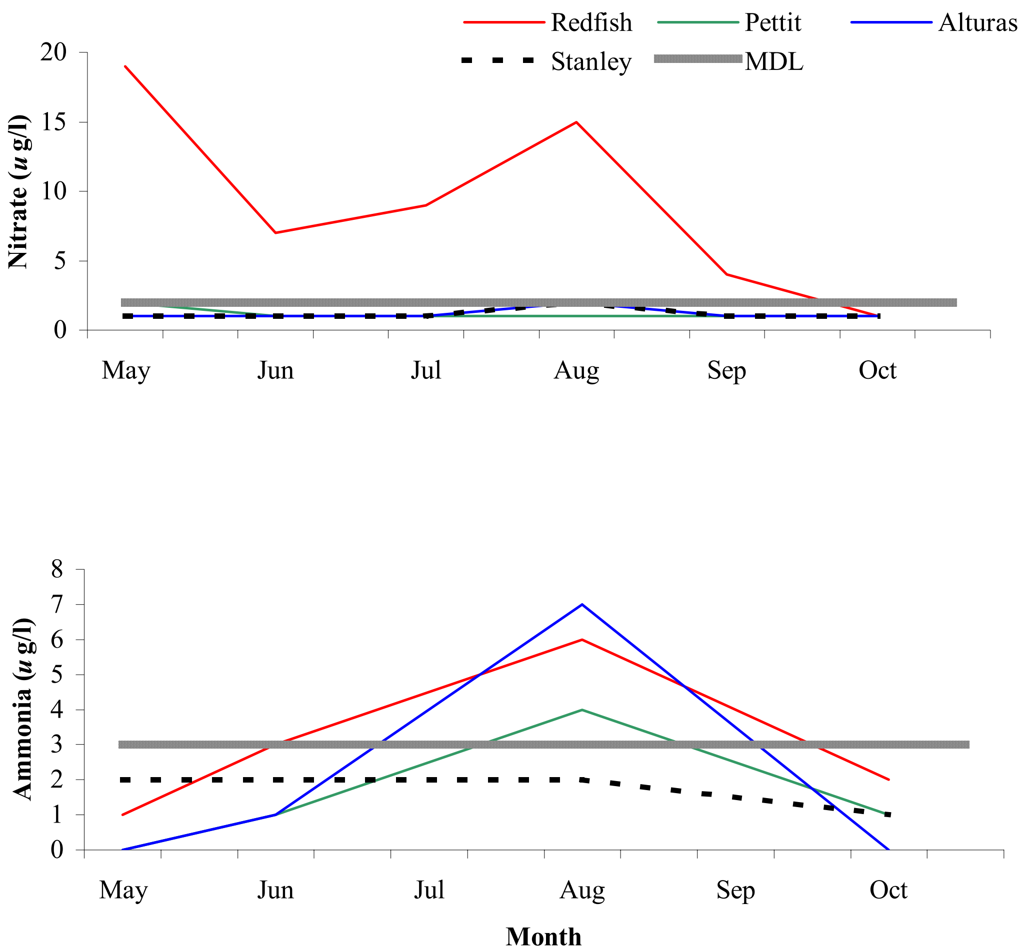

Figure 7. Epilimnetic nitrate and ammonia concentrations (*u*g) in the epilimnetic waters of Redfish, Pettit, Alturas and Stanley lakes during May through October 1998. Grey line denotes method detection level.

Chlorophyll a and phytoplankton

In 1998, surface chlorophyll *a* concentrations ranged from 0.4 to 6.3 *u*g/l in the four Sawtooth Valley lakes. Chlorophyll *a* concentrations during January were 4.1 *u*g/l in Alturas, 1.7 *u*g/l in Redfish and Pettit Lakes and 1.0 *u*g/l in Stanley Lake. During the ice-free season surface chlorophyll *a* concentrations peaked during May and June in Stanley and Alturas lakes, respectively (Figure 8). During this time chlorophyll *a* concentrations were 6.3 *u*g/l in Alturas Lake and 2.7 *u*g/l in Stanley Lake. Redfish and Pettit lakes had June peaks of 2.0 *u*g/l, however these concentrations were exceeded later in the season. Pettit Lake had an ice-free peak of 2.2 *u*g/l in August and Redfish Lake had concentrations of 2.1 *u*g/l in October and November.

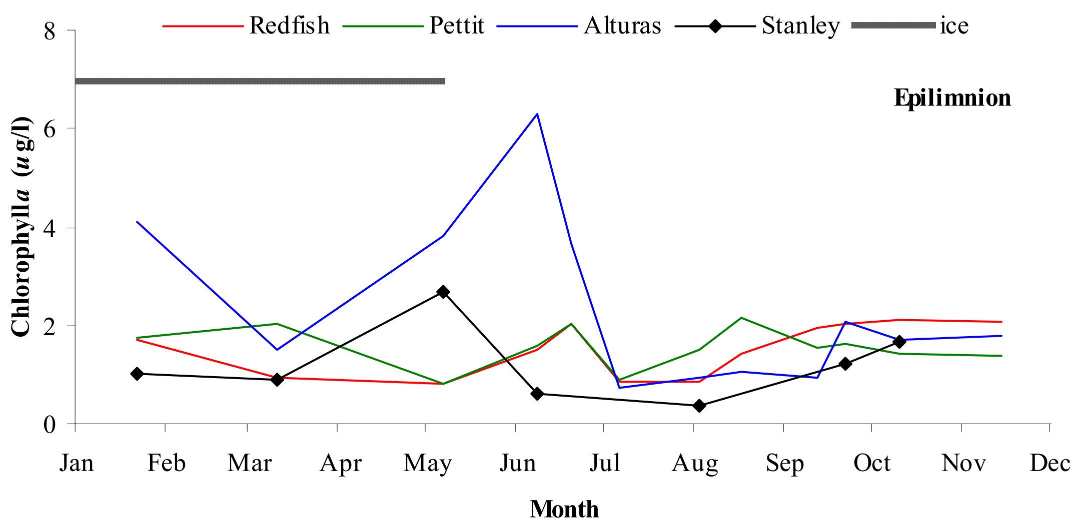

Figure 8. Surface chlorophyll *a* concentrations (*u*g/l) in Redfish, Pettit, Alturas and Stanley lakes, Idaho during 1998. Shaded line indicates ice cover.

Seasonal mean surface chlorophyll *a* concentrations in Redfish and Pettit lakes were 1.4 *u*g/l, which was similar to seasonal mean values in 1997 and much higher than during 1992-1995 (Table 4). Alturas Lake had a seasonal mean surface chlorophyll *a* concentration of 2.3 *u*g/l the highest observed in any of the Sawtooth Lakes since investigations began in 1992. This was a result of the high concentrations observed during May and June 1998.

Prior to lake fertilization (1992-1994), average seasonal mean chlorophyll *a* concentration in Redfish Lake surface waters was 0.5 *u*g/l. During lake fertilization (1995-98) the average seasonal mean chlorophyll *a* concentration was 1.0 *u*g/l, a 91% increase. During the same time periods in Stanley Lake, chlorophyll *a* concentrations increased from 0.9 to 1.2 *u*g/l, a 34% increase. In Pettit and Alturas lakes, chlorophyll *a* concentrations increased from a pre-fertilized (1992-96) average of 0.6 and 0.7 *u*g/l, respectively to 1.4 and 1.8 *u*g/l. for 1997-1998. During the same time periods Stanley Lake increased from 1.0 to 1.3 *u*g/l chlorophyll *a*. The relative increases in chlorophyll *a* concentrations for Pettit, Alturas and Stanley lakes were 146%, 140% and 37%, respectively.

Chlorophyll *a* concentrations at the 1% light level were highest in Pettit Lake, especially during late July when chlorophyll *a* peaked at 5.4 *u*g/l (Figure 9). Pettit Lake remained

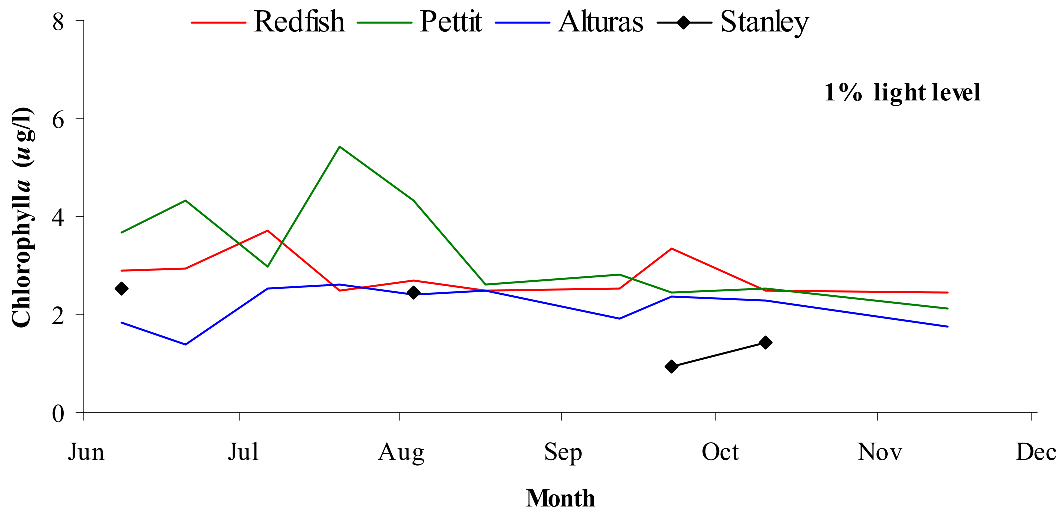

Figure 9. Chlorophyll *a* concentrations (*u*g/l) at the 1% light level in Redfish, Pettit, Alturas and Stanley lakes, Idaho during June through November 1998.

high relative to the other lakes until mid August. Redfish Lake peaked during early July when chlorophyll *a* at the 1% light level was 3.7 *u*g/l. Alturas and Stanley Lakes had lower and less variable chlorophyll *a* concentrations at the 1% light level. Concentrations in Alturas and Stanley Lakes ranged from 1.4-2.6 *u*g/l and 1.5–2.5 *u*g/l, respectively. By late August, chlorophyll *a* concentrations at the 1% light level in Redfish, Pettit and Alturas lakes were between 2-3 *u*g/l where they remained through October. During September and October chlorophyll *a* concentrations in Stanley Lake were lower than in the three fertilized lakes (0.9–1.5 *u*g/l).

Phytoplankton communities in the Sawtooth Valley lakes were dominated by small grazable taxa in 1998. Total phytoplankton densities ranged from 670-12,110 cells/ml and total phytoplankton bio-volume ranged from 0.03 to 0.94 mm^3/l in the four lakes. Generally, Chryso- and Cryptophycean nano-flagellates and autotrophic picoplankton (Cyanophceae) were numerically dominant while Chryso- and Cryptophycean nano-flagellates and Dinophycean dinoflagellates had the highest bio-volume of any phytoplankton taxa. Diatoms (Bacillariophytes) were common during June and July and Chlorophyceans during August and September.

In the epilimnion of Redfish Lake, total phytoplankton densities ranged from 1400-4600 cells/ml, lower than the 2200-6000 cells/ml observed in 1997 (Figure 10a). The small Cyanophyte *Synechococcus* sp. (<2 *u*m) and Chryso- and Cryptophycean nano-flagellates were numerically dominant. Total bio-volume in the epilimnion was 0.10 to 0.53 mm^3/l, which was higher than estimates made in 1997 but still lower than in 1995 when epilimnetic bio-volume ranged from approximately 0.25 to 0.8 mm^3/l (Luecke et al. 1996).

Epilimnetic density and bio-volume exhibited peaks in June and October. During June, Diatoms (*Synedra* sp. and *Cyclotella* sp.), Chryso- and Cryptophycean nano-flagellates (*Chrysochromulina* sp., *Chromulina* sp., and *Rhodomonas* sp.) and Dinophyceans (*Peridinium* sp. and *Gymnodinium* sp.) dominated bio-volume. During August and September the Chlorophycean, *Oocystis* sp. was also a dominant taxon. Chryso- and Cryptophycean nano-flagellates (*Chrysochromulina* sp., *Chromulina* sp.) and Dinophyceans (*Peridinium* sp. and *Gymnodinium* sp.) dominated bio-volume in October.

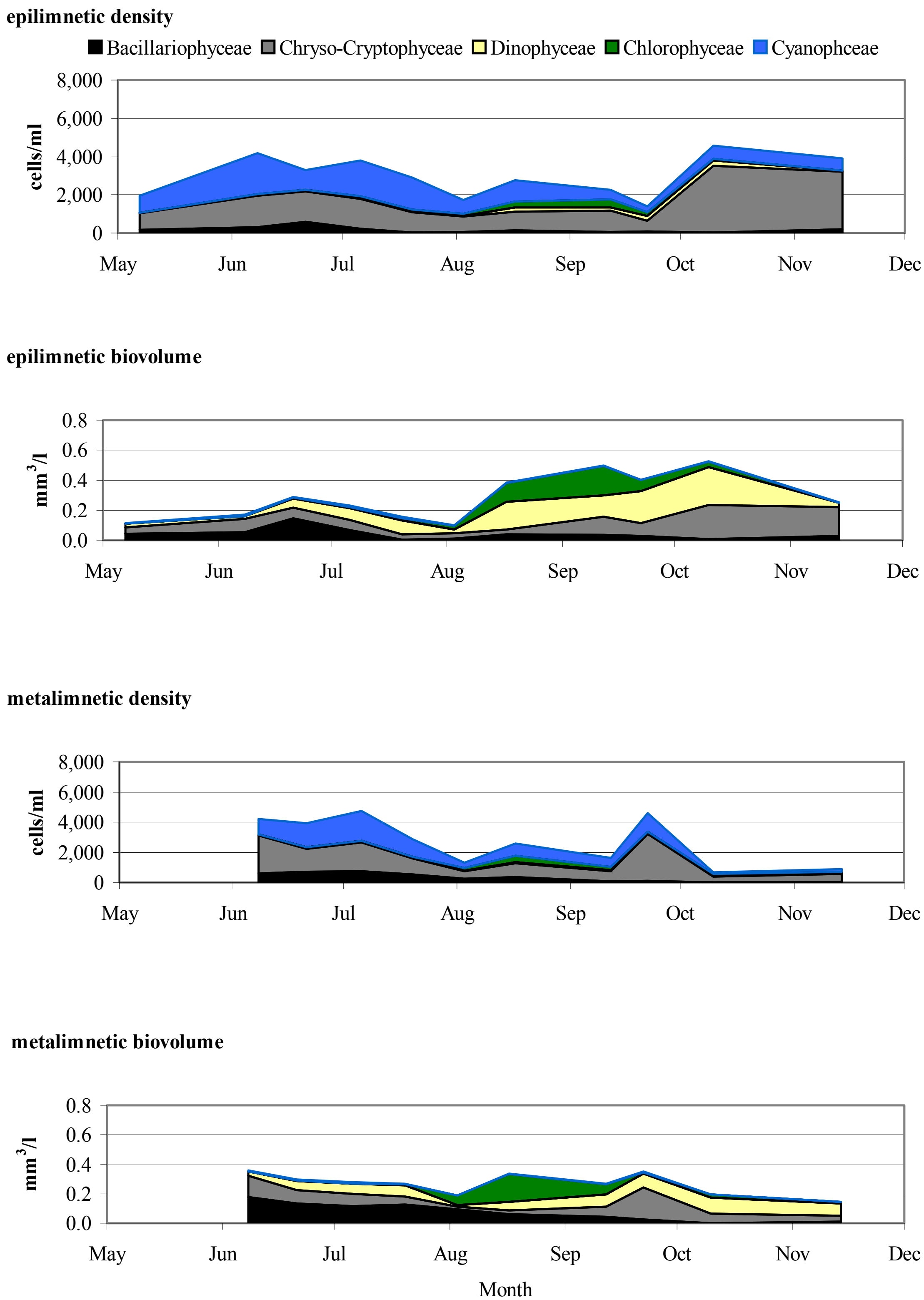

Figure 10a. Epilimnetic and metalimnetic density (cells/ml) and bio-volume (mm^3/l) of phytoplankton in Redfish Lake, Idaho during May through October 1998.

Species composition of epilimnetic bio-volume was similar during 1995, 1997 and 1998, with bio-volume fairly evenly split between taxa.

In the metalimnion, total phytoplankton densities ranged from 670 to 4,740 cells/ml. Chryso- and Cryptophycean nano-flagellates were the most abundant taxa in the metalimnion and consisted primarily of *Chrysochromulina* sp. and *Chromulina* sp. The Cyanophyte *Synechococcus* sp. was the next most abundant. Phytoplankton densities peaked during June (4740 cells/ml) and September (4600 cells/ml). After the September peak, phytoplankton density fell below 1000 cells/ml, which coincided with the increase in epilimnetic density.

Total metalimnetic phytoplankton bio-volume reached a peak of 0.36 mm^3/l during June, much lower than the 0.76 mm^3/l peak observed during late July 1997 and lower than the estimates obtained in July 1995 (approximately 0.55 mm^3/l) and August 1995 and 1997 (approximately 0.8 mm^3/l) (Luecke et al. 1996, Taki et al. 1999). Relative to the epilimnion, metalimnetic bio-volume was high in the early summer and low during the fall. The Bacillariophytes *Cyclotella* sp., *Synedra* sp. and *Stephanodiscus* sp. were common in the metalimnion during the early part of the growing season. Chryso- and Cryptophycean nano-flagellates were predominately *Chrysochromulina* sp. and *Chromulina* sp. *Dinobryon* sp. and *Rhodomonas* sp. were also common during June and July. Approximately 50% of the late September peak was a result of high *Chrysochromulina* sp. bio-volume (0.16 mm^3/l). Dinophycean bio-volume was relatively consistent throughout the growing season and consisted of *Peridinium* sp. and

Gymnodinium sp. During August and early September the Chlorophycean, *Oocystis* sp. was also a dominant taxon.

In Pettit Lake, total phytoplankton densities in the epilimnion were between 1100 and 5180 cells/ml (Figure 10b). Phytoplankton density peaked in late September resulting from increases in Chryso- and Cryptophycean nano-flagellates (primarily *Chrysochromulina* sp.) and the Cyanophyte *Synechococcus* sp. Phytoplankton density and bio-volume in the epilimnion was similar to Redfish Lake, although the spring diatom bloom was less pronounced and the mid-summer *Oocystis* sp. bloom was larger. The Chryso- and Cryptophycean bloom in late September was of shorter duration, which caused epilimnetic phytoplankton densities to decline to 1200-1500 cells/ml (bio-volume 0.12-0.26 mm^3/l) in October and November. In the metalimnion, phytoplankton populations were relatively low and stable. In mid September Chryso- and Cryptophycean nano-flagellates and *Synechococcus* sp. attained seasonal maximums resulting in a seasonal peak density of 5600 cells/ml and bio-volume of 0.42 mm^3/l. In 1998, Pettit Lake did not have the large *Synechococcus* sp. blooms that were observed in 1997 and diatoms were present in low numbers/bio-volume compared to 1997.

Epilimnetic phytoplankton densities were variable in Alturas Lake during 1998, ranging from 900-12,100 cells/ml (Figure 10c). Peak density occurred in early June and resulted in surface chlorophyll *a* concentrations of 6.3 *u*g/l. This peak resulted from increases in the Chryso-Cryptophyceans, *Chrysochromulina* sp. and *Chromulina* sp. Cyanophyte densities were stable and consisted of *Synechococcus* sp. and some *Oscillatoria* sp.

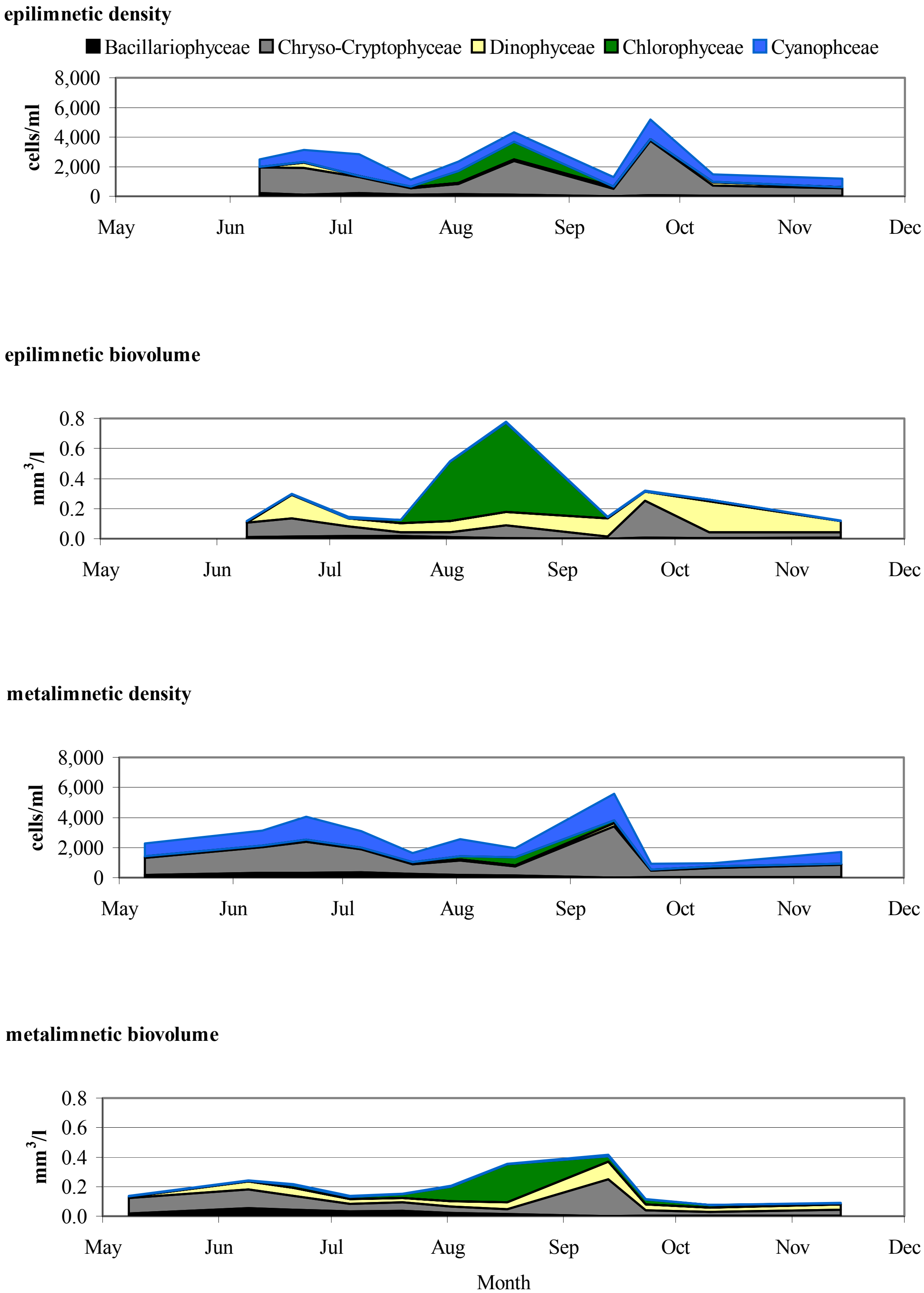

Figure 10b. Epilimnetic and metalimnetic density (cells/ml) and bio-volume (mm^3/l) of phytoplankton in Pettit Lake, Idaho during May through October 1998.

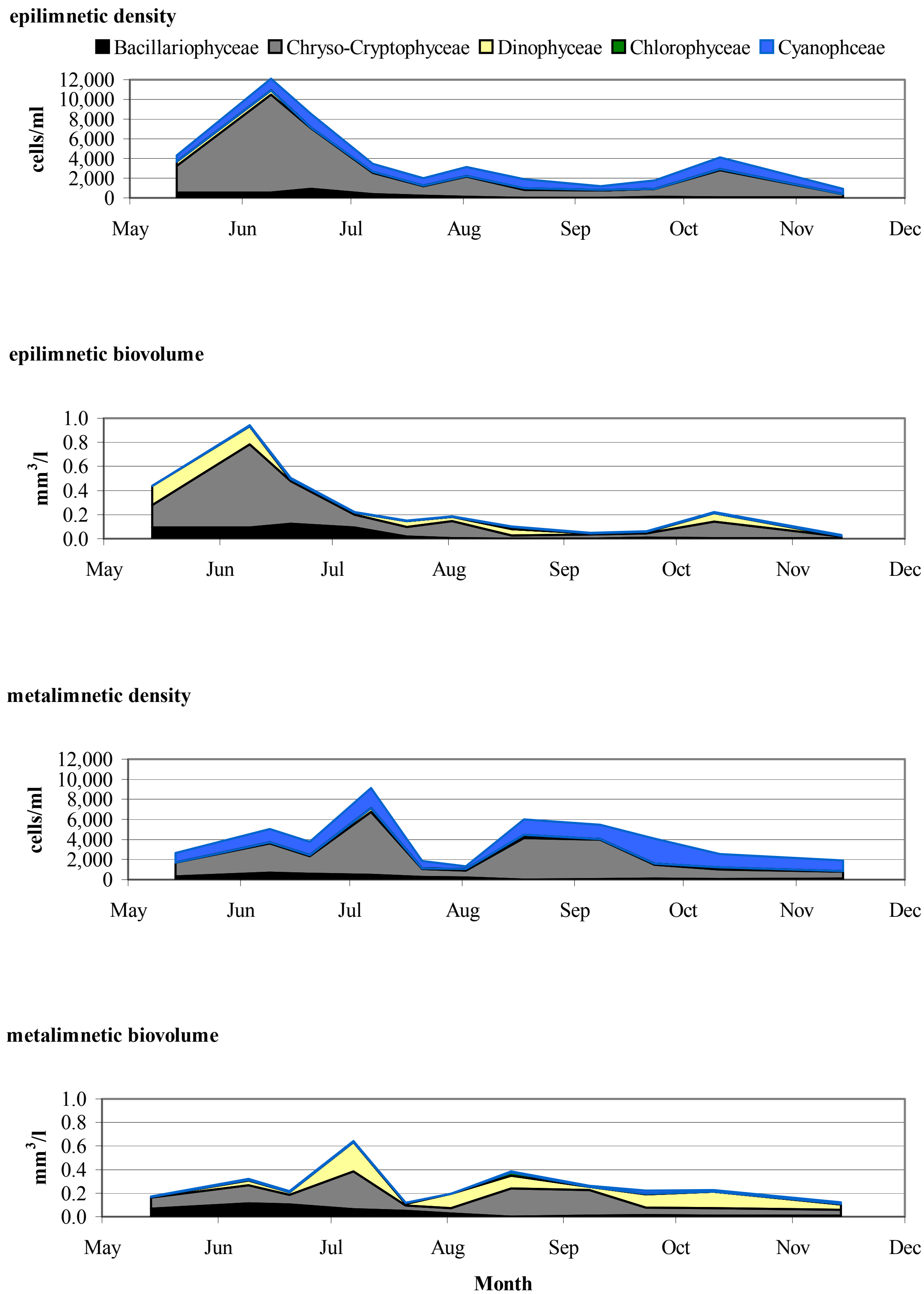

Figure 10c. Epilimnetic and metalimnetic density (cells/ml) and bio-volume (mm^3/l) of phytoplankton in Alturas Lake, Idaho during May through October 1998.

Diatoms (*Cyclotella* sp, *Asterionella formosa* and *Rhizosolenia* sp.) were present at low densities during May and June. Chryso-Cryptophyceans peaked again in October when *Chrysochromulina* sp., *Chromulina* sp. and *Rhodomonas* sp. increased. Epilimnetic bio-volume ranged from 0.03 to 0.94 mm^3/l. Bio-volume was predominately Chryso-Cryptophyceans (*Chrysochromulina* sp., *Dinobryon* sp. and *Chromulina* sp.), Dinophyceans (*Peridinium* sp.) and in the spring and early summer the diatoms, *Cyclotella* sp. and *Asterionella formosa*. Species assemblages were similar in the metalimnion although peaks in density and biomass occurred in early July and late August-early September. Density ranged from 1330-9100 cells/ml and bio-volume from 0.12-0.64 mm^3/l. The peak that occurred in July resulted from blooms of *Chrysochromulina* sp., *Chromulina* sp., *Peridinium* sp. and *Gymnodinium* sp.

Stanley Lake phytoplankton populations were sampled less frequently than the other Sawtooth Valley lakes during 1998. During June through October total phytoplankton density was approximately 1500-7000 cells/ml and was dominated by the Cyanophytes *Synechococcus sp. and Oscillatoria* sp. and the Chryso-Cryptophycean nano-flagellates *Chromulina* sp., *Chrysochromulina* sp. and *Chroomonas acuta* in both the epilimnion and metalimnion (Figure 10d). Chlorophyceans were extremely rare. Total bio-volume was between 0.09 and 0.15 mm^3/l in the epilimnion and 0.05-0.24 mm^3/l in the metalimnion. In the epilimnion, Chryso- and Cryptophycean nano-flagellates comprised the majority by volume and were predominately *Chrysochromulina* sp., *Chryptomonas* sp. and *Rhodomonas* sp. *Chroomonas* sp. was dominant in October. Nano-flagellates also

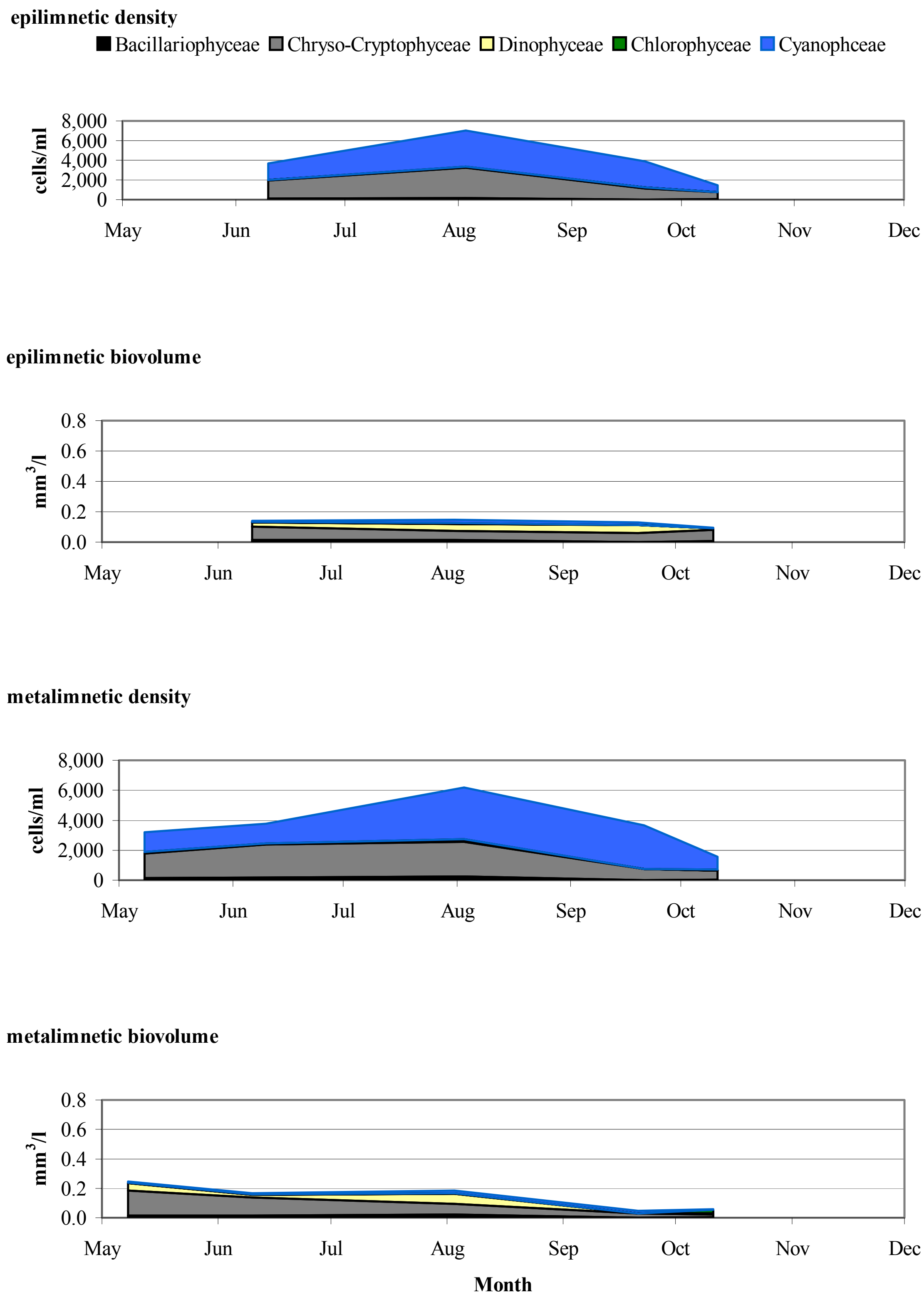

Figure 10d. Epilimnetic and metalimnetic density (cells/ml) and bio-volume (mm^3/l) of phytoplankton in Stanley Lake, Idaho during May through October 1998.

dominated bio-volume in the metalimnion but species composition differed from the epilimnion (*Chryptomonas* sp., *Chrysochromulina* sp., *Dinobryon* sp., and *Rhodomonas* sp.). The Cyanophyte *Synechococcus* sp. and diatoms *Cyclotella* sp., *Achnanthes* sp. and *Synedra acus* were present in low numbers/bio-volume throughout the growing season.

Primary productivity

The State of Washington Water Research Center at Washington State University (WRC) was contracted to estimate primary productivity in the Sawtooth Valley lakes in 1998. Depth integrated hourly and daily primary productivity was variable in the Sawtooth Lakes during June-September 1998. Productivity of the lakes was lower than in 1997, based on comparisons of hourly and daily productivity during September, except in Alturas Lake (Table 7). Mean productivity for the three fertilized lakes was 20-21 mg $C/m^2/hr$ (185-213 mg $C/m^2/day$) (Wierenga et al. 1999). Productivity in unfertilized Stanley Lake was 13 mg $C/m^2/hr$ or 118 mg $C/m^2/day$. Productivity was highest in Redfish and Pettit lakes during June. Stanley Lake had the highest productivity during July and August and Alturas peaked in September. Suspensions of nutrient enhancement activities in Pettit and Alturas lakes provide evidence that fertilization increased primary productivity. In both lakes, primary productivity estimates were lowest when nutrient supplementation activities were suspended. In Pettit Lake, the lowest estimate (151 mg $C/m^2/day$) was in September, 32 days after nutrient supplementation was suspended. Alturas Lake had the lowest productivity estimate of the year on 16 July (81 mg $C/m^2/day$), 26 days after suspension of nutrient applications in that lake.

Table 7. Hourly (mg C/m^2/hour) and daily (mg C/m^2/day) estimates of primary productivity in Redfish, Pettit, Alturas and Stanley lakes for years 1993, 1995-1998.

		Hourly Primary Productivity (mgC/m^2/hr)					
Year	**Lake**	**June**	**July**	**August**	**September**	**October**	**Mean**
1993	**Redfish**	3.2	14.6	13.4	16.2	--	11.8
	Pettit	--	10.6	7.2	13.3	--	10.4
	Alturas	--	--	22.8	--	--	22.8
	Stanley	--	--	11.2	--	--	11.2
1995	**Redfish**	16.5	46.2	23.8	34.4	--	30.2
	Pettit	9.7	16.9	26.3	23.1	--	19.0
	Alturas	8.4	12.7	17.4	11.0	--	12.4
	Stanley	9.6	14.7	22.6	8.7	--	13.9
1996	**Redfish N**	11.3	14.7	16.5	19.0	22.9	16.9
	Redfish S	24.1	9.3	19.7	17.5	17.6	17.6
	Pettit	14.1	8.5	14.7	15.3	11.2	12.8
	Alturas	10.6	7.8	10.1	13.8	--	10.6
1997	**Redfish N**	--	--	--	47.9	25.6	36.8
	Redfish S	--	--	--	46.5	29.4	38.0
	Pettit	--	--	--	33.2	21.9	27.6
	Alturas	--	--	--	27.4	17.5	22.5
	Stanley	--	--	--	31.2	16.8	24.0
1998	**Redfish N**	24.7	14.6	20.4	25.1	--	21.2
	Redfish S	22.9	20.7	17.5	22.1	--	20.8
	Pettit	22.8	20.0	21.3	16.8	--	20.2
	Alturas	17.7	8.4	23.4	30.3	--	20.0
	Stanley	10.7	14.1	14.7	10.9	--	12.6

		Daily Primary Productivity (mgC/m^2/day)					
Year	**Lake**	**June**	**July**	**August**	**September**	**October**	**Mean**
1993	**Redfish**	24.8	113.9	104.5	126.4	--	92.4
	Pettit	--	91.2	61.7	114.4	--	89.1
	Alturas	--	--	193.8	--	--	193.8
	Stanley	--	--	110.2	--	--	110.2
1995	**Redfish**	128.7	360.4	185.6	268.3	--	235.8
	Pettit	83.4	145.3	226.2	198.7	--	163.4
	Alturas	71.2	108.0	147.9	93.5	--	105.2
	Stanley	77.8	119.1	183.1	70.3	--	112.6
1996	**Redfish N**	97.6	130.4	140.1	130.1	146.3	128.9
	Redfish S	206.3	83.6	152.5	122.7	108.4	134.7
	Pettit	117.9	68.7	130.2	148.7	88.0	110.7
	Alturas	105.4	57.0	116.4	113.2	--	98.0
1997	**Redfish N**	--	--	--	431.1	184.3	307.7
	Redfish S	--	--	--	469.7	205.8	337.8
	Pettit	--	--	--	318.7	148.9	233.8
	Alturas	--	--	--	227.4	129.5	178.5
	Stanley	--	--	--	218.4	110.9	164.7

Table 7. Continued

1998	**Redfish N**	276.5	151.0	192.8	232.8	--	213.3
	Redfish S	244.4	205.2	160.0	196.4	--	201.5
	Pettit	240.3	200.0	212.0	150.7	--	200.8
	Alturas	181.8	80.6	208.2	268.3	--	184.7
	Stanley	116.3	131.8	130.8	94.2	--	118.3

In 1998, seasonal productivity estimates in Pettit (20.2 mg C/m^2/hour) and Alturas (20.0 mg C/m^2/hour) lakes were similar to Redfish Lake (21.0 mg C/m^2/hour) productivity. The seasonal estimate of hourly productivity in Stanley Lake was 12.6 mg C/m^2/hour, only 62% of the treated lakes. This represents a shift in relative productivity for Pettit and Alturas Lakes. In 1996 and 1997, hourly productivity in Pettit and Alturas lakes was approximately 70% and 60 % of the mean productivity in Redfish Lake, respectively. Productivity in Stanley Lake was 64% of Redfish Lake in 1997 (Wurtsbaugh and Budy 1997, Taki et al. 1999).

Under fertilization average daily primary productivity in Redfish Lake increased approximately 143% relative to 1993 (Figure 11). During the same time period productivity in Stanley Lake increased by 20%. During 1997-1998, years when Pettit and Alturas lakes were fertilized, Pettit Lake productivity increased 79%, while Alturas and Stanley lakes were 37% and 27% above the 1993-1996 average. In Alturas Lake the relative increase in daily productivity could have been underestimated, because of the relatively high value obtained in 1993 that was based on a single data point.

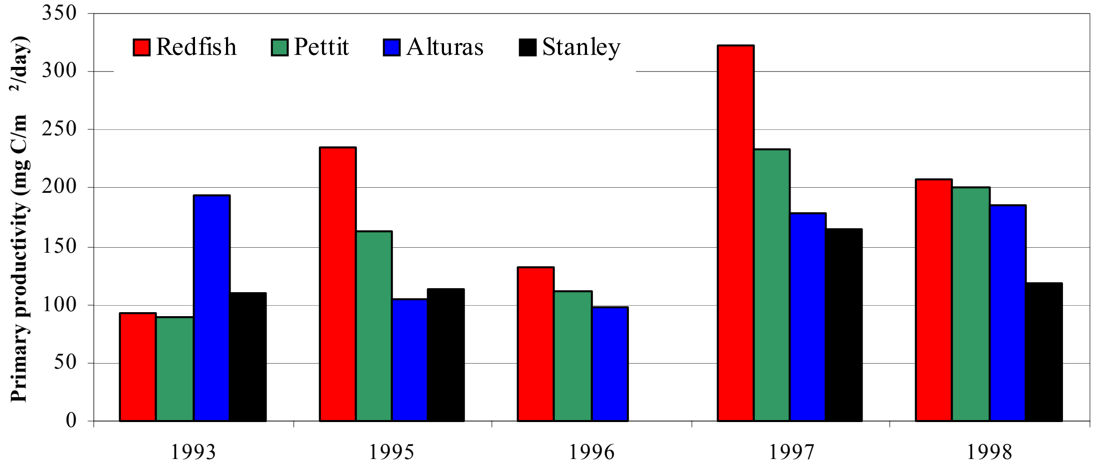

Figure 11. Mean seasonal primary productivity estimates (mg C/m^2/day) in Redfish, Pettit, Alturas, and Stanley lakes, Idaho during 1993 and 1995-1998.

Zooplankton

Annual zooplankton biomass peaks occurred during late July in Pettit Lake, August in Redfish and Stanley lakes and early September in Alturas Lake. Increases in *Daphnia* sp. biomass were largely responsible for seasonal peaks in Alturas and Stanley lakes. Peak biomass in Redfish and Pettit lakes resulted from increases in *Bosmina* sp. and *Holopedium* sp. Peak biomass was highest in Alturas (37.8 *u*g/l) and Stanley Lakes (33.8 *u*g/l) and lowest in Redfish (29.1 *u*g/l) and Pettit Lakes (20.6 *u*g/l). Mean seasonal (May-October) zooplankton biomass was 26.2 *u*g/l in Stanley Lake, 10.9 *u*g/l in Alturas Lake, 10.4 *u*g/l in Redfish Lake and 9.3 *u*g/l in Pettit Lake (Table 4)

In 1998, Redfish Lake seasonal mean zooplankton biomass was higher than in 1996 and 1997, however *Daphnia* sp. biomass was the lowest since 1993 and *Bosmina* sp. was relatively high, an indication that the system may be stressed by elevated grazing pressure (Figure 12a). During January and March whole-lake zooplankton biomass was relatively high (9.7 and 4.3 *u*g/l, respectively) and was dominated by *Bosmina* sp. and cyclopoid copepods. During summer, biomass was comprised of *Bosmina* sp., *Holopedium* sp. and

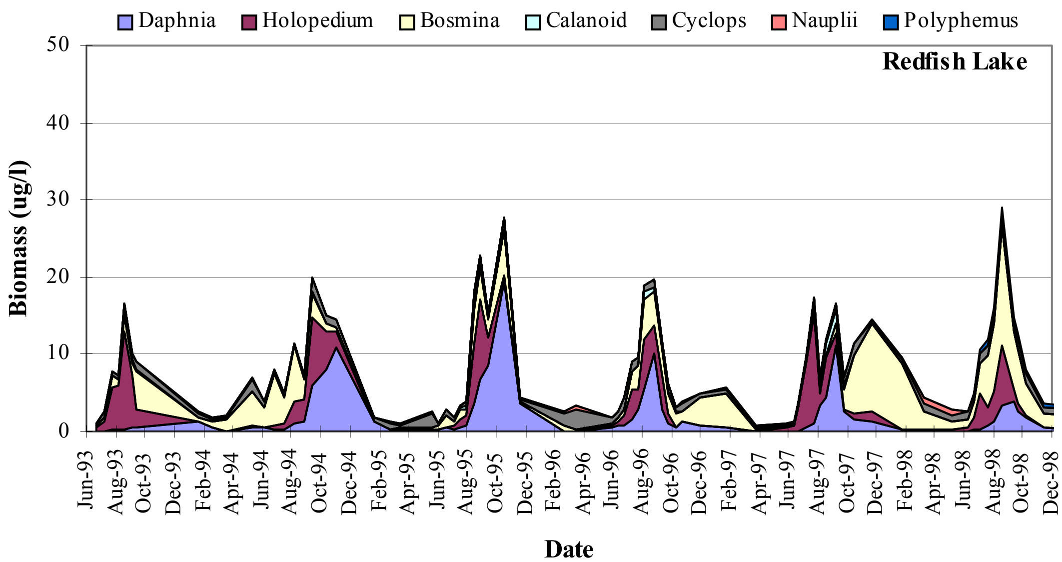

Figure 12a. Redfish Lake zooplankton biomass (*u*g/l) weighted by lake volume, 1993-1998.

Daphnia sp. *Daphnia* sp. biomass peaked in early September 1998 at 4 *u*g/l, well below the 10-11 *u*g/l peaks observed in 1994, 1996 and 1997 and the 20 *u*g/l peak observed in 1995.

Pettit Lake zooplankton biomass remained depressed compared to 1993 and 1994, when biomass reached 40-50 *u*g/l (Figure 12b). More importantly, species composition was dominated by the small-bodied *Bosmina* sp., a major shift from 1994 when *Daphnia* sp. and cyclopoid copepods were abundant. Pettit Lake biomass peaked in late July at 20.6 *u*g/l., coinciding with a peak in *Bosmina* sp. and *Holopedium* sp. *Daphnia* sp. reached a maximum biomass of only 0.5 *u*g/l during September. During January 1998, total zooplankton biomass was 2.7 *u*g/l and was predominately *Bosmina* sp. and nauplii. In March and May 1998, total biomass was 1.5-1.8 *u*g/l and was predominantly *Bosmina* sp. and cyclopoid copepods. This low winter biomass is similar to that observed in 1997, but lower than previous years.

In 1998, Alturas Lake zooplankton biomass and species composition was similar to 1997 (Figure 12c). Alturas Lake zooplankton populations experienced a collapse in the early 1990's and began to recover in 1996. During August-October 1998, zooplankton populations consisted predominantly of *Daphnia* sp. and *Bosmina* sp. Total biomass reached 37.8 *u*g/l in early September, slightly below peak biomass in 1997 and much higher than peaks observed between 1992 and 1996. Prior to 1996, the zooplankton community was almost exclusively *Bosmina* sp. *Daphnia* sp. biomass peaked at 22.7 *u*g/l in 1998, during the summers of 1996 and 1997 *Daphnia* sp. biomass reached 12 and 30 *u*g/l, respectively. Total zooplankton biomass during the winter was the highest observed to date in Alturas Lake and was mostly composed of *Bosmin*a sp. and cyclopoid copepods.

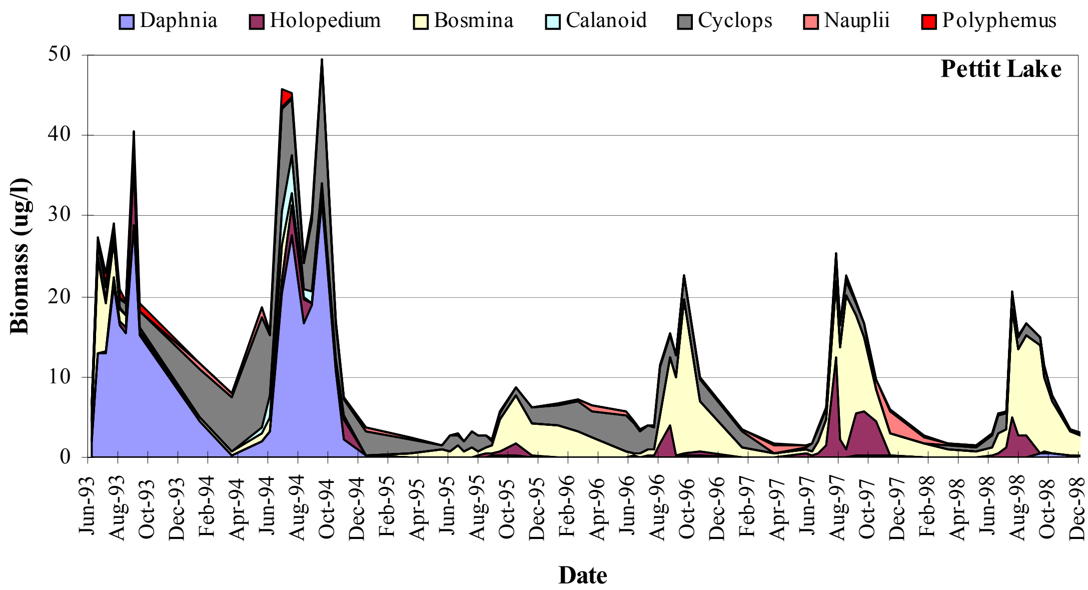

Figure 12b. Pettit Lake zooplankton biomass (*u*g/l) weighted by lake volume, 1993-1998.

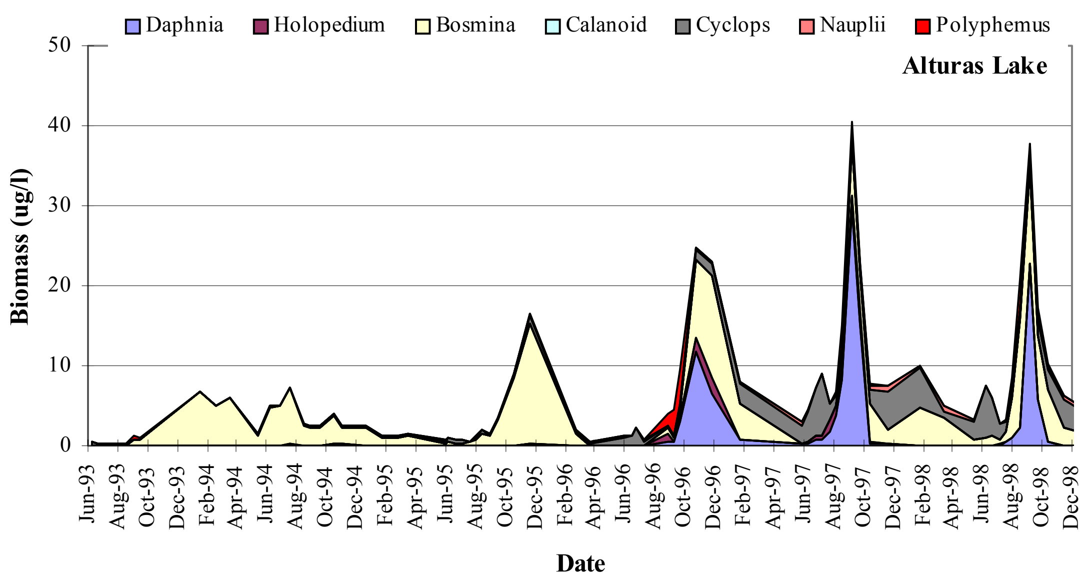

Figure 12c. Alturas Lake zooplankton biomass (*u*g/l) weighted by lake volume, 1993-1998.

Stanley Lake zooplankton populations were relatively stable in 1998. Peak zooplankton biomass was the lowest observed to date but the seasonal mean was the highest observed since 1992. During summer 1998, zooplankton species composition was similar to that observed in 1995-1997, with most biomass represented by *Daphnia* sp. and calanoid copepods (Figure12d). *Holopedium* sp. accounted for less biomass than previously observed and *Daphnia* sp. and calanoid copepods had the highest seasonal mean biomass observed to date. The summer peak occurred in August, a result of the seasonal peak biomass of *Daphnia* sp. (21.1 *u*g/l), *Holopedium* sp. (3.2 *u*g/l) and relatively high calanoid copepod biomass (7.6 *u*g/l). In March 1998, total biomass was 2.8 *u*g/l and was predominately nauplii and cyclopoid copepods.

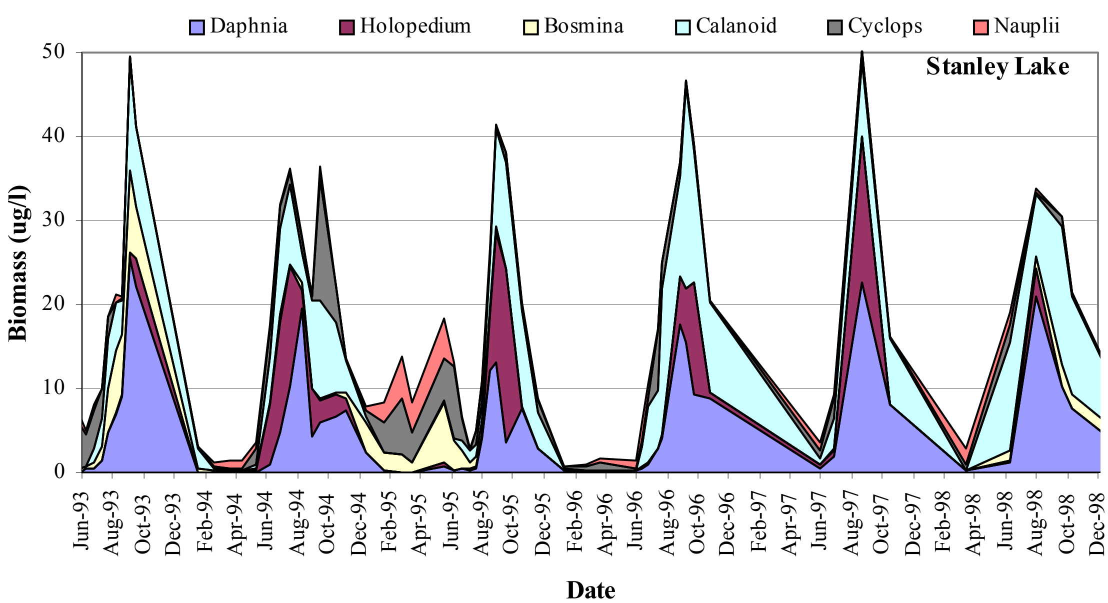

Figure 12d. Stanley Lake zooplankton biomass (*u*g/l) weighted by lake volume, 1993-1998.

DISCUSSION

Supplemental nutrients were added to Redfish, Pettit, and Alturas lakes to increase primary productivity and through trophic transfer increase secondary productivity within the lakes. Nutrient supplementation was initiated in Redfish Lake in 1995 because anticipated stocking of endangered sockeye salmon combined with the existing population of kokanee were expected to exceed the lakes carrying capacity. Trawl and hydroacoustic estimates of *O. nerka* populations in Alturas and Pettit lakes have shown large fluctuations in fish abundance and/or biomass (Teuscher and Taki 1996, Taki and Mikkelsen 1997, Taki et al. 1999). During peaks in *O. nerka* population cycles, intense grazing pressure on macrozooplankton caused shifts in species composition and declines in zooplankton biomass, density, and size. In Alturas Lake, the zooplankton communities remained depressed for over 5 years following a collapse. Pettit Lake zooplankton communities collapsed in 1995 and remain depressed at this time. To reduce the risk of overgrazing the zooplankton populations in Redfish Lake, a nutrient supplementation program was initiated in 1995. Modest nutrient additions were used to help support the very few sockeye salmon stocked into Redfish Lake, during 1996. In 1997, Redfish Lake was fertilized intensively and Pettit and Alturas lakes received nutrient supplementation for the first time. In 1998 fertilization continued in Redfish, Pettit and Alturas lakes. Identifying impacts from the lake fertilization program has been confounded by changes in meteorological conditions in the basin. Prior to fertilization of Redfish Lake, snowpack and subsequent discharge in the watershed was at or below normal. Since fertilization began in 1995, snowpack and discharge has been above average in the basin.

Gross (1995) modeled nutrient loading into Redfish Lake during the record low water year of 1992 and during the normal water year of 1993 and found a positive correlation between discharge and nutrient loading. Assuming this relationship also applies to Stanley Lake, changes in water transparency, light penetration, chlorophyll *a* concentrations and primary productivity would be expected. Data from Stanley Lake, which has not received nutrient supplementation and has a stable *O. nerka* population, supports this idea. Comparisons of seasonal averages from the 1992-1994 data set compared with the 1995-1998 data set show a 22% reduction in mean seasonal water transparency, 28% decline in mean seasonal light penetration, 35% increase in mean seasonal surface chlorophyll *a*, and a 20% increase in mean daily primary productivity.

Comparison of pre- (1991-1994) and post-fertilization (1995-1998) seasonal means in Redfish Lake show smaller reductions in water transparency and light penetration and larger increases in surface chlorophyll *a* and primary productivity than occurred during the same time periods in Stanley Lake. Mean seasonal surface water transparency declined by 11%, light penetration decreased by 21%, surface chlorophyll *a* increased by 91% and mean daily primary productivity increased by 143%. Disproportionately larger shifts in Redfish Lake chlorophyll *a* and primary productivity relative to observed changes in Stanley Lake provide evidence that nutrient supplementation stimulated primary productivity in Redfish Lake. The relatively large decrease in water transparency and light penetration observed in Stanley Lake may be caused by suspended abiotic particles resulting from the interaction between discharge and the different drainage area/lake surface area ratios of the lakes.

Pettit and Alturas lakes were fertilized in 1997 and 1998. Similar difficulties were encountered in assessing the impacts of nutrient additions because of variable meteorological conditions. Since 1995, mean annual discharge in the upper Salmon River has been above average. The year 1997 had the second highest discharge observed for the period of record (1913-1998) and in 1998 discharge in the upper Salmon River was 12% higher than average. As a result it was expected that external (natural) nutrient loading would be above average and that internally regenerated nutrients would also be high, thus the utility of using pre/post data sets to evaluate the fertilization treatments was reduced. Comparison with the Stanley Lake data set for the pre-fertilized years (1991-1996) with the fertilized years (1997-1998) provides evidence that lake fertilization was effective. Mean seasonal surface water transparency declined by 20% in Pettit Lake, 15% in Alturas Lake and 17% in Stanley Lake. Depth of the 1% light level declined by 17% in Pettit Lake, 22% in Alturas Lake and 17% in Stanley Lake. Mean seasonal surface chlorophyll *a* increased by 146% in Pettit, 140% in Alturas and 37% in Stanley Lake. Primary productivity increased 79% in Pettit Lake, 37% in Alturas Lake and 27% in Stanley Lake. Again, disproportionate increases in surface chlorophyll *a* and primary productivity in Pettit Lake and surface chlorophyll *a* in Alturas Lake compared to Stanley Lake provides evidence that nutrient supplementation was effective. As expected, relative increases in primary productivity in Pettit and Alturas lakes were less than those observed in Redfish Lake. Nutrient applications in Pettit and Alturas lakes were approximately 50% of the applications in Redfish Lake on an areal basis (mg P/m^2). In addition, primary productivity in Alturas Lake during 1993 was based on a single estimate. This single estimate of 193.8 mg C/m^2/day was high relative to the other lakes in 1993 (range 89.1-110.2 mg C/m^2/day) and to Alturas Lake in 1995 (105.2 mg C/m^2/day) and 1996 (98.0 mg C/m^2/day) (Figure 11), which may underestimate the relative increase in primary productivity under fertilization. If the single 1993 estimate is excluded, average primary productivity becomes 101.6 mg C/m^2/day during 1995-1996 in Alturas Lake and primary productivity increases by 79% under fertilization.

Numerous studies have reported effects of nutrient additions by sampling before, during, and after fertilization treatments. However, these studies may be subject to misinterpretation if annual variation caused by meteorological forcing or other impacts was not accounted for (Schindler 1987). Because drought conditions existed during the early years of this study and high water years have been the norm since lake fertilization began, Stanley Lake has been used as a reference lake. However, Stanley Lake should only be considered a gross indicator of variable conditions since the lake is morphologically dissimilar to Redfish, Pettit and Alturas lakes. Stanley Lake has a drainage area 48.6 times the size of the lake compared to ratios of 17.6, 16.9 and 22.4 for Redfish, Pettit and Alturas lakes, respectively. This results in a much shorter water retention time and should result in higher nutrient loading (Gross 1995). However, the shorter retention time could also cause a higher degree of "washout" of nutrients, phytoplankton, resting stages, and eggs compared to the other Sawtooth Valley lakes, especially during high water years (Goldman et al. 1989). If Stanley Lake is more susceptible to washout, then nutrient supplementation impacts may be over-estimated in the other Sawtooth Valley lakes. The relatively large declines in water transparency and light penetration resulting from the September 1998 storm event provides an example of how Stanley Lake responds differently than the other study lakes to meteorological effects.

Cascading trophic interactions are known to influence primary productivity of lakes (Carpenter et al. 1985, Carpenter and Kitchell 1987, Carpenter and Kitchell 1988). Although phytoplankton data for the Sawtooth Valley lakes are limited it appears phytoplankton species assemblages have remained stable under fertilization and are typically oligotrophic. Small grazable autotrophic picoplankton, nano-flagellates, and diatoms dominate phytoplankton species assemblages in the Sawtooth Valley lakes. The predominance of these phytoplankton and the absence of accumulations of non-grazable taxa (sinks) are indicators of efficient energy transfer between trophic levels, which should result in improved forage production for endangered sockeye salmon (Stockner and MacIsaac 1996, Stockner 1998). Some attempts to stimulate lake productivity with nutrient additions have been unsuccessful; energy sinks developed from inefficient trophic transfer, which prevented energy flow to juvenile sockeye salmon (Stockner and Hyatt 1984, Stockner 1987, Stockner and Shortreed 1988).

Potential benefits of the lake fertilization program to sockeye salmon include increased growth and survival and a reduced risk of exceeding lake carrying capacity and causing a collapse of the macrozooplankton population. The consequences of a collapsed forage base in Redfish Lake were considered severe since Redfish Lake represents 61% of the sockeye salmon rearing habitat in the Sawtooth Valley and recovery appears to take many years as evidenced by zooplankton data from Alturas and Pettit lakes. Several potential problems are associated with lake fertilization, including increased intra-specific competition and effects of lake eutrophication. Habitat use and foraging behavior of kokanee are similar to sockeye salmon (Rieman and Myers 1992) so potential benefits to sockeye salmon from fertilization may be offset by increases in growth, survival and fecundity of kokanee. A positive response by kokanee to lake fertilization could result in increased competition with sockeye salmon in future years. Managers should carefully weigh the short-term benefits of lake fertilization against the longer-term adverse impacts of intra-specific competition.

Successful lake fertilization may increase precipitated organic matter resulting in larger oxygen deficits in deep waters. This is particularly important in Pettit Lake where *O. nerka* habitat is significantly reduced (26% by volume) by low oxygen concentrations below approximately 30 m depth. Based on dissolved oxygen profiles it appears that Pettit Lake did not mix completely in 1996, 1997 or 1998. Whether Pettit Lake ever mixes completely is unknown. Cladocerans withstand D.O. concentrations less than 1 mg/l and cyclopoid copepods are even more tolerant of low oxygen concentrations (Pennack 1989). Zooplankton biomass below 30 m in Pettit Lake was relatively high (2.8-10.4 *u*g/l) during 1998, an indication that zooplankton standing crop was not significantly reduced by hypoxic conditions in the deep waters of Pettit Lake. The effects on predator/pray balance in Pettit Lake will be minimized if zooplankton vertically migrate out of this hypoxic region or if *O. nerka* make foraging excursions into the hypoxic region. Alternatively, this hypoxic region could provide a refuge for macrozooplankton and reduce foraging opportunities for *O. nerka*. In addition, this deep, nutrient rich, isolated water mass could impact the nutrient budget of Pettit Lake. In the future, if the lake mixes deeper than it has during the late 1990's, accumulated nutrients will be released, which could result in reduced water quality and shifts in phytoplankton species assemblages. Oxygen depletion near the substrate has increased in all the Sawtooth Valley lakes and should be monitored closely in the future.

Increased natural nutrient loading combined with lake fertilization during 1995-1997 increased spring nutrient concentrations in the Sawtooth Valley lakes. Excluding Pettit Lake because of the sampling problem during May 1998, the remaining lakes had elevated nitrate and TP concentrations at spring turnover, although Stanley Lake exhibited the smallest proportional increases. Because of increased spring nutrient concentrations, lake fertilization should be initiated later in the growing season to prevent water quality degradation during spring phytoplankton blooms and violations in DEQ water quality criteria.

Macrozooplankton response to fertilization is variable but 1.2 to 2.0 fold increases in abundance and biomass have been observed (Stockner and MacIsaac 1996). In Pettit and Alturas Lakes, *O. nerka* populations have experienced major shifts in abundance and biomass with associated changes in zooplankton biomass and species composition. These changes have precluded evaluation of nutrient supplementation at the macrozooplankton level. However, in Redfish Lake it appears that macrozooplankton populations have been maintained at least in part by lake fertilization. Since 1995, *O.*

nerka biomass estimates (2x trawl) in Redfish Lake have increased by more than 3 times and exceeded the lake's unfertilized carrying capacity each year (Stockner 1997). During this time macrozooplankton biomass and sockeye salmon over-winter survival and size at emigration has remained remarkably stable (Jay Pravecek IDFG, Doug Taki, SBT, personal communication).

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REFERENCES

Britton, L. J., and P. E. Greeson. 1987. Methods for collection and analysis of aquatic biological and microbiological samples. Techniques of Water – Resources Investigations of the United States Geological Survey, Book 5, Chapter A4. United States Government Printing Office, Washington, D.C.

Budy, P., C. Luecke, and W. A. Wurtsbaugh. 1996. A comparison of four years of limnology in the Sawtooth Lakes with emphasis on fertilization in Redfish Lake. Pages 117-137 in Snake River Sockeye Salmon habitat and limnological research. Annual Report 1995. U.S. Department of Energy, Bonneville Power Administration, Division of Fish and Wildlife, Portland, Oregon.

Budy, P., C. Luecke, W. A. Wurtsbaugh, and H. Gross. 1993. Limnological conditions of Sawtooth Valley Lakes with respect to potential growth of juvenile Snake River Sockeye Salmon. In Snake River Sockeye Salmon (*Oncorhynchus nerka*) habitat/ limnological research. Annual report 1992. U.S. Department of Energy, Bonneville Power Administration, Division of Fish and Wildlife, Portland, Oregon.

Budy, P., G. Steinhart, C. Luecke, and W. A. Wurtsbaugh. 1995. Limnological investigations of Sawtooth Valley Lakes, 1994. Pages 54-64 in Snake River Sockeye Salmon habitat and limnological research. Annual report 1994. U.S. Department of Energy, Bonneville Power Administration, Division of Fish and Wildlife, Portland, Oregon.

Carpenter, S. R., and J. F. Kitchell. 1987. The temporal scale variance in limnetic primary production. American Naturalist 129: 417-433.

Carpenter, S. R., and J. F. Kitchell. 1988. Consumer control of lake productivity. BioScience, Vol. 38, No. 11: 764-769.

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

Emmett, W. W. 1975. The channels and waters of the upper Salmon River, Idaho: Hydrologic evaluation of the upper Salmon River area, Idaho. U. S. Geological Survey professional paper, Paper 870A. Washington, D.C.

Flagg, T. A., and W. C. McAuley. 1994. Redfish Lake sockeye salmon captive broodstock rearing and research, 1993. National Marine Fisheries Service Annual Report to US DOE, Bonneville Power Administration, Division of Fish and Wildlife, Portland, Oregon.

Goldman, C. R., A. Jassby, and T. Powell. 1989. Interannual fluctuations in primary production: Meteorological forcing at two subalpine lakes. Limnology and Oceanography, 34(2): 310-323.

Griswold, R. G. 1997. Limnology of the Sawtooth Lakes, 1996. Chapter 2 in Taki, D. and A. Mikkelsen. Snake River salmon habitat and limnological research. Shoshone-Bannock Tribes Annual Report to United States Department of Energy, Bonneville Power Administration, Environment, Fish and Wildlife. Portland, Oregon.

Gross, H. P. 1995. Evaluation of lake fertilization as a tool to assist in the recovery of the Snake River Sockeye Salmon (*Oncorhynchus nerka*). Master's Thesis, Utah State University.

Gross, H. P., W. A Wurtsbaugh, C. Luecke, and P. Budy. 1993. Nutrient limitation of phytoplankton in oligotrophic lakes of the Sawtooth Valley, Idaho. Pages 29-52 in Snake River Sockeye Salmon (*Oncorhynchus nerka*) habitat/limnological research. Annual report 1992. U.S. Department of Energy, Bonneville Power Administration, Division of Fish and Wildlife, Portland, Oregon.

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

Hunter, D. A., J. E. Reuter, and C. R. Goldman. Standard operating procedures. Lake Tahoe Interagency Monitoring Program. University of California at Davis, Tahoe Research Group. Unpublished.

Hyatt, K. D., and J. G. Stockner. 1985. Responses of sockeye salmon (*Oncorhynchus nerka*) to fertilization of British Columbia coastal lakes. Canadian Journal of Fisheries and Aquatic Sciences. 42: 320-331.

Johnson, K. A., and J. J. Pravecek. 1995. Snake River sockeye salmon captive broodstock research. Idaho Department of Fish and Game Annual Report, 1993 to United States Department of Energy, Bonneville Power Administration, Environment, Fish and Wildlife. Portland, Oregon.

Koenings, J. P., J. A. Edmundson, G. B. Kyle, and J.M. Edmundson. 1987. Limnology field and laboratory manual: methods for assessing aquatic production. Alaska Department of Fish and Game, Division of Fisheries Rehabilitation, Enhancement and Development. Juneau, Alaska.

Kyle, G. B. 1994. Nutrient treatment of 3 coastal Alaskan lakes: trophic level responses and sockeye salmon production trends. Alaska Fishery Research Bulletin. 1(2): 153-167.

Lagler, K. F. 1956. Freshwater Fishery Ecology, 2nd edition. WM. C. Brown Company Publishers, Dubuque, Iowa.

LeBrasseur, R. J., C.D. McAllister, and T.R. Parsons. 1979. Addition of nutrients to a lake leads to greatly increased catch of salmon. Environmental Conservation 6(3): 187-190.

LeBrasseur, R. J., and six co-authors. 1978. Enhancement of sockeye salmon (*Oncorhynchus nerka*) by lake fertilization in Great Central Lake: Summary Report. Journal of the Fisheries Research Board of Canada. 35: 1580-1596.

Luecke, C., M. Slater, and P. Budy. 1996. Limnology of Sawtooth Valley Lakes in 1995. Pages 76-91 in Snake River Sockeye Salmon habitat and limnological research. Annual Report 1995. U.S. Department of Energy, Bonneville Power Administration, Division of Fish and Wildlife, Portland, Oregon.

McCauley, E. 1984. The estimation of the abundance and biomass of zooplankton in samples. Chapter 7 in J. A. Downing and F. Rigler, editors. A manual on methods of secondary productivity in freshwaters, 2nd edition. Blackwell Scientific, Oxford, UK.

Pennack, R. W. 1989. Fresh-water invertebrates of the United States: Protozoa to Mollusca, 3rd edition. John Wiley and Sons, Inc., New York.

Rieman, B. E., and D. L. Myers. 1992. Influence of fish density and relative productivity on growth of kokanee in ten oligotrophic lakes and reservoirs in Idaho. Transactions of the American Fisheries Society. 121: 178-191.

Robinson, D. G., and W. E. Barraclough. 1978. Population estimates of sockeye salmon (*Oncorhynchus nerka*) in a fertilized oligotrophic lake. Journal of the Fisheries Research Board of Canada. 35: 851-860.

Schindler, D. W. 1987. Detecting ecosystem responses to stress. Canadian Journal of Fisheries and Aquatic Sciences. 44:6-25.

Spaulding, S. 1993. Snake River Sockeye Salmon (*Oncorhynchus nerka*) habitat/limnological research. Annual Report 1992. U.S. Department of Energy, Bonneville Power Administration, Portland, OR.

Steinhart, G., H. P. Gross, P. Budy, C. Luecke, and W. A. Wurtsbaugh. 1994. Limnological investigations and hydroacoustic surveys of Sawtooth Valley Lakes. Annual Report 1993. In Snake River Sockeye Salmon Habitat and Limnological Research. U.S. Department of Energy, Bonneville Administration, Division of Fish and Wildlife.

Stockner, J. G. 1987. Lake fertilization: the enrichment cycle and lake sockeye salmon (*Oncorhynchus nerka*) production. Canadian Special Publication of Fisheries and Aquatic Sciences 96: 198-215.

Stockner, J. G. 1997. Review: An assessment of rearing capacity of Stanley Basin lakes for sockeye salmon (*Oncorhynchus nerka*); recommendations for enhancement by whole-lake fertilization; and comments on limnological monitoring. Report to Biolines Environmental Consulting, Stanley, Idaho.

Stockner, J. G. 1998. Phytoplankton populations in 4 Sawtooth Valley sockeye nursery lakes, Idaho, 1997. Report to Biolines Environmental Consulting, Stanley, Idaho.

Stockner, J. G., and K. D. Hyatt. 1984. Lake Fertilization: State of the art after 7 years of application. Can. Tech. Report of Fish. and Aquat. Sci., No. 1324: 33p.

Stockner, J. G., and E. A. MacIsaac. 1996. British Columbia Lake enrichment program: Two decades of habitat enhancement for sockeye salmon. Regulated Rivers: Research and Management. 12:547-561.

Stockner, J. G., and K. S. Shortreed. 1985. Whole-lake fertilization experiments in coastal British Columbia: Empirical relationships between nutrient inputs and phytoplankton biomass and production. Canadian Journal of Fisheries and Aquatic Sciences. 42:649-658.

Stockner, J. G., and K. S. Shortreed. 1988. Response of Anabaena and Synechococcus to manipulation of nitrogen:phosphorus ratios in a lake fertilization experiment. Limnology and Oceanography, 33(6): 1348-1361.

Taki, D., B. Lewis, and R. G. Griswold. 1999. Shoshone-Bannock Tribes, Snake River Sockeye Salmon Habitat and Limnological Research, 1999, Annual Report 1997 to Bonneville Power Administration, Portland, Or, Contract No. 91BI22548; Project No. 91-071-00, 92 electronic pages (BPA Report DOE/BP-22548-6)

Taki, D., and A. Mikkelsen. 1997. Snake River sockeye salmon habitat and limnological research. Shoshone-Bannock Tribes Annual Report to United States Department of Energy, Bonneville Power Administration, Environment, Fish and Wildlife. Portland, Oregon.

Teuscher, D., and D. Taki. 1996. Snake River Sockeye Salmon habitat and limnological research. Annual Report 1995. U.S. Department of Energy, Bonneville Power Administration, Portland, OR.

Teuscher, D., and five co-authors. 1995. Snake River Sockeye Salmon habitat and limnological research. Annual Report 1994. U.S. Department of Energy, Bonneville Power Administration, Portland, OR.

Teuscher, D., and six co-authors. 1994. Snake River Sockeye Salmon habitat and limnological research. Annual Report 1993. U.S. Department of Energy, Bonneville Power Administration, Portland, OR.

Vollenweider, R. A. 1965. Calculation models of photosynthesis depth curves and some implications regarding day rate estimates in primary productivity measurements. Pages 425-457 in C.R. Goldman, editor. Primary Productivity in Aquatic Environments. 1st Italiano Hydrobiologia Memoirs, Supplement 18, University of California Press, Berkeley, CA.

Waples, R. S., O. W. Johnson, and R. P. Jones, Jr. 1991. Status review for Snake River sockeye salmon. U.S. Department of Commerce, National Oceanic and Atmospheric Administration. Technical Memo. NMFS F/NWC-195.

Wetzel, R. G., and G. E. Likens. 1991. Limnological Analyses, 2nd edition. Springer-Verlag. New York.

Wierenga, R. E., S. T. Juul, and W. Funk. 1999. Primary productivity in the Alpine Lakes of the Sawtooth Mountains, Idaho. Report of the State of Washington Water Research Center, Washington State University, Pullman, Washington to Biolines Environmental Consulting, Stanley, Idaho.

Wurtsbaugh, W. A., and P. Budy. 1997. Spatial – temporal analysis of primary productivity in the mountain lake habitat of endangered sockeye salmon. 1996 Annual Report to Biolines Inc. Department of Fisheries and Wildlife / Ecology Center, Utah State University, Logan, Utah.