Exxon Valdez Oil Spill
Restoration Project Annual Report

Restoration of Coghill Lake Sockeye Salmon: 1995 Annual Report on Nutrient Enrichment

Restoration Project 95259
Annual Report

This annual report has been prepared for peer review as part of the *Exxon Valdez* Oil Spill Trustee Council restoration program for the purpose of assessing project progress. Peer review comments have not been addressed in this annual report.

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Study History: Following the 1989 *Exxon Valdez* oil spill off Bligh Reef in Prince William Sound (PWS), Coghill Lake was selected as a system for sockeye salmon restoration to replace fishery stocks damaged by the oil spill. Edmundson et al. (1992) suggested that the decline in returns of Coghill Lake sockeye salmon was caused by a decrease in macrozooplankton from excessive foraging by high densities of rearing fry produced from large escapements. In 1993, a 5-year nutrient enrichment project (restoration project 93024) was implemented to increase lake productivity. Nutrient enrichment and assessment studies continued in 1994 (Restoration Project 94259) and in 1995 (Restoration Project 95259).

Abstract: Prior to its recent decline, Coghill Lake was an important sockeye salmon (*Oncorhynchus nerka*) producer in western Prince William Sound. It has been suggested that consecutive years of high escapements adversely impacted the forage base (zooplankton) and reduced the lake's rearing capacity. During the first three years of nutrient enrichment of Coghill Lake, the seasonal mean phosphorus concentration increased 22%, seasonal mean algal biomass (chlorophyll *a*) increased 250%, and a greater biomass of cladocera (*Bosmina*) zooplankters were present (and utilized by rearing fry) in the fall. Increases in secondary production contributed to increased smolt production in 1994 and 1995 (average of 1.4 million) compared to before nutrient enrichment (average of 275,000). While productivity has increased during nutrient enrichment, restoring the run is contingent upon obtaining adequate fry recruitment and continued improvement of the zooplankton forage base. In 1995, 30,382 sockeye salmon returned to the lake, partially due to recently adopted management strategies to reduce commercial fishery interceptions of these salmon. The major project goal is to balance in-lake fry densities via juvenile recruitment from escapement and/or hatchery fry releases with the existing forage base, which should expand with continued nutrient enrichment to achieve restoration of Coghill Lake sockeye salmon.

Key Words: Glacial, meromictic, nutrient enrichment, *Oncorhynchus nerka*, Prince William Sound, sockeye salmon restoration, zooplankton.

Project Data: (will be addressed in the final report)

Citation:

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Introduction

Overview

Historically, Coghill Lake produced the largest sockeye salmon returns within the Prince William Sound (PWS) commercial fishery. The highest return of 1.1 million sockeye salmon occurred in 1982; however, since 1990 the adult return (excluding hatchery fish) averaged less than 33,000 (Table 1). In addition, in the last five years the return per spawner for Coghill Lake sockeye salmon has been less than 1 (Table 2). Several hypotheses have been proposed as causes for this decline in sockeye salmon production. Edmundson et al. (1992) suggested that the decline was caused by a decrease in macrozooplankton from excessive foraging by rearing fry. During 1980-1982 escapements averaged 160,000, which were nearly three times the 30-year mean escapement (65,000). Progeny from these escapements are believed to have severely reduced the zooplankton forage base, which adversely affected the lake's rearing capacity. Such top-down control by planktivorous fish and subsequent changes in fish production have been documented in other lakes (Carpenter et al. 1985; Kyle et al. 1988; Schmidt et al. 1994; Koenings and Kyle 1996; Kyle 1996).

Willette et al. (1995) suggested that the *Exxon Valdez* oil spill (EVOS) in 1989 may have exacerbated the decline in sockeye salmon production because juveniles migrated through oil-contaminated habitat in western PWS. However, there is no documented evidence that this occurred. It has also been speculated that climatic effects (e.g., temperature) on freshwater and/or marine survival may have contributed to the rapid decline of the Coghill Lake sockeye salmon run. While we recognize the importance of broad-scale temperature or energy variables to lake productivity (Brylinsky and Mann 1973) and salmonid production (Plante and Downing 1993), there are insufficient data to suggest that climatic effects are a primary cause of the observed rapid decrease in sockeye salmon production. In addition, the interception of sockeye salmon in the commercial harvest of adults returning to the Main Bay Hatchery in PWS has been proposed as a mechanism for this decline, but overall catches do not indicate this as a significant factor. While the specific cause(s) for the decline in Coghill Lake sockeye salmon remain unknown, pre-enrichment limnological sampling revealed that this lake is a nutrient-limited system that supports a relatively low standing stock (biomass) of macrozooplankton (Edmundson et al. 1992).

The plan to restore Coghill Lake sockeye salmon relies on nutrient enrichment to increase productivity, and attaining adequate numbers of rearing fry. Specifically, the restoration plan is to expand the forage base (zooplankton biomass) and to attain adequate fry recruitment (commensurate with the forage base) by achieving the escapement goal of 25,000 through changes in management techniques (Donaldson 1994; PWSAC 1995) or by hatchery stocking if the escapement goal is not reached for two consecutive years. Nutrient enrichment is a proven technique to increase a lake's capacity to produce forage (zooplankton) for rearing sockeye salmon, which can result in greater smolt biomass and higher adult returns (LeBrasseur et al. 1978; Stockner and Hyatt 1984; Stockner and Shortreed 1985; Kyle 1994; Kyle et al. 1996).

In considering the nutrient enrichment program for Coghill Lake, concern initially arose regarding the meromictic nature (monimolimnion) of this system and its effect on nutrient recycling. That is, the trapping of metabolites derived from the decomposition of organic matter within the monimolimnion, and the lack of re-circulating nutrients into the trophogenic zone (Hutchinson 1957; Wetzel 1975). In addition, because Coghill Lake is glacially influenced, the presence of elevated turbidity derived from glacier melt, could counteract primary production through decreased light penetration and reduce the euphotic volume (Koenings et al. 1986; Grobbelaar 1989; Lind et al. 1994). These concerns are warranted because both phenomena affect lake productivity. However, two other meromictic lakes in Alaska have been treated with nutrients and results were positive. In Redoubt Lake (McCoy 1977) located near Sitka, increases in primary and secondary production resulted in larger smolts and higher adult returns (Kyle et al. 1996). Similarly, increased primary production and zooplankton biomass was observed in treated Hugh-Smith Lake located near Ketchikan (Peltz and Koenings 1989). Nutrient enrichment also increased productivity in glacially-turbid Owikeno and Kitlope lakes in British Columbia (Stockner 1987).

In 1993, the EVOS Trustee Council approved the restoration plan (nutrient enrichment) for Coghill Lake to supplant fishery resources damaged by EVOS. The Alaska Department of Fish and Game (ADF&G), in cooperation with the U.S. Forest Service (USFS), initiated a 5-year nutrient enrichment project in 1993. During the first year (1993) of nutrient treatment, the mean total phosphorus concentration increased by 13% and the mean algal biomass (chlorophyll *a*) concentration increased 3-fold compared to the pre-enrichment means (Willette et al. 1995). In addition, macrozooplankton (*Cyclops* and *Bosmina*) densities and biomass more than doubled compared to the previous four years. As a result, nutrient enrichment continued in 1994 and similar results were achieved (Edmundson et al. 1995). This report provides results of the third year (1995) of nutrient treatment and discusses the effects of the enrichment program to date on lake productivity and juvenile sockeye salmon production.

Objectives

The primary objectives of the Coghill Lake restoration project are: 1) apply liquid fertilizer to the lake to increase nutrient concentrations, primary production, and zooplankton biomass; 2) determine the responses of primary and secondary production to nutrient enrichment; 3) assess the abundance, diet, condition, and size of rearing sockeye salmon fry; and 4) determine the abundance, age, size, and condition of sockeye salmon smolts emigrating from the lake. In addition, we assess adult escapements and the escapement goal relative to the current forage base and historical return-per-spawner data.

Description of Study Area

Coghill Lake (61° 4' N, 147° 54'W) is located ~130 km northwest of Cordova in PWS at an elevation of 18 m (Figure 1). This lake has a surface area of 12.7 km^2, a mean depth of 46.3 m, and a total volume of 587 x 10^6 m^3 (Figure 2). Coghill Lake is meromictic with a dense (saline) and permanently stagnant water mass referred to as a monimolimnion (Hutchinson 1957; Wetzel 1975). This layer extends from a depth of about 30 m to the bottom of Coghill Lake, and comprises 45% of the total lake volume (Edmundson et al. 1992). In contrast, the lake mixes freely from the surface to a depth of ~20 m. This layer or mixolimnion is separated from the monimolimnion by a steep concentration gradient or chemocline (20-30 m). Coghill Lake has provided the majority of sockeye salmon production in Prince William Sound. Sockeye salmon fingerlings were stocked in this lake for the first time in the fall of 1994 (300,000). In the fall of 1995, 900,000 fingerlings were stocked. Previous to 1994 sockeye salmon smolts were stocked in the estuarine area near the mouth of Coghill River in an attempt to imprint these fish to Coghill Lake. This lake has few competitors or predators relative to sockeye salmon as determined by net sampling during hydroacoustic surveys and standard fish index surveys.

METHODS

Fertilizer Application

Supplemental nutrient (phosphorus) loading rates for Coghill Lake were estimated using models developed by Vollenweider (1976). The annual supplemental loading rate of phosphorus for 1995 was estimated at 1,412 kg. As a result, a total of 64,650 kg of pharmaceutical-grade liquid fertilizer containing 20% nitrogen and 5% phosphorus (20-5-0) was applied to the lake (Table 3). In addition, 9,350 kg of 32-0-0 fertilizer was applied (Table 3) to correct for a summer depletion of inorganic nitrogen and to maintain the total nitrogen to phosphorus within the 20:1 to 30:1 ratio. Application occurred in the middle third (5.5 km^2) of the lake (Figure 2). The fertilizer was dispensed via a fixed-wing aircraft on a weekly basis (a 2-3 day period) and consisted of 6 to 9 passes over the fertilization zone each day of application. Fertilizer was applied from 01 July to 19 August. A summary of fertilizer application amounts and dates for 1993 and 1994 is provided in Appendix A.

Limnological Sampling

During 1995, comprehensive limnological surveys were conducted eight times: 28 June, 19 July, 02 August, 17 August, 06 September, 26 September, 11 October, and 02 November. The same three limnetic sampling stations used in the past were used in 1995. Stations A and B represented the same survey sites used during 1986-1993; and Station D, established in 1993 between stations A and B, is in the middle of the fertilizer application zone (Figure 2). For each survey, light penetration (foot candles) was measured with an International Light photometer at 1-m increments from the surface to a depth of 27 m or to the light extinction. The vertical extinction coefficient (K_d), or the amount of light retained per meter, was equivalent to the slope derived via a 0-intercept linear regression for each set of photometer readings using the equation:

(1) $$\ln(I_0/I_z) = K_d Z$$

where I_Z = light intensity (foot candles) at depth, I_0 = light at subsurface, K_d = extinction coefficient, and Z = depth. The euphotic zone depth (EZD), the depth at which 1% of the subsurface light (400-700 nm) penetrates, was calculated by substituting ln(100) into equation 1 and rewriting as:

(2) $$EZD = \ln(100)/K_d \text{ or } 4.6052/K_d.$$

Water clarity was measured with a standard 20-cm Secchi disk. Temperature and dissolved oxygen concentration were measured at the surface and at 2-m increments to a depth of 10 m, and then at 5-m increments from 10 to 60 m.

Water Sampling

Water samples were collected from 1 and 20 m at the three stations (Figure 2). For each depth sampled, ~8 L of water were collected using a non-metallic Van Dorn sampler and placed in pre-cleaned polyethylene carboys. The samples were then transported to Cordova and filtered/preserved for laboratory analysis. Each sample was partitioned into three types: 1) refrigerated for general water-quality testing; 2) frozen for analysis of total phosphorus and total Kjeldahl nitrogen; and 3) filtered through a Whatman GFF glass-fiber filter and frozen for analysis of dissolved nutrients. Samples were stored in acid-cleaned polyethylene bottles and shipped to the limnology laboratory in Soldotna for analysis.

In the laboratory, conductivity (compensated to 25° C) was measured using a YSI conductance meter, and pH (@ 25° C) was measured with a Corning pH/ion meter. Alkalinity was determined by acid (0.02 N H_2SO_4) titration to pH 4.5 (AHAP 1985). Turbidity, expressed as nephelometric turbidity units (NTU) was measured with a HF DRT1000 turbidimeter, and color was determined on a filtered sample by measuring the spectro-photometric absorbance at 400 nm and converting to platinum-cobalt (Pt) units (Koenings et al. 1987). Calcium and magnesium were determined from separate EDTA (0.01N) titrations after Golterman (1969), and total iron was analyzed by reduction of ferric iron with hydroxylamine during hydrochloric acid digestion as described by Strickland and Parsons (1972). Filterable reactive phosphorus (FRP) was analyzed by the molybdenum-blue/ascorbic acid reduction of Murphy and Riley (1962) as modified by Eisenreich et al. (1975). Total phosphorus was determined using the FRP procedure after acid-persulfate digestion. Nitrate + nitrite was analyzed as nitrite following cadmium reduction of nitrate, and total ammonia was determined using the phenylhypolchlorite methodology of Stainton et al. (1977). Total Kjeldahl nitrogen (TKN) was determined as total ammonia following block digestion (Crowther et al. 1980). Finally, reactive silicon was determined using the method of ascorbic acid reduction to molybdenum blue after Stainton et al. (1977).

Phytoplankton and Zooplankton Sampling

Water samples collected from 1 and 20 m at the three stations were also used for algal biomass (chlorophyll *a*) analysis. These samples were prepared by filtering 0.5-1.0 L of lake water through a Whatman GFF glass-fiber filter, to which 1-2 ml of saturated magnesium carbonate solution were added just prior to completion of the filtration. The filters were stored frozen in individual plexislides for later analysis. Pigment was extracted after homogenizing glass-fiber filters in 90% acetone using a tissue grinder and pestle. Chlorophyll *a* concentrations (corrected for phaeophytin) were determined using the fluorometric procedure of Strickland and Parsons (1972). The low strength acid addition recommended by Reimann (1978) was used to estimate phaeophytin. Water samples for estimates of phytoplankton cell density and taxa from the 1-m depth strata were collected in 1995 from stations A and B (for comparison to previous years), preserved in a Lugols acetate solution (Koenings et al. 1987), and analyzed by Eco-Logic Ltd. of British Columbia, Canada.

Vertical zooplankton tows were taken from the 25-m depth to the surface at the three sampling stations using a 0.2-m diameter, 153-μm mesh, conical net. The net was pulled manually at ~0.5 m sec^{-1}. The contents were rinsed into a polybottle and preserved with buffered formaldehyde to a final 10% formalin-sample solution. Cladocerans and copepods were identified using taxonomic keys by Brooks (1957), Pennak (1978), Wilson (1959), and Yeatman (1959). Enumeration consisted of counting the animals in triplicate 1-ml subsamples taken with a Hansen-Stempel pipet in a 1-ml Sedgewick-rafter cell. Zooplankton body length was measured to the nearest 0.01 mm for at least 10 individuals along a transect in each of the 1-ml subsamples (Koenings et al. 1987). Zooplankton biomass was estimated from species-specific regression equations derived between zooplankter body length and weight (Koenings et al. 1987).

Fry Sampling

Hydroacoustic surveys conducted in August and September of 1994 to estimate juvenile sockeye salmon abundance and distribution are presented in this report as the data were not processed in time for the 1994 report. These surveys were conducted and analyzed by the Prince William Sound Science Center under a State of Alaska contract (Number 94-009). Methods for the 1994 surveys are provided in the contract report (Thomas and Kirsch 1995). In 1995, hydroacoustic surveys were done on 14 August, 14 September, and 10 October by ADF&G staff using sampling procedures described in Kyle (1990). A 420-kHz BioSonics dual-beam echosounder (model 105) with a model-171 tape recorder interface and a 6/15° transducer was used. Fish signals were recorded electronically with a Sony digital audio tape recorder (model TCD-D10) and on chart paper using a BioSonics model-

115 chart recorder. In 1995, fish signals (downlooking data) were recorded along 12 transects (Figure 3). In addition, side-scanning data were recorded along diagonal transects (Figure 3) to obtain a mean fish density near the surface of the lake (0-2 m) which is not possible in the down-looking mode.

The recorded hydroacoustic data were processed by Dr. Richard Thorne of BioSonics, Inc. under a State of Alaska contract. For the down-looking data, a lake abundance estimate for each transect ($\hat{N}_i$) was calculated as the product of the transect density (no. m^{-2}) and the lake surface area. The total fish abundance estimate ($\hat{N}$) was then calculated as the mean of the individual transect estimates. The estimated variance of $\hat{N}$ was calculated as the sample variance divided by the number of transects (n = 12), and the standard error (S. E.) was calculated as the square root of the variance. Confidence intervals (95%) are reported as $\hat{N} \pm 2.2(S.E.)$, where 2.2 is the t-value for $p = 0.05$ with 11 degrees of freedom. The side-looking data (near surface density) was estimated as the mean density of the transects, weighted by the number of minutes towed in each transect. Fish abundance was then calculated as the product of the mean density and the surface area of the lake. Because the sampling was non-random (Figure 3), we did not compute a variance estimate or confidence interval for the near-surface population.

Juvenile sockeye salmon fry were collected during 15-16 August, 15-16 September, and 11 October for stomach content analysis, food electivity indices, size, and condition factor. Due to a difficulty in obtaining the desired number of samples a different net was used for each sampling trip. In August a 7.5-m long mid-water trawl with a 2 x 2 m opening was deployed; in September a 5-m long monofilament mid-water trawl with a 1.5 x 1.5 m opening was used; and during the October trip a 12-m long open-water trawl with a 3 x 2 m opening was towed using two boats instead of one as was the case in August and September. Tows were conducted in areas of the lake where hydroacoustic data indicated a higher density of fish.

All fish retained for analysis were preserved in 10% buffered formalin for at least six weeks to ensure stabilization of the sample. Analyses of each sample consisted of a fork length measurement (tip of snout to fork of the caudal fin) to the nearest millimeter and weighing to the nearest 0.1 g. A scale smear was taken from each fish, spread and separated on a glass slide, and aged using a microfiche projector. Stomachs were eviscerated and the contents were enumerated and identified to the lowest possible taxonomic level. Zooplankton were measured to the nearest 0.01 mm and prey (zooplankton) body weight was estimated from empirical regression formulae between zooplankter body length and dry weight (Koenings et al. 1987). Insects or insect body parts were placed on a glass slide, dried, and then weighed separately.

Smolt Sampling

Three inclined-plane traps (Todd 1994) were used to estimate the population of sockeye salmon smolts emigrating Coghill Lake and to collect smolts for size and age information

(Figure 4). The traps sampled approximately 20% of the river surface area at the sample site. Traps 1 and 2 were installed on 26 April and trap 3 was installed on 08 May. The three traps were fished until 19 June, and were not fished on 26-27 May due to heavy debris and ice floating down river from the lake. Trap efficiencies were estimated from a mark-recapture technique (Rawson 1984). A total of eight mark-recapture experiments were conducted between 13 May and 12 June. Bismark brown dye was used to mark (dye) the smolt. All smolts captured were examined for dye coloration. Results of trap efficiency trials were applied to their respective time periods to estimate weekly abundance. Population estimates were also adjusted for smolts found dead in the traps. Methods described by Rawson (1984) were used to estimate the smolt population and variance. During the trapping operation all smolts were individually enumerated, and 40 per day were sampled for age, weight, and length (AWL) information. Stratified random sampling methods (Cochran 1977; Scheaffer 1986) were used to estimate age-class proportions and their respective variances; daily strata were established and stratum weights were based on smolt abundance estimates. Mean weight and length of each age-class and associated variances were estimated using stratified sampling methods for subpopulations (Cochran 1977).

Adult Escapement and Harvest

Adult sockeye salmon returning to Coghill Lake were enumerated with the use of a weir from 11 June through 02 August by the ADF&G Commercial Fisheries Management and Development (CFMD) Division as part of its management of this run. Returning adults were enumerated and a portion were sampled for age, size, and sex following procedures described by Crawford and Simpson (1991). When sockeye salmon were passed over the weir they were scanned for missing adipose fins to estimate the hatchery contribution to the escapement. Based on an expected low return to Coghill Lake, sockeye salmon with missing adipose fins were recorded but the heads were not removed to detect tags and codes.

The commercial harvest of Coghill sockeye salmon takes place in several distinct areas in PWS including the Eshamy and Coghill Districts. However, the entire Coghill District was last open to commercial fishing in 1989. Since then, the commercial gillnet fleet has been restricted to fishing in either the Esther Subdistrict, or in the hatchery terminal harvest area in an effort to reduce the harvest of Coghill Lake sockeye salmon by providing a migration corridor. Similar restrictive measures have been incorporated in the Eshamy District.

During 1995, commercial fishing began in the Esther Subdistrict on June 15 for 24 h to harvest Wally Noerenberg Hatchery chum salmon. The second 24-h period on June 19 included all waters of the Esther Subdistrict. Beginning on June 23, only the waters of the Esther Subdistrict within one mile of Esther Rock were open to commercial fishing for 12 h. The next fishing period on June 29 once again included only the waters within one mile of Esther Island for 12 h and an additional 12 h in the Wally Noerenberg Hatchery terminal harvest area. Following the June 29 fishing period, only this terminal area was open to commercial fishing from 0800 to 2000 h on Monday and Friday of each week through 17 July. In addition to the above restrictions, commercial fishing in the Eshamy District was restricted to only the Main Bay Subdistrict.

Age class and run timing differences between Coghill and Eshamy stocks were to distinguish commercial harvest reported on fish tickets. While it is generally known that some Coghill Lake sockeye salmon are intercepted in fisheries in the Southwest, Eshamy, and Unakwik Districts (Merritt and Donaldson 1995), most of the catch in the Coghill District is assumed to be primarily Coghill Lake sockeye salmon. No estimates were made of the harvest of Coghill Lake sockeye salmon outside of the Coghill District.

Data Analysis

The effect of nutrient enrichment on general water chemistry, nutrient concentration, chlorophyll *a* (algal biomass), and zooplankton production were assessed using analysis of variance (ANOVA) to test for differences between pre-enrichment (PE) and enrichment (E) years. These analyses were accomplished using a randomized complete block design to test for overall year effect, followed by a contrast statement to compare the PE and E years. We also tested for spatial (station) differences. Because station D was not established until 1993, only stations A and B were included in these analyses. All tests were conducted at the $\alpha = 0.05$ significance level. Statistical analysis was facilitated using SYSTAT version 5 (Wilkinson 1990) or SAS/STAT version 6 (SAS Institute 1990).

The effect of nutrient enrichment on sockeye salmon fry was evaluated by testing for changes in condition factor between months and years. To assess changes in fry condition, a relative weight (W) index was computed based on the length-weight regression model:

(3) $$W = aL^b e^{\varepsilon}$$

which can be written in linear form as: $\ln(W) = \ln(a) + b \ln \text{length } (L) + \varepsilon$, where $\ln(a)$ is the intercept and b is the slope. Regression analysis was used to estimate the relationship between $\ln(W)$ and $\ln(L)$. Then, the condition factor (K_i) of individual fish was estimated by:

(4) $$K_i = \frac{w_i}{\hat{w}_i}$$

where w_i is the weight of an individual fish and $\hat{w}_i$ is the predicted weight based on the fitted length-weight regression equation (LeCren 1951). Seasonal and changes in the condition index were then evaluated using 1-way ANOVA followed by pairwise comparisons.

A second approach to assess change in fry condition was based on a separate-slopes analysis of covariance (ANCOVA) model to compare length-adjusted weights between groups of fry (ln-ln transformed data). With this approach, mean weight was compared after removing the effect of length, thus providing an alternative evaluation of condition. The electivity index (Ivlev 1961) was calculated to determine the active selection of prey items by rearing sockeye fry. This index has a range of -1 to +1; negative values indicate either avoidance or inaccessibility of a prey item, zero indicates random selection, and positive values indicate preference. The electivity index (E_i) for prey species i was estimated as:

(5)
$$E_i = \frac{r_i - p_i}{r_i + p_i}$$

where r_i is the relative abundance of prey item i in the stomach of the predator expressed as a proportion or percentage of the total stomach contents, and p_i is the relative abundance of the same prey item in the lake expressed as a proportion or percentage of the total density.

Analysis of spawner-return data for estimating escapement that produces maximum sustained yield (MSY) followed Ricker's (1975) model as described in Hilborn and Walters (1992). Two approaches, based on the dichotomous nature of the escapement data were compared. First, all historical data (brood years 1962 to 1989) were analyzed. Second, only brood years with escapements of less than 100,000 were analyzed, which excluded brood years 1980 to 1982, 1985, and 1987. The latter approach eliminated the influence of unusually large escapements. Residuals from both analyses were tested for first order autocorrelation using the Durbin-Watson statistic. We also applied a tabular method, similar to a Markov transition matrix, to determine the escapement range or mean escapement that maximizes mean yield (brood-year returns minus escapement). Escapement intervals of 20,000 fish were used; successive intervals were allowed to overlap at the midpoint to increase the resolution of the analysis.

RESULTS

Physical Features

Turbidity from glacier meltwater is a major factor controlling light penetration in Coghill Lake. Considering all survey dates during 1988-1995, mean turbidity within the 1-m stratum ranged from <1 to 25 NTU (Appendix B). There was no significant difference in turbidity between sampling stations A and B; and although turbidity during the enrichment period (1993-1995) was higher than during pre-enrichment, it was not significantly higher ($p = 0.113$) (Table 4). During 1995, turbidity within the 1-m stratum averaged 7.2 NTU (Figure 5A) which was somewhat higher than the mean of 5 NTU for all years (Figure 5B) and represented about a 2.6-fold increase compared to 1994 (2.8 NTU). As a result, the euphotic zone depth (EZD) in 1995 averaged 8.1 m which was

40% shallower than in 1994 (13.8 m) and was consistent with the higher turbidity levels. We found a significant ($p < 0.001$; $r^2 = 0.70$) inverse relationship between turbidity and euphotic zone depth (EZD) for Coghill Lake (Figure 6A). Between 1 and 5 NTU, EZD decreases rapidly from about 10 to 5 m, but the negative response lessens at higher turbidity levels. The regression equation (log-log transformed data) was: log EZD = 1.189 - 0.573(log turbidity). In addition, the light attenuation coefficient (K_d), which is the proportion of light retained per meter and is directly related to turbidity, ranged from 0.2 to 2.1 m^{-1} (Figure 6B). Since K_d and EZD are analogous indices of light penetration, K_d also exhibits a curvilinear response to increasing turbidity. After log transformation of the data, the regression equation was: $\log K_d = -0.516 + 0.573(\log \text{turbidity})$. Organic color also acts to attenuate light, but in Coghill Lake there is very little color (mean 6 Pt units), and although color was significantly lower ($p < 0.001$) during enrichment (5.1 Pt units) compared to pre-enrichment (7.2 Pt units) (Table 4), the difference is well within the variation observed in non-colored lakes (Koenings and Edmundson 1991).

Over the past 8 years in Coghill Lake, average temperatures of the 1-m stratum ranged from 8.5° C in 1988 to 11.6° C in 1994 (Table 5). Maximum summer temperatures ranged from 12.4° C in 1988 to 17.2° C in 1993, and generally occurred from late July to early August. Despite the inter-annual variation in mean and maximum near-surface temperatures, the average temperature (T_s) of the water column (0- 60 m) varied little and ranged from 6.0 to 6.8° C . Total heat content (product of water volume and temperature) of the mixolimnion (0-25 m) in early August ranged from 1,800 x 10^6 to 2,176 x 10^6 calories. Heat content of the monimolimnion (25-60 m) ranged from 1,713 x 10^6 to 1,811 x10^6 calories. On an areal basis, the total heat content of the water column ranged from 27,717 to 31,462 calories cm^{-2}. Total caloric content of the mixolimnion (HC) and areal heat content (AHC) were linearly related to maximum epilimnetic (1-m) temperatures (T_{max}), but not to seasonal mean temperatures of the 1-m stratum (T_1). The regression equations were: $HC = 683.8 + 87.752T_{max}$ ($r^2 = 0.76$; $p = 0.005$) and $AHC = 19{,}025 + 721.7T_{max}$ ($r^2 = 0.65$; $p = 0.015$).

Water Chemistry

A summary of 1995 water chemistry parameter concentrations by date from the 1-m and 20-m strata of the mixolimnion are presented in Appendices B and C, and the means for all years are presented in Appendices D and E. On a temporal basis, the pH was lower ($p < 0.001$) during enrichment compared to pre-enrichment (Table 4); however, the 0.2 pH unit difference is probably not biologically significant. No other general water chemistry parameters changed significantly during enrichment.

For the nutrients, total phosphorus (TP), total Kjeldahl nitrogen (TKN), nitrate-nitrite, and reactive silicon there were significant ($p < 0.05$) differences in mean concentration before and during nutrient enrichment (Table 4). Before enrichment the seasonal TP concentration averaged 8 $\mu g\ L^{-1}$ and during enrichment it averaged 10.3 $\mu g\ L^{-1}$ for a 22% increase. In addition, there was a significant difference ($p = 0.002$) between stations A and B. TKN also differed significantly between stations (67 and 57 $\mu g\ L^{-1}$) and nearly doubled in seasonal mean concentration during enrichment (from 47 to 93 $\mu g\ L^{-1}$). For nitrate-nitrite, there was no significant difference between stations but the mean concentration significantly changed from 18.8 $\mu g\ L^{-1}$ before enrichment to 13.7 $\mu g\ L^{-1}$ during enrichment. The mean reactive silicon concentration during enrichment (mean 511 $\mu g\ L^{-1}$) was significantly ($p < 0.001$) less than pre-enrichment (mean 726 $\mu g\ L^{-1}$). Total filterable phosphorus, filterable reactive phosphorus, and ammonia remained very low during all years sampled, and did not significantly change during enrichment.

Since application of the fertilizer took place primarily during July and August, we questioned whether the overall difference (ANOVA) in yearly mean values adequately describe fertilization effects. That is, the difference in means may hide the fact that nutrient concentrations were higher (or lower) during summer when the fertilizer was applied. Thus, we examined the season by year interaction among spring, summer, and fall mean values for the water chemistry and nutrient variables. Figure 7 displays the temporal trends by season for those variables which exhibited a significant ($p < 0.05$) interaction.

Spring and summer turbidity levels were low (< 5 NTU) and relatively consistent during pre-enrichment and enrichment. However, turbidity increased markedly in the fall, but elevated levels (> 5 NTU) occurred during both the pre-enrichment and enrichment periods. Interaction was also significant ($p < 0.001$) for conductivity, but this plot is difficult to interpret as there is no apparent temporal pattern or trend, and thus appears unrelated to fertilization. Spring TP levels were generally low (< 5 $\mu g\ L^{-1}$) and consistent throughout the eight years sampled. In contrast, summer TP concentration increased nearly 2-fold during enrichment (mean 11.2 $\mu g\ L^{-1}$) compared to the pre-enrichment period (mean 5.8 $\mu g\ L^{-1}$). Fall TP concentrations did not increase significantly and followed the same trend as that for turbidity; concentrations averaged 10.7 $\mu g\ L^{-1}$ and 11.8 $\mu g\ L^{-1}$ respectively, before and after enrichment. Summer TKN concentrations (mean 117 $\mu g\ L^{-1}$) were more than twice that observed prior to nutrient enrichment (mean 47 $\mu g\ L^{-1}$). Moreover, spring and fall TKN levels increased by more than 50% during enrichment. In contrast to TP and TKN, reactive silicon levels in the summer decreased from an average of 608 $\mu g\ L^{-1}$ before enrichment to 258 $\mu g\ L^{-1}$ in 1995. Both spring and fall silicon concentrations were variable and did not exhibit any temporal trend.

Phytoplankton

During 1995, the chlorophyll *a* concentration (algal biomass) within the 1-m stratum ranged from 0.69 $\mu g\ L^{-1}$ in late June to 7.50 $\mu g\ L^{-1}$ in mid-August, and averaged 2.81 $\mu g\ L^{-1}$ (Figure 8A). In 1995, high concentrations persisted throughout August and into early September, and peak concentrations in 1995 were the highest found during the eight years sampled (Figure 8B). The 1995 seasonal mean concentration was more than double the 8-year mean and 60% higher than in 1994. No significant spatial or station differences in chlorophyll *a* concentration were found (Table 4); however, the mean concentration during the three years of enrichment (2.24 $\mu g\ L^{-1}$) was 250% higher compared to the pre-enrichment mean (0.64 $\mu g\ L^{-1}$). The mean chlorophyll *a* concentration in the summer (Figure 7) was five times higher during enrichment (mean 3.30 $\mu g\ L^{-1}$) compared to pre-enrichment (mean 0.66 $\mu g\ L^{-1}$). The concentration in the fall increased by more than 30% following enrichment (mean 1.44 $\mu g\ L^{-1}$), but spring concentrations remained at < 0.5 $\mu g\ L^{-1}$.

The density of diatoms in Coghill Lake increased dramatically during enrichment (Figure 9). In all pre-enrichment years except for 1989, diatom densities were less than 500 m^{-3}, but during enrichment (1993-1995) densities exceeded about 2,000 m^{-3}, and reached as high as 13,000 m^{-3}. For the chryso-cryptophyte group, the increase in density was not as dramatic; however, overall on a relative basis densities were consistently higher during the years of enrichment (Figure 10). The higher densities of these two groups of phytoplankton also translated into higher densities of edible (useable by zooplankton for food) forms. For example, the density of edible diatoms during pre-enrichment (except for the July 1989 sample) were less than 200 m^{-3}, whereas during enrichment the majority of samples exceeded 1,000 m^{-3} (Figure 11). For the chryso-cryptophytes the edible forms were consistently high on a relative basis in the summer (July-August) during enrichment (Figure 12).

Zooplankton

A summary of zooplankton mean density, biomass, and size by station for each year sampled is presented in Appendix F. The zooplankton community in Coghill Lake is comprised primarily of *Cyclops* and *Bosmina.* In addition to *Bosmina*, two other cladocerans, *Daphnia* and *Chydorinae* occur sporadically in very low densities. In 1995, *Cyclops* densities increased from less than 5,000 m^{-2} in June to a maximum of 35,000 m^{-2} in mid August (Figure 13A), and averaged 20,800 m^{-2} over the season. After the peak in mid-August, *Cyclops* densities began to decrease but leveled off in late September and mid October at about 15,000 m^{-2}. Similar to *Cyclops*, *Bosmina* densities peaked in mid August, but unlike *Cyclops* high densities did not continue after the September 26 sample date (Figure 13A). *Bosmina* densities peaked at 5,200 m^{-2}, and the average density in 1995 was about 2,700 m^{-2} over the sample season. The densities of *Cyclops* and *Bosmina* in 1995 compared to other years (for stations A and B) are presented in Figure 13B. The density of *Cyclops* was similar to 1994 and about the same compared to the average since 1991. In contrast, the density of *Bosmina* in 1995 (as in 1994) was higher compared to previous years. The trend in biomass for these two taxa in 1995 (and in previous years) was identical to that for density (Figure 13C). The 1995 zooplankton biomass averaged 45 mg m^{-2}, which was similar to the 42 mg m^{-2} observed in 1994. Of this total biomass, *Cyclops* comprised 89%, *Bosmina* 7%, and *Daphnia* 4%.

The total season mean zooplankton density in 1995 was 25,790 m^{-2}, and the trend of relatively higher total zooplankton density observed in the fall during 1993 and 1994 continued in 1995 (Table 6). The seasonal mean zooplankton density in 1995 was lower than in 1993 and 1994, but still higher than the average of 19,300 m^{-2} for the two years preceding nutrient enrichment with the most complete (June through October sample period) and comparable (similar times) samples (Table 6). The average for the three years of enrichment was 35,150 m^{-2}. The two time periods in which zooplankton samples were collected consistently (i.e., late August and mid-late October before and during nutrient enrichment) indicated densities were dramatically higher during enrichment. That is, for August the average density during nutrient enrichment (1993-1995) was 1.6-fold higher than the average for pre-enrichment (1989-1990) (60,214 m^{-2} versus 38,389 m^{-2}) and that for October was 2.4-fold higher (22,476 m^{-2} versus 9,355 m^{-2}).

Because of the atypical (suspect) sampling results in 1988 especially in the fall, testing of spatial and treatment effects (ANOVA) was done with and without the 1988 zooplankton data (Table 7). An interaction plot of the means by season indicates that in the fall of 1988 *Cyclops* density was on average eight times greater than other years (Figure 14A). It is possible that this increase could accurately reflect conditions in the lake; however, if productivity was high at this time of year, then an increase in *Bosmina* density should also have occurred. This was not the case, as *Bosmina* density in 1988 was very similar to other years before enrichment (Figure 14B).

Based on ANOVA, seasonal mean densities, body size, and biomass for *Cyclops* and *Bosmina* were not significantly different ($p < 0.05$) between station A and B (Table 7). During the three years of nutrient enrichment (1993-1995), seasonal mean *Cyclops* density was significantly ($p = 0.036$) lower compared to all pre-enrichment years (1988-1992) but not significantly different ($p = 0.131$) for all years excluding 1988 (Table 7). The body size of *Cyclops* before and after enrichment did not significantly change for the years including and excluding the 1988 data, therefore the change in *Cyclops* biomass was identical to the change in density.

In contrast, *Bosmina* density and biomass were significantly ($p < 0.001$) greater after enrichment for both data sets (with and without 1988 data) (Table 7). *Bosmina* biomass exhibited an 8-fold increase during enrichment compared to the pre-enrichment years. Although mean *Bosmina* biomass was significantly greater during enrichment, a standing stock of 2.8 mg m^{-2} is relatively low compared to *Cyclops* in this lake and the total zooplankton biomass in Coghill Lake is low compared to other Alaskan lakes (Kyle 1996).

Fry Abundance, Diet, Condition, and Size

Results of the hydroacoustic surveys conducted by the Prince William Sound Science Center in 1994 (Thomas and Kirsch 1995) revealed a total fish population estimate of 1,090,516 for the August survey conducted with a 120-kHz echosounder. In September two surveys wee done; one with a 200-kHz system and another with a 420-kHz system on two consecutive nights. The total fish population estimates in September were very similar; 7,047,510 for the 200-kHz system compared to 6,973,288 for the 420-kHz system. Survey results indicated that there were high densities of fish near the surface, and extrapolation of the population in the upper 2-8 m strata to the 0-2 meter strata that is not sampled by the gear would increase the August survey by a factor of 1.24, and the

September surveys by 1.19 (200-kHz survey) and 1.21 (420-kHz survey). Townetting indicated that 89% and 82% of the total fish population estimate in August and September, respectively were juvenile sockeye salmon.

A summary of densities and population estimates for the 1995 hydroacoustic surveys based on the down-looking data is presented in Appendix G. The 14 August survey revealed a down-looking population estimate of 53,074. The mean side-looking density in the 0-2 m strata of 22 fish per 1000 m^{-2} extrapolated to the surface area of the lake corresponds to an additional 278,000 fish, for a total population estimate of 331,074. The 95% confidence interval for the down-looking estimate was 34,926, indicating a high variability in the transect estimates (Appendix G1). For the 14 September survey, the population estimate based on the down-looking data was 171,372 ± 63,238 (Appendix G2). The mean side-looking density of 11 fish per 1000 m^{-2} corresponded to an additional 135,000 fish, for a total estimate of 306,372. A population estimate based on the down-looking date for the 10 October survey was 381,169 ± 58,184 (Appendix G3). The side-looking density was 8 fish per 1000 m^{-2} for the October survey, which corresponded to an additional 95,000 fish for a total population estimate of 476,372. Townetting indicated that 80%, 62%, and 98% of the total fish population estimates in August, September, and October, respectively were juvenile sockeye salmon.

A total of 323 sockeye salmon fry were collected in August, September, and October 1995, of which 183 (181 age-0) were analyzed for stomach content analysis (Table 8). In both August and September the dominant prey composition (by biomass) consisted of various insects (insects were not identified as the samples consisted mainly of body fragments). The mean biomass of insect parts in August and September was 640 and 450 μg fry^{-1}, respectively. In October, a diet shift was observed as *Cyclops* (mean of 104 μg), and *Bosmina* (mean of 60 μg) comprised the dominant prey taxa. However, rotifers were also an important food source as 47 μg of rotifers were found per fry stomach in October. Since insects in the fish stomachs could not be enumerated, as parts were often only found, they were not included in the mean abundance calculations. Excluding insects, the dominant prey taxa for August and September were *Bosmina* (mean of 51 and 41 fry^{-1}, respectively) and rotifers (mean of 47 and 30 fry^{-1}, respectively). In October, a mean abundance of 579 rotifers fry^{-1} was observed, and the mean number of *Bosmina* per fry stomach (54) remained consistent with the previous two months.

Prey selection based on Ivlev's (1961) index of electivity (E) indicated a strong preference (by abundance) for both *Bosmina* and ovigerous *Bosmina* (E = +0.83 to +0.99) in all three months (Table 9). In addition, there was a modest preference (E = +0.16 to +.040) for *Cyclops* in the 3 months sampled. Although rotifers comprised 40-87% of the stomach contents of fry, their overwhelming prevalence in the lake (95-98%) revealed a negative selection for these prey (E = -0.06 to -0.42).

Age-0 sockeye salmon fry increased significantly ($p < 0.05$) in size from August to October 1995 (Table 10). The mean age-0 fry size in August was 38.7 mm and 644 mg, in September was 45.3 mm and 1,011 mg, and in October was 50.6 mm and 1,403 mg.

Although fry condition factor was not significantly different among sample months, there was an increase from 0.98 in August to 1.02 in October. Length-adjusted wet weights however, showed a significant ($p < 0.05$) increase from August to September (915 to 976 mg), but the increase from September to October (976 to 999 mg) was not significant. In 1995, the mean weight and length of age-0 fry were the largest since monitoring began (1991) (Figure 15A), and the condition factor improved compared to 1993 and 1994 (Figure 15B).

Smolt Abundance, Age Composition, and Size

A total of eight trap efficiency experiments were conducted in 1995 to determine period and total population estimates of emigrating sockeye salmon smolts (Table 11). Trap efficiency ranged from 3% to 20% and averaged 12%. The peak outmigration occurred during the week of 20-25 May when a total of 74,470 smolts were caught, which represented an estimated 836,946 smolts. A total of 152,145 smolts were caught during 27 April and 19 June, and represented a smolt population estimate of 1,599,992 ± 258,745. The 1995 smolt outmigration estimate was the highest recorded since sampling began in 1989 (Table 12), and included production from 300,000 hatchery fry released in 1994. The mean number of smolts produced during enrichment was 1,435,750 compared to 275,900 before enrichment, which represents a 5-fold increase. Age-1 smolts comprised 96% and age-2 comprised 4% of the total outmigration (Table 12). In 1995, age-1 smolts averaged 60 mm and 1.5 g, whereas age-2 smolts averaged 71 mm and 2.5 g (Table 12). The 1995 age-1 mean smolt size was relatively consistent with the average for previous years, except for 1989 when the age-1 smolts were the smallest ever sampled (52 mm; 1.1 g).

Adult Escapement, Harvest, and Yield

In 1995, the adult sockeye salmon escapement was 30,382 (Table 1) of which 29,832 were estimated to be wild fish (personal communication, Slim Morstad, ADF&G Cordova). The harvest of wild fish in 1995 was an estimated 38,428, and an estimated 40,039 hatchery fish (produced from smolt releases) were harvested (Table 1). The total sockeye salmon run (not including hatchery fish) was 68, 810 in 1995 which compares to an average of 176,728 since 1961.

Results of the Ricker analyses (all years and years with escapements < 100,000) are shown in Figure 16. Neither analysis exhibited first order autocorrelation in the residuals ($p > 0.05$). The analysis using all historical data resulted in an estimated escapement of 62,000 sockeye that would produce a maximum sustained yield (MSY) of 255,000. However, after removing escapements greater than 100,000, an estimated escapement of 24,500 would result in a MSY of 207,000. The difference in these results is primarily due to the influence of brood years 1980-1982, a period in which large escapements produced similar and relatively high return-per-spawner (Figure 16). Removal of the 1980-1982 data from the analysis gave an estimated escapement of 34,000 sockeye that would produce a MSY of 180,000. Results of the tabular assessment are given in Table

13. The highest mean yield of 231,400 was obtained from a mean escapement of 37,600 (30,000 to 50,000 interval). Comparably a high mean yield of 219,800 was also obtained from a mean escapement of 31,300 (20,000 to 40,000 interval).

DISCUSSION

Nutrient enrichment of Coghill Lake continues to show positive effects on lake productivity. Specifically, the mean total phosphorus (TP) concentration increased by 22% after enrichment and the mean chlorophyll *a* concentration increased by 250% (Table 4). We believe the higher TP and chlorophyll *a* concentrations during enrichment are not attributed to a general increase in nutrient input from the surrounding watershed, nor an increase from the input of inorganic phosphorus associated with glacier melt. This is further supported by the fact that the reactive silicon, found in the surrounding watershed but not in the fertilizer, was lower in mean concentration during enrichment compared to pre-enrichment (Table 4); this suggests a decrease in terrigenous loading (Dillon and Rigler 1974). Although TP and iron concentrations are highly correlated with glacial turbidity (Edmundson and Koenings 1985; Koenings et al. 1986), the 22% increase in mean TP concentration during enrichment cannot be explained by turbidity because the mean turbidity concentration before and during enrichment did not significantly ($p = 0.113$) differ (Table 4). Moreover, iron concentrations during both pre-enrichment and enrichment periods were nearly identical; indicating no substantial increase in glacial silt (turbidity) loading. The mean TP concentration is at the level expected (10 $\mu g\ L^{-1}$) based on our nutrient loading calculations (Vollenweider 1976); and the higher TP concentration in 1995 (11.9$\mu g\ L^{-1}$) compared to other years (Appendix D) is likely due to the increase in fertilizer applied in 1995 (Table 3; Appendix A).

The substantially higher standing crop of algal biomass (250% increase in chlorophyll *a*) from enrichment improved not only the quantity but also the quality of phytoplankton. That is, those taxa of phytoplankton (e.g., diatoms) that zooplankton can feed on were found at densities exceeding pre-enrichment levels by as much as 120 times (Figure 12). This magnitude of increase provides the robust forage that zooplankton need in the face of grazing pressure from rearing sockeye salmon fry (Stockner 1987).

In 1995, *Cyclops* density was nearly identical as that in 1994, and for the three years of enrichment the density was significantly higher than during pre-enrichment (excluding 1988 data) (Table 7). The density of *Bosmina* in 1995 was lower than in 1994 but was substantially higher than during pre-enrichment (Table 7). From the interaction plot (Figure 14) it is quite evident that the 1988 zooplankton data is questionable and consequently influences the comparison of zooplankton density and biomass between enrichment and pre-enrichment periods (Table 7). There appears to have been a sampling error for 1988 because the seasonal mean *Cyclops* density was about eight times higher than other years, yet the *Bosmina* density in 1988 was similar to previous years (Figure 14). In addition, we would expect the 1988 zooplankton, which consists mainly of *Cyclops*, to be one of the lowest given the 1987 record escapement of ~190,000 and the active selection of *Cyclops* (Table 9) by sockeye salmon fry. If the 1988 data are excluded from the analysis, *Cyclops* densities were significantly higher (27% increase) during enrichment compared to pre-enrichment.

Although mean zooplankton density and biomass were higher during enrichment the current standing stock of zooplankton in Coghill Lake is relatively low compared to other sockeye salmon nursery lakes (Kyle 1996). However, in the last three years sockeye salmon smolt production increased 5-fold (Table 12), which would have a significant impact on the zooplankton forage base. Cladoceran populations are known to increase under enriched conditions (Brooks 1969; Allen 1976; Vanni 1986), and are a preferred prey item of foraging sockeye juveniles as exemplified by the relatively high number of *Bosmina* found in stomach contents of sockeye fry (Table 8), and a strong positive electivity index (Table 9). The fact that *Bosmina* (and *Cyclops* to a lesser degree) populations have remained low during enrichment may reflect high predation pressure and low recruitment since densities have been severely depressed for several consecutive years. It is known that collapsed zooplankton populations resulting from heavy grazing by juvenile sockeye salmon can take several generations to recover (Kyle et al. 1988; Koenings and Kyle 1996).

As an apparent result of an improved forage base extending through the fall during enrichment, rearing sockeye fry were larger in 1995 compared to previous years (Figure 15). This can be attributed not only to nutrient enrichment, but also to a relatively low number of fry in the lake during 1995 (Appendix G). In 1994, the fall population of rearing fish was about 7 million while in 1995 the estimate was less than 500,000. The smaller mean size of fry in 1994 may indicate increased interspecific competition for a limited forage base.

Stomach foregut samples from August and September of 1993 and 1994 (Edmundson et al. 1995) differed from that in 1995. In August of 1993 and 1994 *Cyclops* were the most abundant item found in sockeye salmon fry and represented >90% of the total biomass. In September of 1993 the fry contained mainly *Cyclops* while in 1994 the dominant prey were *Bosmina* which represented 90% of the biomass. In contrast, of the fry sampled in August and September of 1995 insect parts represented >80% of the diet biomass. Although a few insect parts can dramatically increase (and affect) prey biomass relative to zooplankton prey biomass. The observed difference in diet content could be due more to sampling and fry location. That is, in 1993 and 1994 the fry were sampled from several time periods throughout the day when estimation of daily ration was done. In 1995, the sampled fry were collected at night during the hydroacoustic survey. In addition, during the 1995 survey a very high proportion of the fry were found near the surface based on side-looking data. The combination of the fry being collected only at night where they were found near the surface may explain the difference in diet content among the years sampled.

Since we lack juvenile sockeye information and limnological (zooplankton) data for Coghill Lake prior to the decline in sockeye production, the specific cause(s) for this decline is unknown. Nonetheless, current production is the result of a very low return-per-spawner, very low nutrient pool, and small standing stock of zooplankton. Thus, the restoration project for Coghill sockeye focuses on increasing lake productivity through nutrient enrichment, which has been demonstrated as a successful tool to increase sockeye production (LeBrasseur 1978; Stockner and Hyatt 1984; Stockner and Shortreed 1985; Kyle 1994; Kyle et al. 1996). In addition, the restoration project incorporates balancing fry recruitment via management regulations of the escapement and/or fry stocking based on available forage.

Although the effects of nutrient enrichment in Coghill Lake are positive, achieving adequate escapement has been problematic due to the interception of sockeye in the commercial gillnet fisheries within the Eshamy and Coghill Districts. These fisheries target enhanced sockeye and chum salmon stocks returning to Main Bay and Noerenberg hatcheries. However, current management strategies have restricted the gillnet fishery in order to reduce the harvest of Coghill Lake sockeye and to ensure the targeted escapement goal (Donaldson 1994; PWSAC 1995).

The high escapements (> 140,000) in 1980 to 1982 resulted in similar and relatively average return-per-spawner and produced relatively large yields (Table 13). However, for five successive years (brood years 1985 to 1989) after 1982 the return-per-spawner has remained below 1.0, which we interpret as a collapse in Coghill Lake sockeye salmon production. The pattern of these results indicate that the effect of excessively high escapements on freshwater rearing capacity at least partially explains the collapse. Results of the tabular analysis using all data years indicate that an optimum escapement for Coghill Lake lies between 30,000 and 40,000 fish. However, the use of all data years in a Ricker analysis should be used with caution because of the influence of the 1980-1982 brood years. The Ricker analysis using escapements less than 100,000, which provided an optimum escapement of about 25,000, appears representative of system performance prior to the high escapements.

Finally, we recommend that caution be used when stocking of fry is combined with natural fry recruitment respective to the current rearing capacity of Coghill Lake. Stocking should be controlled such that when it is combined with the expected natural fry recruitment, that the rearing capacity is not exceeded. Modeling of the rearing capacity can be done through the zooplankton-smolt biomass model of Koenings and Kyle (1996) and estimating fry recruitment from brood year escapements. Overstocking and/or over-escaping the system will disrupt restoration of the zooplankton forage base which we are attempting to increase through nutrient enrichment.

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Table 1. Historical escapement and harvest of Coghill Lake sockeye salmon, 1961-1995.

Year	Escapement	Harvest: Wild fish	Harvest: Hatchery fish	Total run
1961	54,800	13,000		67,800
1962	26,900	13,800		40,700
1963	64,000	17,000		81,000
1964	22,200	28,900		51,100
1965	62,500	66,000		128,500
1966	82,500	49,300		131,800
1967	33,000	36,600		69,600
1968	11,800	76,100		87,900
1969	81,000	134,900		215,900
1970	35,200	36,300		71,500
1971	15,000	45,500		60,500
1972	51,000	134,600		185,600
1973	55,000	77,300		132,300
1974	22,300	99,900		122,200
1975	34,900	147,900		182,800
1976	9,100	60,500		69,600
1977	31,600	170,800		202,400
1978	42,300	203,500		245,800
1979	48,300	78,800		127,100
1980	142,300	59,100		201,400
1981	156,100	103,000		259,100
1982	180,300	947,400		1,127,700
1983	38,800	38,500		77,300
1984	63,600	95,000		158,600
1985	163,300	350,100		513,400
1986	74,100	400,100		474,200
1987	187,300	416,400		603,700
1988	72,000	83,900		155,900
1989	36,900	108,100		145,000
1990	8,300	10,500	1,800	18,800
1991	9,700	3,100	2,400	12,800
1992	29,600	14,600	44,100	44,200
1993	9,200	27,700	45,100	36,900
1994	7,300	8,000	26,000	15,300
1995	30,400	38,400	40,000	68,800
Mean	56,900	119,800	26,600	176,700

Table 2. Return of sockeye salmon by brood year for the major age classes, and the return per spawner for Coghill Lake, 1962-1995.

Brood Year	Brood year Escapement	Age class return (number of fish) 1.1	1.2	1.3	2.2	2.3	Total return	Return per spawner
1962	26,900		17,800	34,000	2,200	500	54,500	2.0
1963	64,000	200	4,400	53,800	300	5,300	64,000	1.0
1964	22,200		32,500	124,300	4,200	2,100	163,100	7.3
1965	62,500	200	25,200	48,900	1,600	1,700	77,600	1.2
1966	82,500	300	9,900	54,800	300	20,900	86,200	1.0
1967	33,000		3,800	140,100	1,400	8,000	153,300	4.6
1968	11,800		22,500	108,100	3,200	3,600	137,400	11.6
1969	81,000		12,900	60,800	7,900	10,100	91,700	1.1
1970	35,200		49,300	158,200	8,800	4,600	220,900	6.3
1971	15,000	100	5,600	32,600	2,800	5,700	46,800	3.1
1972	51,000		29,500	164,100	6,700	18,300	218,600	4.3
1973	55,000		25,500	203,100	3,300	1,800	233,700	4.2
1974	22,300	500	21,000	76,300	10,500	2,600	110,900	5.0
1975	34,900		38,300	136,700	7,700	8,800	191,500	5.5
1976	9,100	100	52,400	100,000	12,700	8,400	173,600	19.1
1977	31,600	2,000	137,100	1,108,300	1,800	2,000	1,251,200	39.6
1978	42,300	700	8,800	51,300	2,100	7,400	70,300	1.7
1979	48,300	300	17,400	105,300	6,400	21,000	150,400	3.1
1980	142,300	200	37,800	344,000	51,600	40,100	473,700	3.3
1981	156,100	400	92,500	355,900	14,600	32,800	496,200	3.2
1982	180,300	200	58,600	547,000	5,800	600	612,200	3.4
1983	38,800	100	11,800	86,800	500	7,200	106,400	2.7
1984	63,600	1,300	64,800	133,700	2,100	1,100	203,000	3.2
1985	163,300	30	1,700	13,000	1,200	800	16,730	0.1
1986	74,100	30	4,400	17,300	80	5,200	27,010	0.4
1987	187,300	20	2,200	53,700	1,400	2,700	60,020	0.3
1988	72,000	20	6,900	41,700	1,250	600	50,470	0.7
1989	36,900	10	2,600	4,700	400	900	8,610	0.2
1990	8,300	50	3,500	10,200	500			
1991	9,700	100	18,700					
1992	29,600	100						
1993	9,200							
1994	7,300							
1995	30,400							

Table 3. Fertilizer application dates and the amounts of phosphorus and nitrogen applied to Coghill Lake in 1995.

Application dates	Amount (gal) 20-5-0	Amount (gal) 32-0-0	Amount (kg) P	Amount (kg) N
1-Jul	1,402	0	144	0
6-Jul	347	0	36	0
8-Jul	347	0	36	0
10-Jul	347	0	36	0
15-Jul	347	0	36	0
15-Jul	352	0	36	0
17-Jul	352	0	36	0
20-Jul	0	350	0	564
26-Jul	365	0	38	0
27-Jul	0	350	0	564
31-Jul	375	0	39	0
31-Jul	380	0	39	0
31-Jul	375	0	39	0
31-Jul	375	0	39	0
2-Aug	375	375	39	604
3-Aug	360	0	37	0
3-Aug	405	0	42	0
3-Aug	400	0	41	0
4-Aug	400	0	41	0
6-Aug	425	0	44	0
6-Aug	375	0	39	0
7-Aug	400	0	41	0
14-Aug	400	400	41	644
15-Aug	375	400	39	644
16-Aug	1,600	0	165	0
17-Aug	1,600	0	165	0
18-Aug	800	0	82	0
19-Aug	425	0	44	0
Total	13,704	1,875	1,412	3,019

Table 4. Mean values for selected water chemistry parameters, nutrient concentration, and algal pigments derived from the 1-m stratum at stations A and B (1986-1995), and mean values for the pre-enrichment (PE) and enrichment (E) periods. Also shown are the results of ANOVA using a randomized complete block design to test for differences between stations and between PE and E periods. Probability (p) values < 0.05 indicate a significant difference.

Variable	Units	Station A	Station B	ANOVA A vs B	Treatment PE	Treatment E	ANOVA PE vs E
General Chemistry							
Conductivity	(μmhos cm^{-1})	74	74	0.915	75	71	0.178
pH	(Units)	7.1	7.1	0.169	7.2	7.0	<0.001
Alkalinity	(mg L^{-1})	19	20	0.374	20	18	0.131
Turbidity	(NTU)	5.2	4.7	0.145	4.6	5.2	0.113
Color	(Pt Units)	6.5	6.4	0.595	7.2	5.1	<0.001
Calcium	(mg L^{-1})	7.6	7.5	0.601	7.7	7.2	0.089
Magnesium	(mg L^{-1})	1.3	1.5	0.267	1.4	1.3	0.661
Iron	(μg L^{-1})	238	198	0.122	214	207	0.780
Nutrients							
Total-P	(μg L^{-1})	9.6	7.8	0.002	8.0	10.3	0.001
Total Filterable - P	(μg L^{-1})	3.6	4.1	0.496	3.7	4.2	0.481
Filterable Reactive - P	(μg L^{-1})	2.2	2.6	0.364	2.4	2.3	0.831
Total Kjeldahl - N	(μg L^{-1})	67	57	0.020	47	93	<0.001
Ammonia	(μg L^{-1})	3.3	2.6	0.115	3.0	3	0.887
NItrate/Nitrite	(μg L^{-1})	19.5	15.8	0.076	18.8	13.7	0.043
Reactive Silicon	(μg L^{-1})	666	642	0.133	726	492	<0.001
Algal Pigments							
Chlorophyll *a*	(μg L^{-1})	1.10	1.19	0.687	0.64	2.24	<0.001
Phaeophytin *a*	(μg L^{-1})	0.27	0.40	0.354	0.35	0.33	0.896

Table 5. Summary of thermal characteristics for Coghill Lake (1988-1995); seasonal (Jun-Oct) mean (T_1) and maximum (T_{max}) temperatures for the 1-m stratum, average temperature (Ts) of the water column (0-60 m), and heat content in early August.

Year	T_1 (°C)	T_{max} (°C)	Ts (°C)	Heat content (calories x 10^6) Mixolimnion	Heat content (calories x 10^6) Monimolimnion	Total calories (m^{-2})
1988	8.5	12.4	6.3	1,882	1,800	29,042
1989	9.6	16.3	6.8	2,176	1,793	31,330
1990	9.6	15.9	6.6	2,062	1,811	30,551
1991	9.9	13.5	6.2	1,855	1,808	28,899
1992	10.2	14.2	6.0	1,800	1,713	27,714
1993	11.4	17.2	6.8	2,199	1,789	31,462
1994	11.6	15.0	6.6	2,059	1,799	30,436
1995	10.3	13.9	6.1	1,827	1,750	28,217

Table 6. Summary of Coghill Lake zooplankton density (no.m^{-2}) by month for 1989-1995.

Year	Mid June	Mid July	Early August	Late August	Early September	Late September	Mid-late October	Seasonal mean
				Pre-Enrichment				
1989								
A	ND	18,312	ND	71,125	ND	ND	7,696	
B	ND	14,777	ND	61,040	ND	ND	18,313	
Mean		16,545		66,083			13,005	31,877
1990								
A	5255	3,742	ND	9,713	ND	849	6,529	
B	7245	13,854	ND	11,677	ND	3,751	4,883	
Mean		8,798		10,695		2,300	5,706	6,750
				Enrichment				
1993								
A	1,210	36,893	26,380	92,357	57,112	ND	44,320	
B	1,561	136,093	40,234	103,786	31,103	ND	28,867	
Mean	1,386	86,493	33,307	98,072	44,108	ND	36,594	49,993
1994								
A	238	23,094	34,641	12,226	41,434	40,754	3,566	
B	12,667	38,080	52,853	48,651	43,980	55,103	7,946	
Mean	6,453	30,587	43,747	30,439	42,707	47,929	5,756	29,660
1995								
A	2,674	11,886	28,101	15,963	[a]	27,579	17,516	
B	4,754	32,868	21,821	88,300	[a]	25,371	32,642	
Mean	3,714	22,377	24,961	52,132		26,475	25,079	25,790

[a]Clogging of the zooplankton net due to algae caused these samples to be unreliable for density determination.

ND No data collected

Table 7. Mean values for macrozooplankton variables for stations A and B (1988-1995), and mean values for the pre-enrichment (PE) and enrichment (E) periods. Also shown are the results of ANOVA using a randomized complete block design to test for differences between stations and between PE (including and excluding 1988) and E periods. The probability (p) values < 0.05 indicate a significant difference.

	Station		ANOVA	Including 1988		ANOVA	Excluding 1988		ANOVA
Taxa	A	B	A vs B	PE	E	PE vs E	PE	E	PE vs E
				Density (number m^{-2})					
Cyclops	32,513	42,019	0.055	41,426	30,334	0.036	21,950	30,334	0.131
Bosmina	840	1,057	0.367	131	2,311	<0.001	131	2,311	<0.001
				Body size (mm)					
Cyclops	0.73	0.73	0.958	0.73	0.72	0.796	0.73	0.72	0.747
Bosmina	0.42	0.37	0.436	0.45	0.35	<0.001	0.46	0.35	<0.001
				Biomass (mg m^{-2})					
Cyclops	60.8	71.5	0.403	73.1	54.6	0.177	36.6	54.6	0.082
Bosmina	1.3	1.0	0.253	0.3	2.6	<0.001	0.3	2.6	<0.001

Table 8. Mean abundance (number fry $^{-1}$) and biomass (µg fry $^{-1}$) of stomach contents found in sockeye fry collected from Coghill Lake in 1995. The data were analyzed using one-way analysis of variance; observed *p*-values are given. Means that share the same superscript letter do not differ significantly ($p > 0.05$).

Variable	Taxa	August	September	October	*p*-value
Abundance	*Bosmina*	51.2^{a}	40.9^{a}	54.1^{a}	0.704
	Ovig. *Bosmina*	4.7^{ab}	1.6^{a}	10.2^{b}	0.019
	Cyclops	5.9^{a}	2.5^{a}	25.4^{b}	<0.001
	Ovig. *Cyclops*	0	0	0	n/a
	Harpacticoida	0.09^{a}	0.03^{a}	0.14^{a}	0.31
	Chydorinae	0.42^{a}	0^{a}	0^{a}	0.10
	Ovig. *Chydorinae*	0.12^{a}	0^{a}	0^{a}	0.093
	Daphnia	0	0^{a}	0.43^{b}	<0.001
	Rotifera	46.5^{a}	29.9^{a}	578.9^{b}	<0.001
	Insects	*	*	*	n/a
Biomass (µg)	*Bosmina*	47.9^{a}	35.2^{a}	60.2^{a}	0.355
	Ovig. *Bosmina*	5.6^{ab}	1.7^{a}	12.2^{b}	0.018
	Cyclops	24.3^{a}	10.4^{a}	103.6^{b}	<0.001
	Ovig. *Cyclops*	0	0	0	n/a
	Harpacticoida	0.22^{a}	0.08^{a}	0.35^{a}	0.346
	Chydorinae	0.38^{a}	0^{a}	0^{a}	0.087
	Ovig. *Chydorinae*	0.15^{a}	0^{a}	0^{a}	0.083
	Daphnia	0^{a}	0^{a}	0.89^{b}	<0.001
	Rotifera	3.3^{a}	2.1^{a}	47.4^{b}	<0.001
	Insects	639.8^{a}	449.8^{a}	8.0^{b}	<0.001

*Insects were not enumerated as they often consisted of body fragments.

Table 9. Ivlev's electivity index based on in-lake and fry stomach content abundance of the major zooplankton taxa in Coghill Lake, 1995. The index ranges from -1 to +1, with negative values indicating avoidance and positive values indicating preference.

		Abundance (%)		Electivity
Month	Taxa	Lake	Fry	index
16-Aug	*Bosmina*	0.4	47.0	0.98
	Ovig. *Bosmina*	0.1	4.4	0.96
	Chydorinae	0.0	0.4	1.00
	Ovig *Chydorinae*	0.0	0.1	0.47
	Cyclops	3.9	5.4	0.16
	Ovig. *Cyclops*	0.0	0.0	na
	Harpacticoida	0.0	0.1	1.00
	Rotifera	95.4	42.7	-0.38
	Cyclops nauplii	0.2	0.0	-1.00
16-Sep	*Bosmina*	0.48	54.58	0.98
	Ovig. *Bosmina*	0.19	2.07	0.83
	Chydorinae	0.0	0.0	na
	Ovig *Chydorinae*	0.0	0.0	na
	Cyclops	1.45	3.37	0.40
	Ovig. *Cyclops*	0.01	0.0	-1.00
	Harpacticoida	0.0	0.04	1.00
	Rotifera	97.86	39.94	-0.42
	Cyclops nauplii	0.004	0.0	-1.00
11-Oct	*Bosmina*	0.04	8.09	0.99
	Ovig. *Bosmina*	0.01	1.55	0.99
	Chydorinae	0.0	0.0	na
	Ovig *Chydorinae*	0.0	0.06	-1.00
	Cyclops	1.85	3.79	0.34
	Ovig. *Cyclops*	0.16	0.0	-1.00
	Harpacticoida	0.0	0.02	1.00
	Rotifera	97.94	86.49	-0.06
	Cyclops nauplii	0.0	0.0	na

Table 10. Mean wet weight, fork length, condition (K, wet weight based), and length-adjusted wet weight of Coghill Lake sockeye salmon fry collected in 1995. Also shown are probability values (p) from one-way analysis of variance results. Means that share the same superscript letter do not differ significantly ($p > 0.05$).

Variable	August	September	October	p-value
Wet Weight (mg)	633^{a}	1011^{b}	1403^{c}	< 0.001
Fork Length (mm)	38.7^{a}	45.3^{b}	50.6^{c}	< 0.001
Condition factor (K)	0.98^{a}	1.01^{a}	1.02^{a}	0.326
Length-adjusted weight (mg)	915^{a}	976^{b}	999^{b}	0.019

Table 11. Summary of smolt mark-recapture trials, trap catch efficiency, and population estimates by sample period for Coghill Lake, 1995.

	Number of Smolt					Catch	Population	
Sample period	Live	Dead	Total	Marked	Captured	efficiency (%)	Estimate	Variance
Apr 27- May 16	20,364	297	20,661	285	57	20.0	103,249	1.46E+08
May 17-19	18,040	563	18,603	413	25	6.1	309,220	3.34E+09
May 20-25	73,693	777	74,470	535	48	9.0	836,946	1.28E+10
May 26-29	2,697	26	2,723	500	15	3.0	95,714	5.26E+08
May 30 - Jun 1	8,621	605	9,226	203	33	16.3	54,378	7.16E+07
Jun 2-5	7,026	84	7,110	183	18	9.8	75,009	2.56E+08
Jun 6-10	8,293	81	8,374	150	19	12.7	68,480	1.97E+08
Jun 11-19	10,721	257	10,978	150	29	19.3	56,996	8.58E+07
Total	149,455	2,690	152,145	2,419	244	Mean = 12.0	1,599,992	1.74E+10
						95% C.I. =	1,341,247 - 1,858,737	

Table 12. Summary of smolt data for Coghill Lake sockeye salmon 1989-1995.

			Age 1		Age 2	
Smolt year	Total number of smolts	Percent age-1	Mean length (mm)	Mean weight (g)	Mean length (mm)	Mean weight (g)
			Pre-enrichment			
1989	374,400	99	52	1.1	89	5.6
1990	na	62	60	1.8	72	2.9
1991	163,700	81	55	1.5	75	4.0
1993	289,600	95	61	1.8	79	4.2
Mean	275,900	84	57	1.6	79	4.2
			Enrichment			
1994	1,271,500	97	57	1.4	76	3.5
1995	1,600,000	96	60	1.5	71	2.5
Mean	1,435,750	97	59	1.5	74	3.0

Table 13. Tabular assessment of sockeye salmon recruitment and yield for successive escapement intervals of 20,000 fish in Coghill Lake for brood years 1962-1989. Yield is defined as brood-year returns minus escapement and is reported as the mean for each escapement interval.

Escapement interval	n	Mean escapement	Mean return	Return-per-spawner	Mean yield
0-20,000	3	12,000	119,300	9.9	107,300
10,000-30,000	5	19,600	102,500	5.2	82,900
20,000-40,000	9	31,300	251,100	8.0	219,800
30,000-50,000	8	37,600	269,000	7.2	231,400
40,000-60,000	4	49,100	168,200	3.4	119,100
50,000-70,000	5	59,200	159,400	2.7	100,200
60,000-80,000	5	67,300	84,400	1.3	17,200
70,000-90,000	4	77,400	63,800	0.8	-13,600
80,000-100,000	2	81,800	89,000	1.1	7,200
>100,000	5	165,900	331,700	2.0	165,900

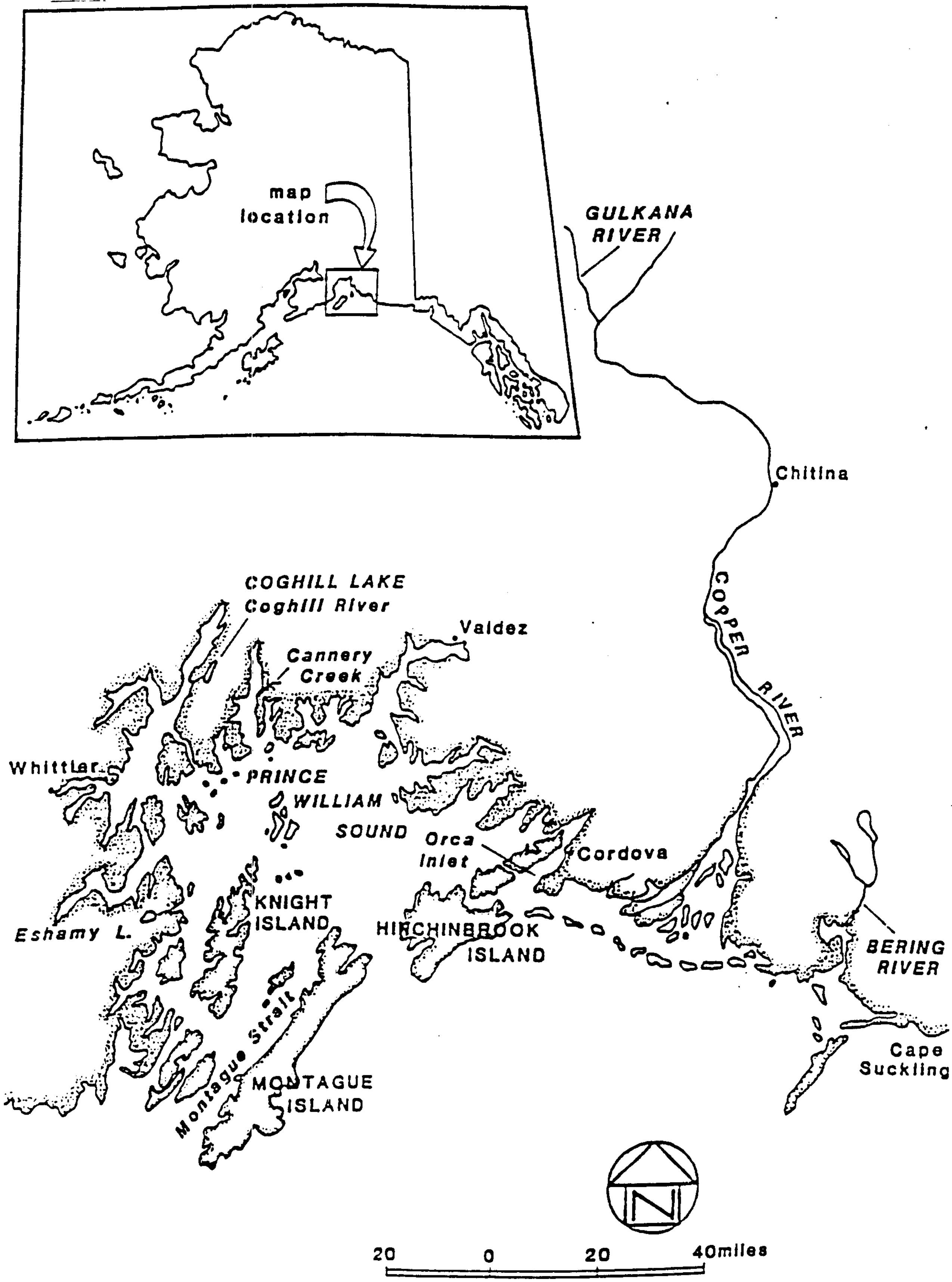


Figure 1. Geographical location of Coghill Lake in Prince William Sound.

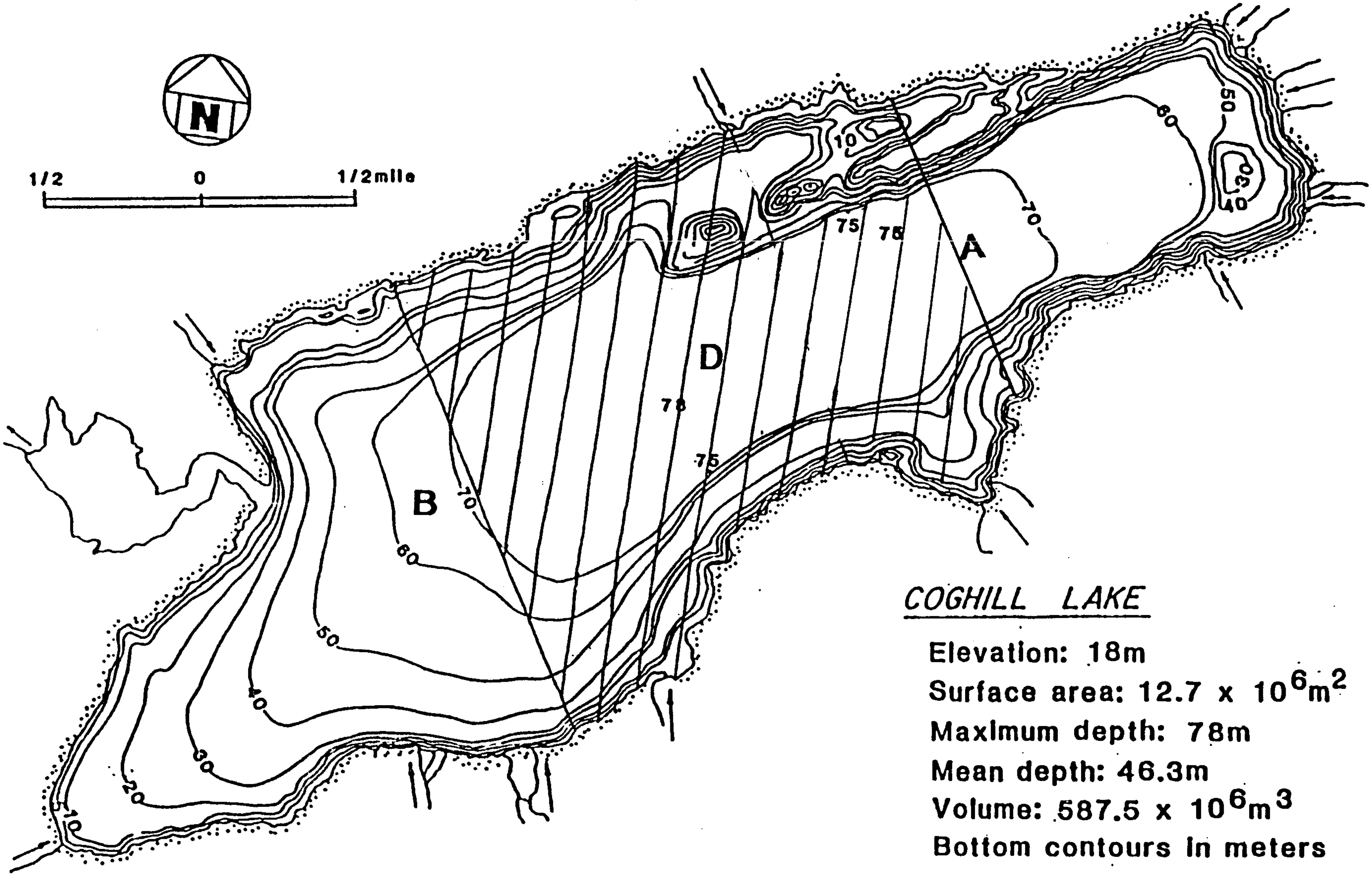


Figure 2. Morphometric map of Coghill Lake showing the location of the limnological sampling stations and the fertilizer application zone.

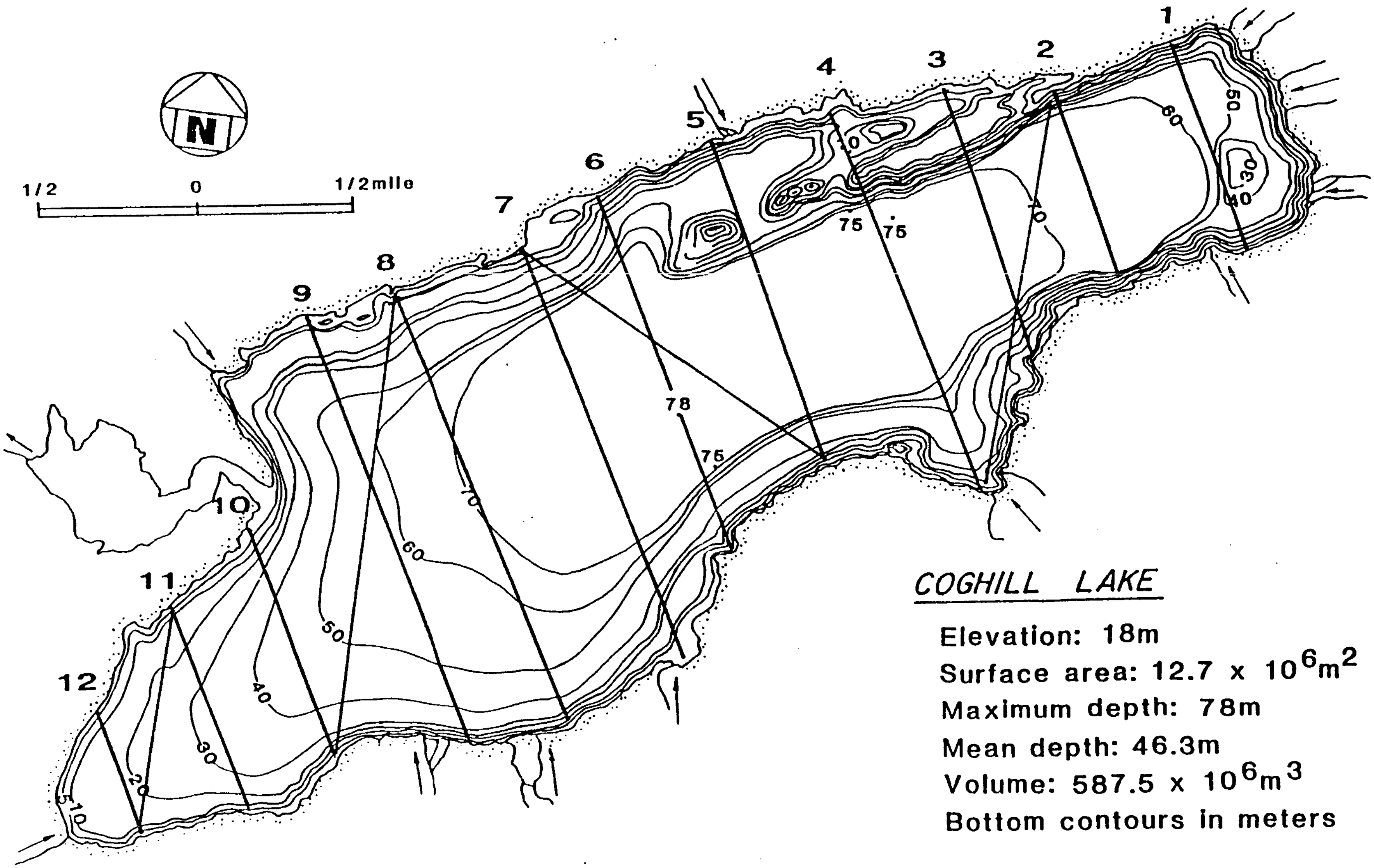


Figure 3. Location of the 12 transects used during the three hydroacoustic surveys conducted on Coghill Lake in 1995. The diagonal transects were used to collect side-looking data.

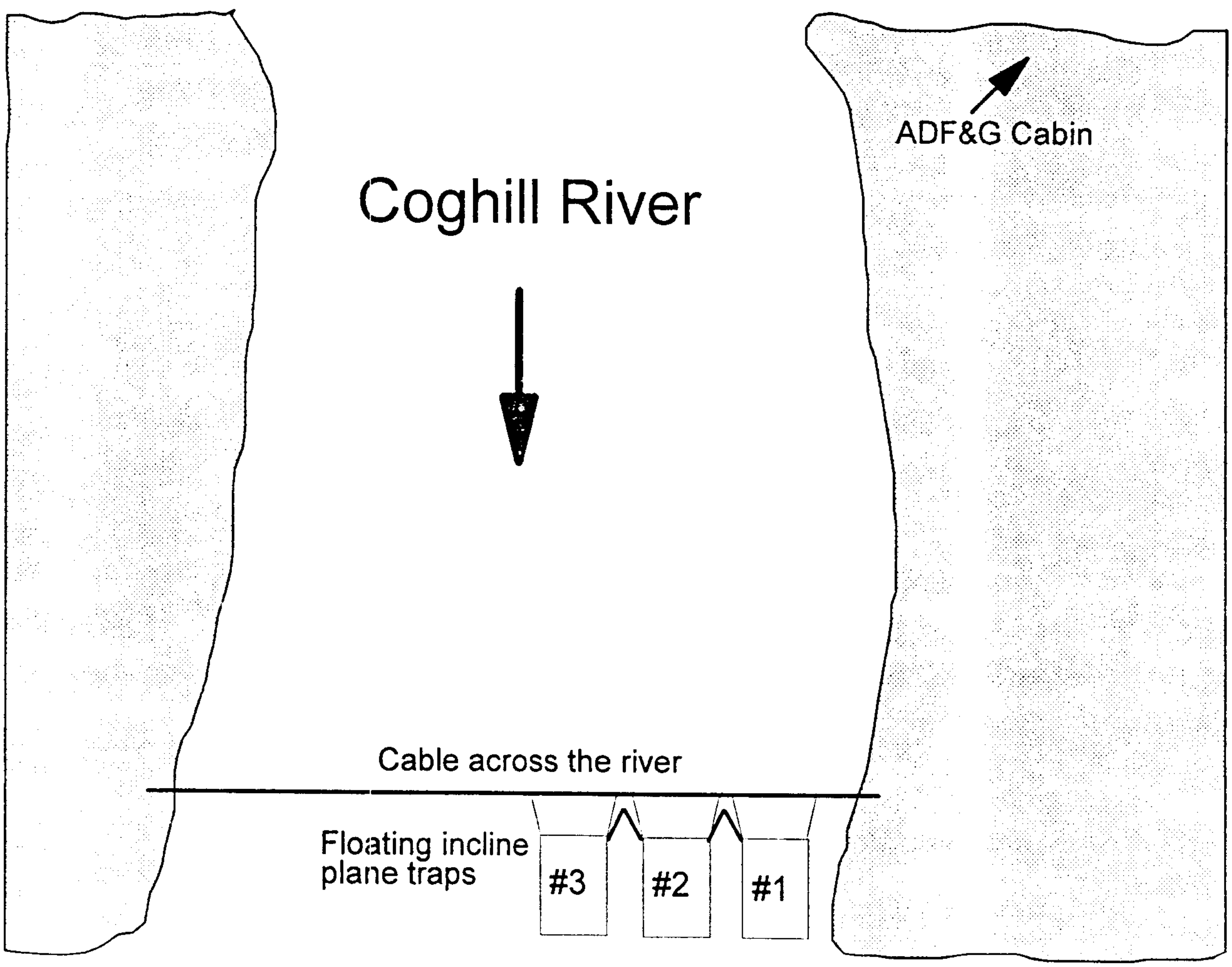
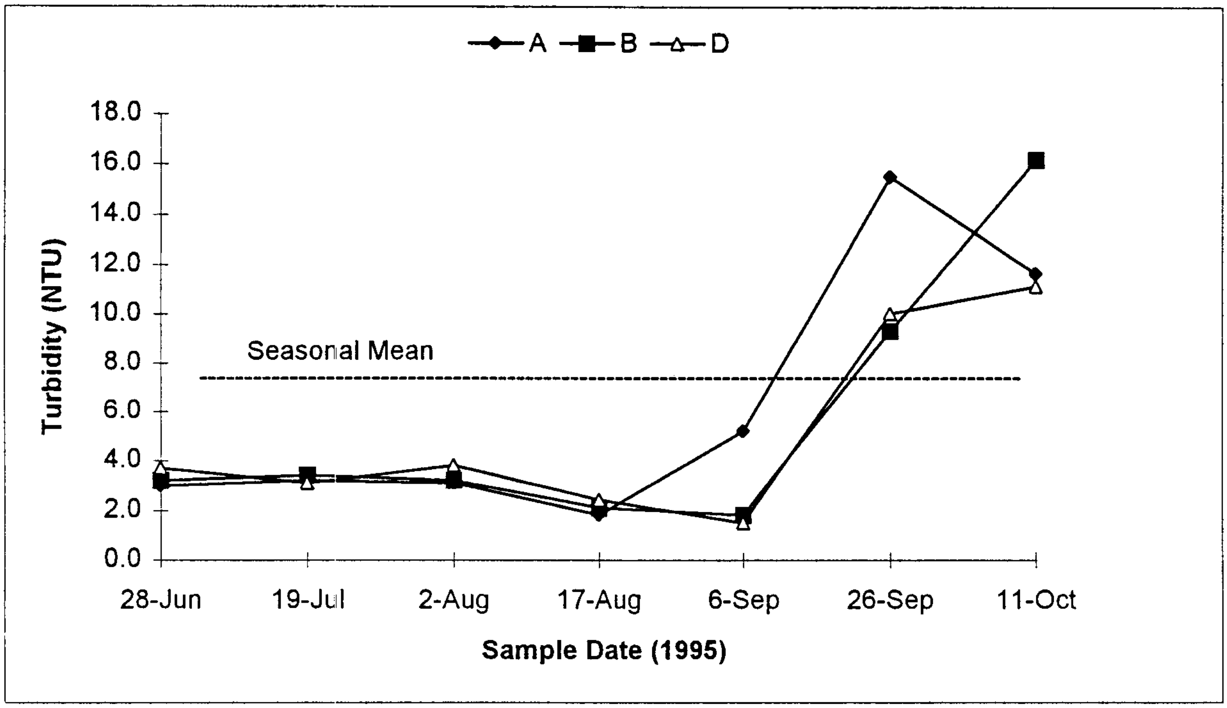


Figure 4. Configuration of the three inclined-plane (smolt) traps in Coghill River, 1995.

A.



B.

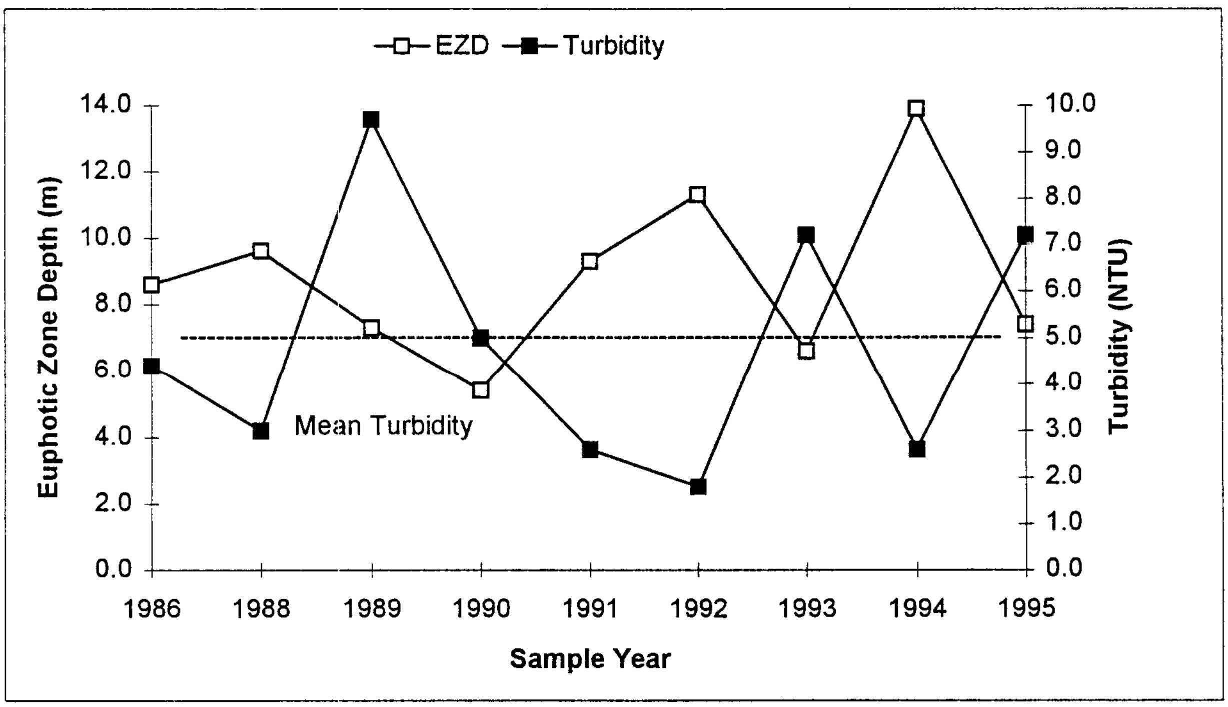
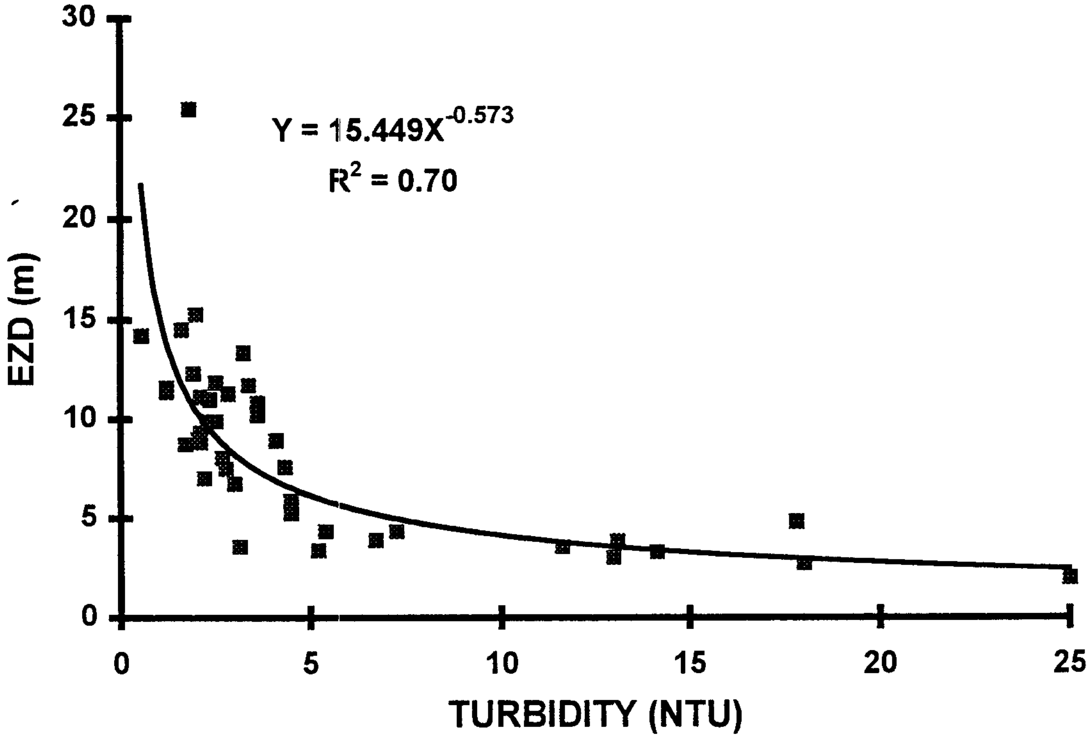


Figure 5. Seasonal changes in turbidity at three stations during 1995 in Coghill Lake (A), and seasonal mean turbidity and euphotic zone depth (EZD) for 1986-1995 (B).

A



B

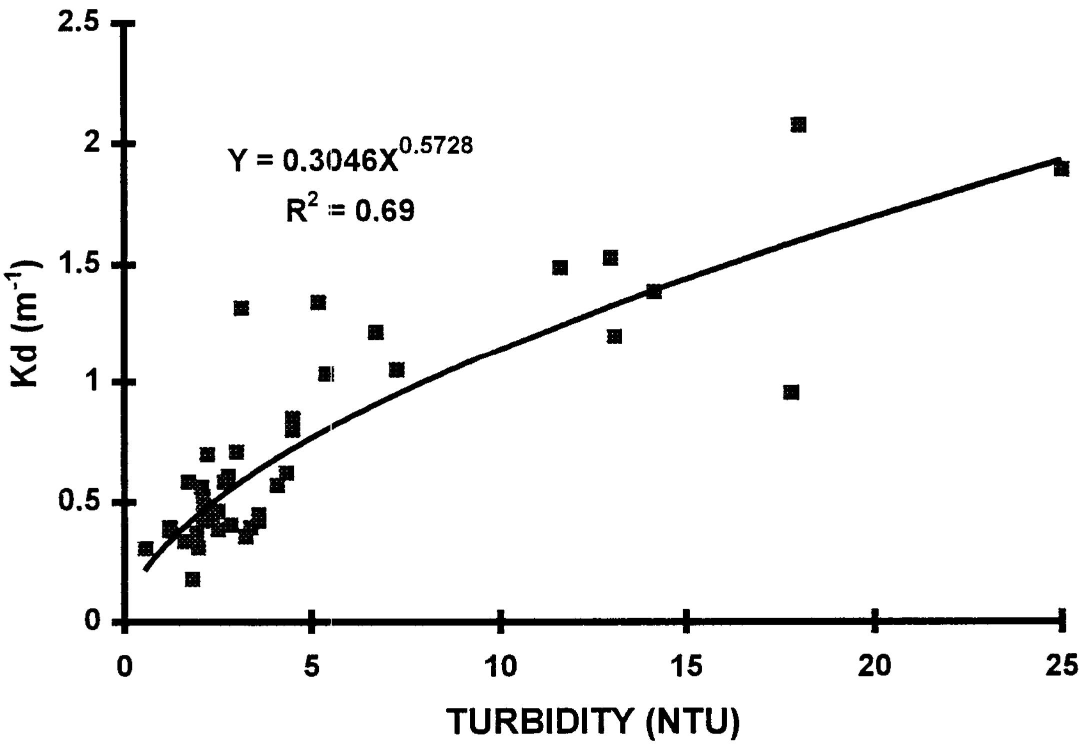


Figure 6. The relationship between turbidity and the euphotic zone depth (EZD) (A); and attenuation coefficient (K_d) (B) for Coghill Lake, 1988-1995.

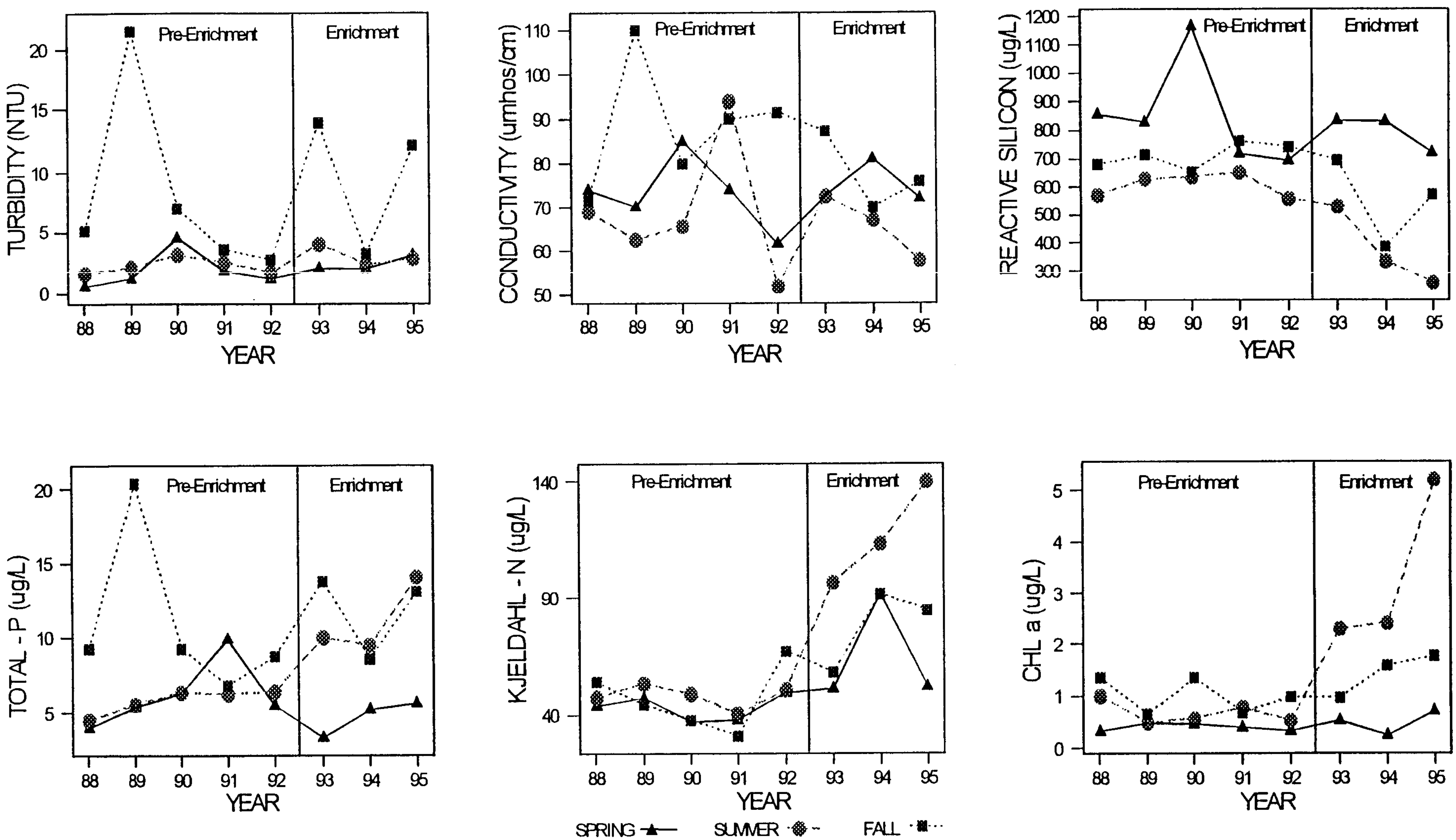


Figure 7. Interaction plot of mean responses for turbidity, conductivity, reactive silicon, total phosphorus, Kjeldahl nitrogen, and chlorophyll *a* by season (spring, summer, fall) and year (1988-1995). A split-block experimental design was used for the analysis.

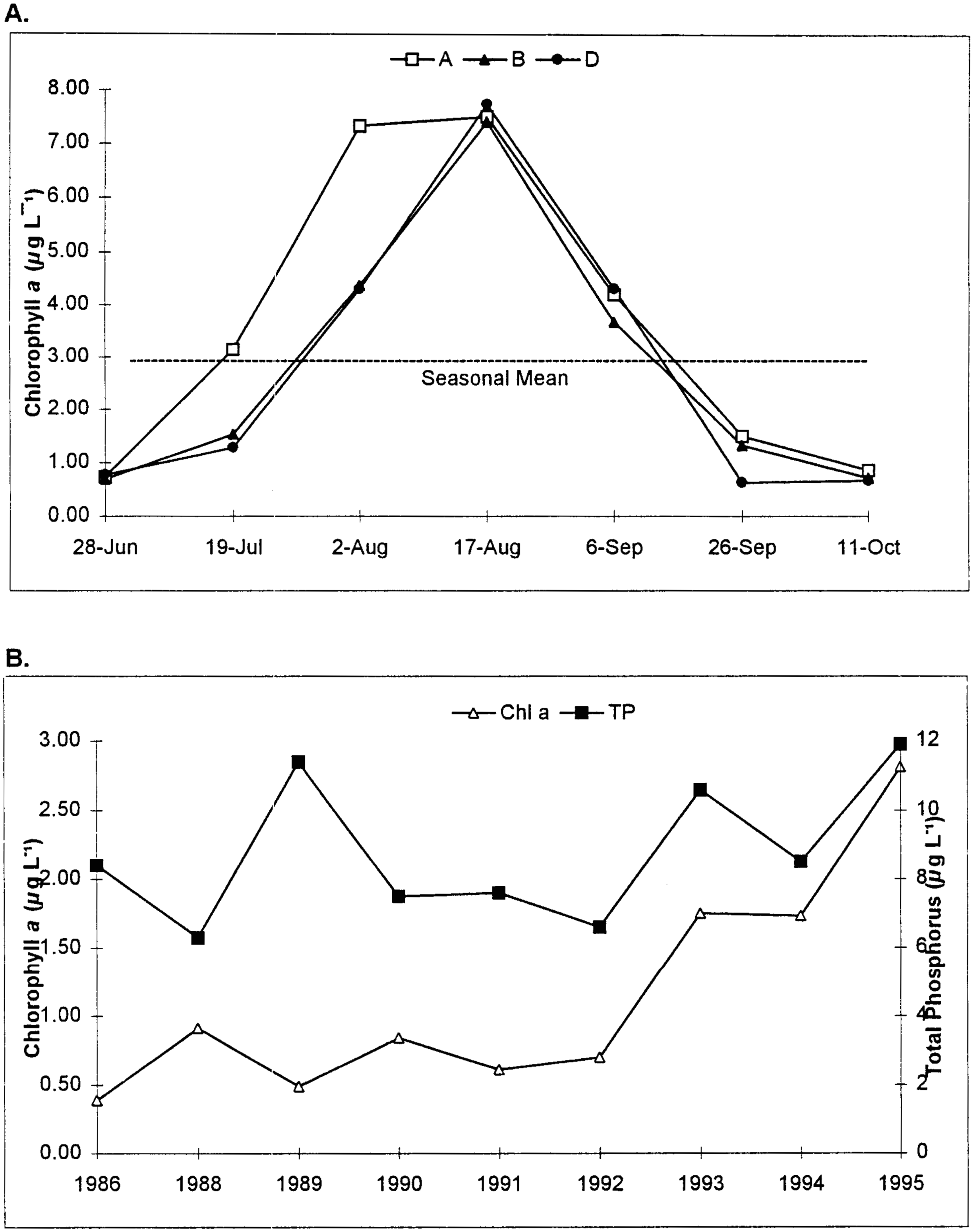


Figure 8. Seasonal changes in chlorophyll *a* concentration at the 1-m stratum for three stations in Coghill Lake during 1995 (A), and yearly (1986-1995) comparison of seasonal mean chlorophyll *a* and total phosphorus concentrations from the 1-m stratum of Coghill Lake (B).

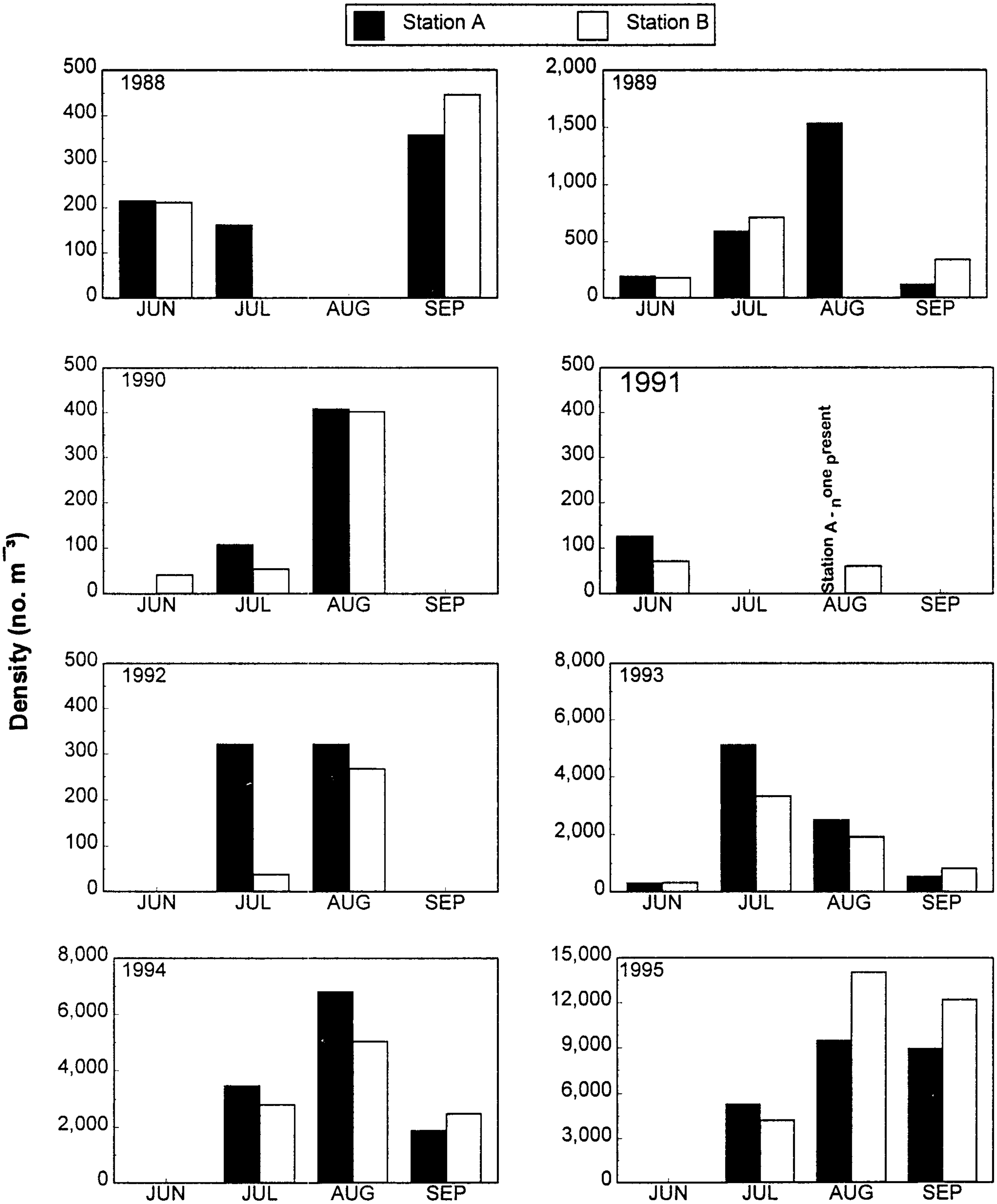


Figure 9. Density of diatoms in Coghill Lake for stations A and B, 1988-1995. Missing data indicate no data available. Note different scales for 1989, and 1993-1995.

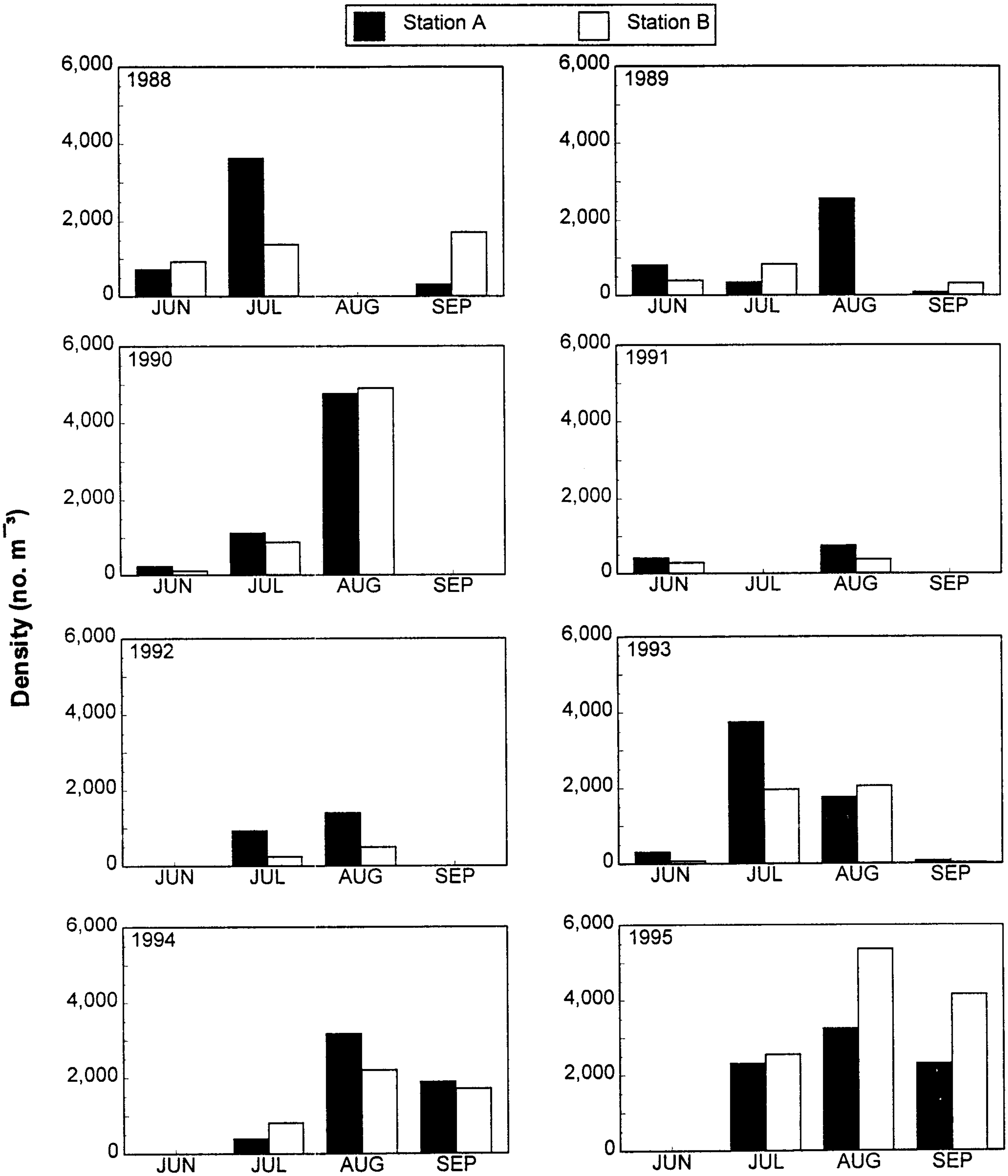


Figure 10. Density of chryso-cryptophytes in Coghill Lake for station A and B, 1988-1995. Missing data indicate no data available.

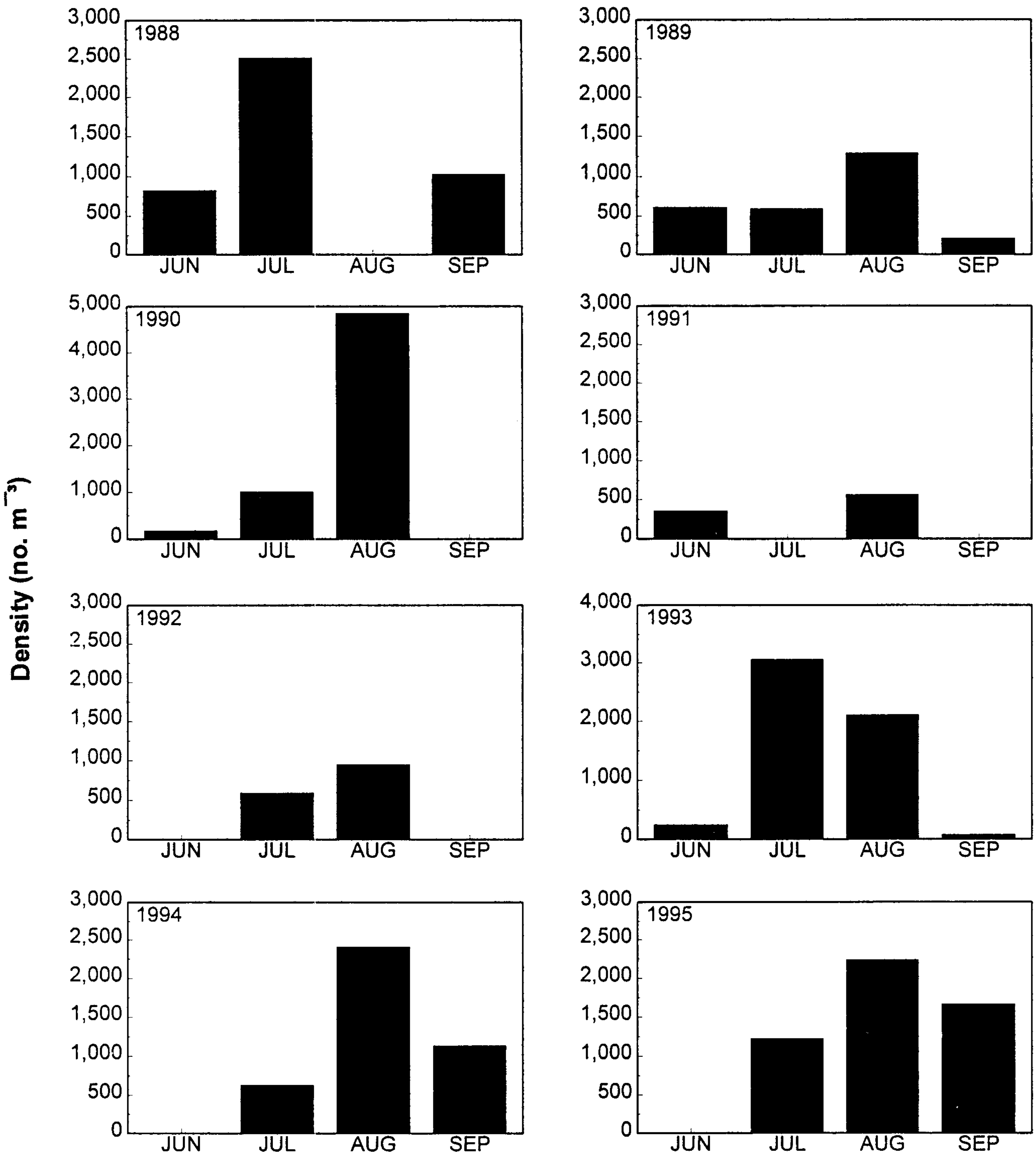


Figure 11. Density of edible chryso-cryptophytes in Coghill Lake for both stations combined, 1988-1995. Missing data indicate no data available. Note different scales for 1990 and 1993.

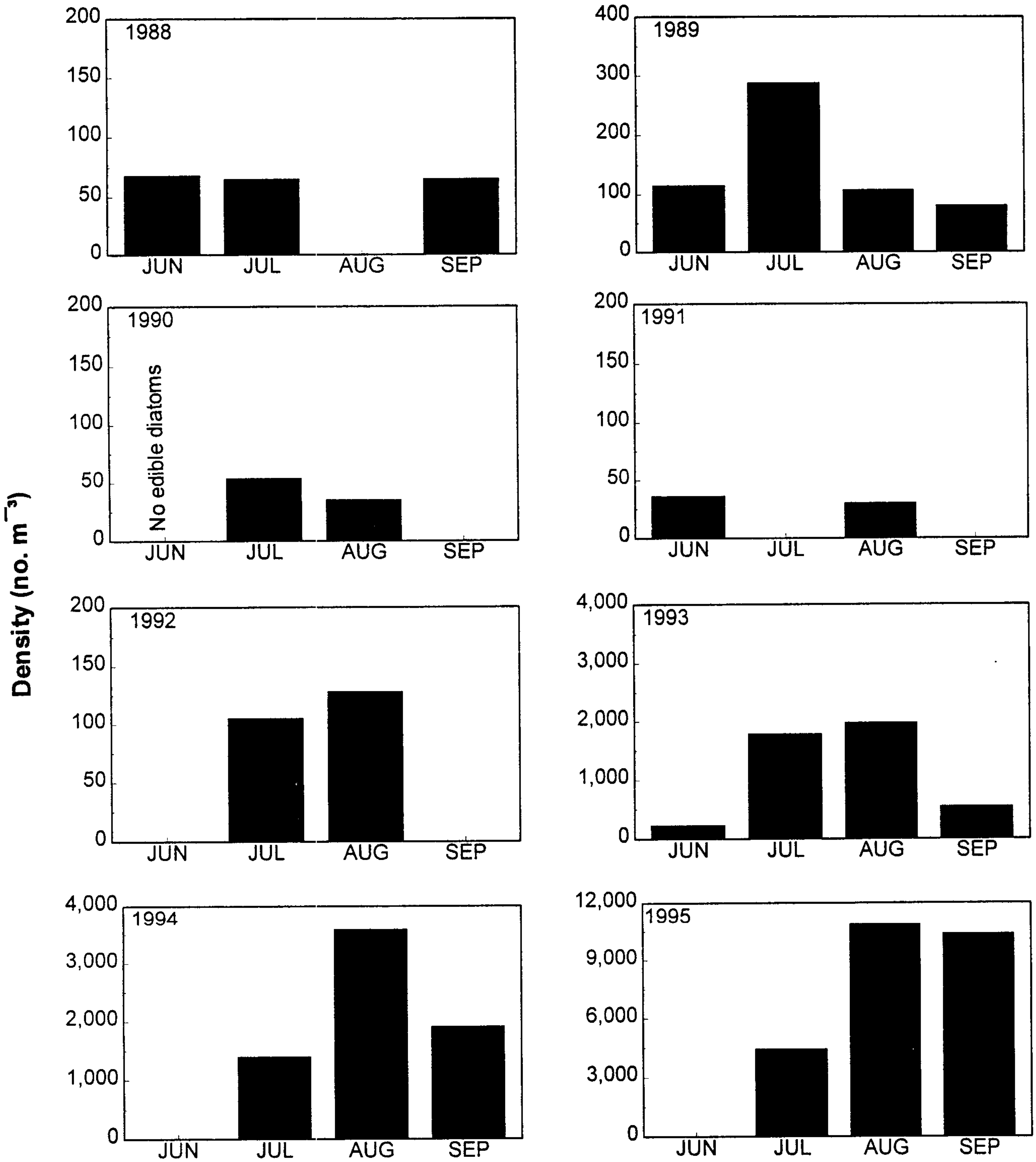


Figure 12. Density of edible diatoms in Coghill Lake for both stations combined, 1988-1995. Missing data indicate no data available. Note different scales for 1989 and 1993-1995.

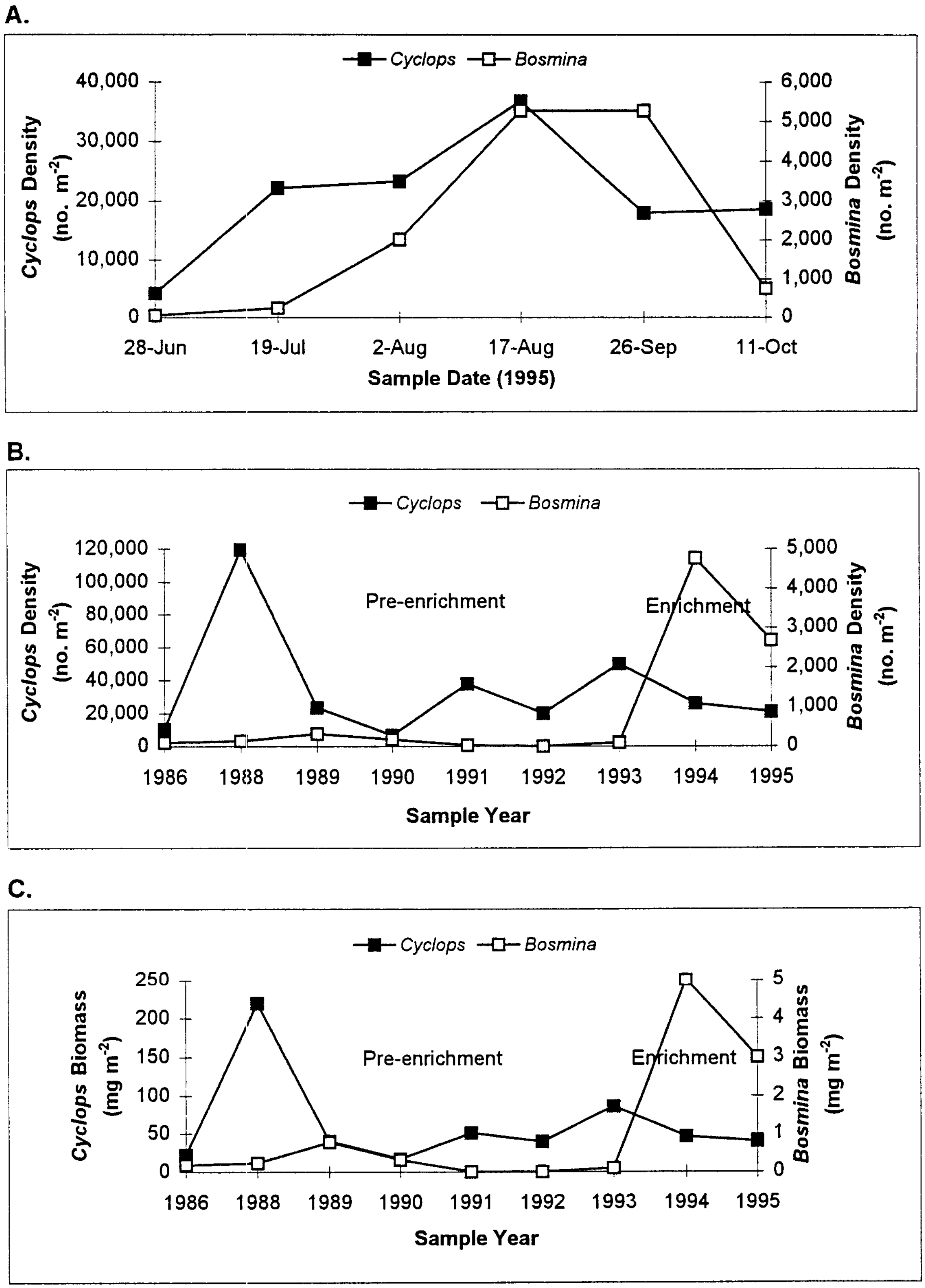


Figure 13. Seasonal changes in *Cyclops* and *Bosmina* densities in Coghill Lake for 1995 (A), and seasonal mean *Cyclops* and *Bosmina* density *(B)* and biomass (C) for 1986-1995.

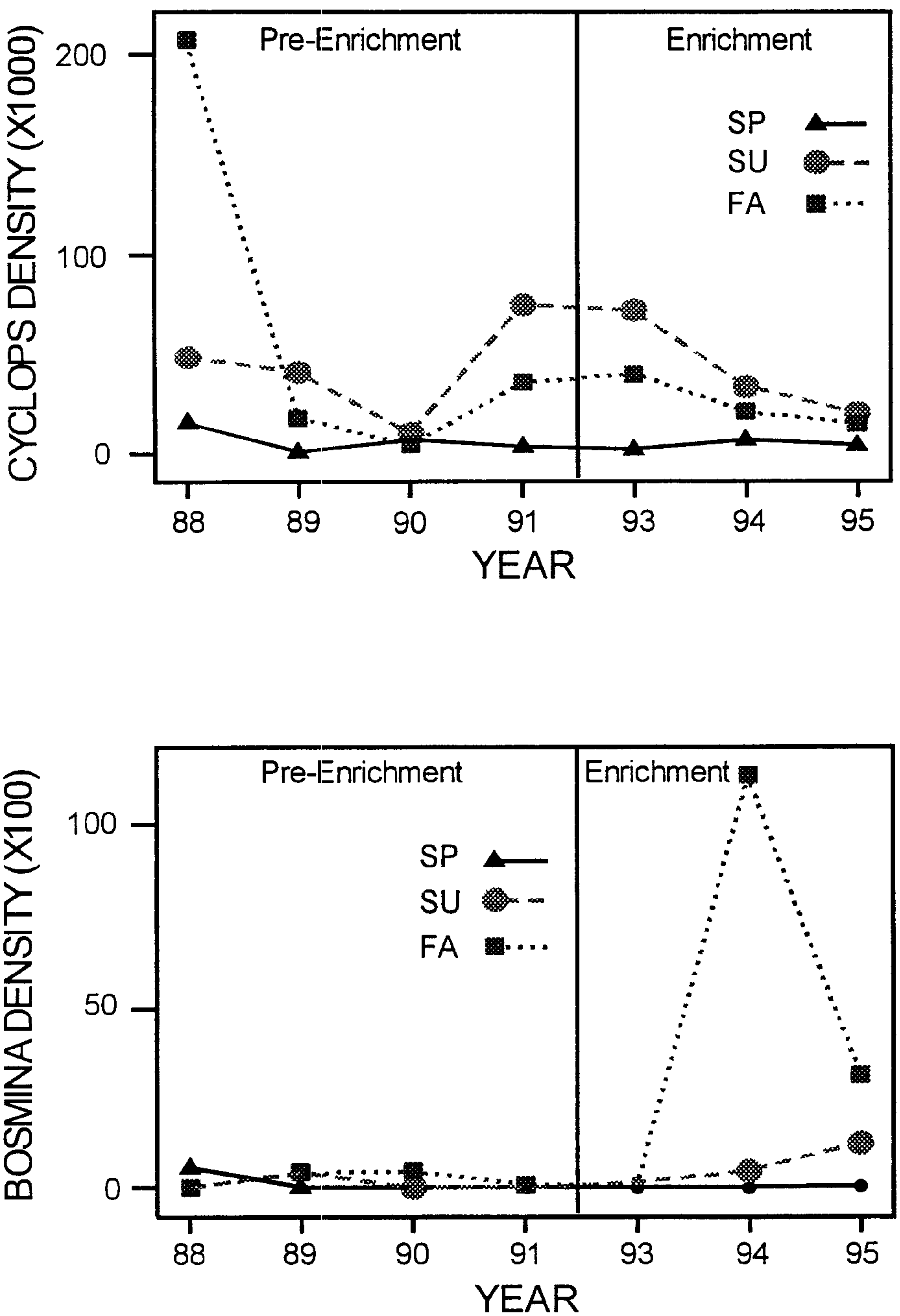


Figure 14. Interaction plot of mean responses for *Cyclops (A)* and *Bosmina* (B) density (no. m^{-2}) by season (spring [SP], summer [SU], and fall [FA]) and year (1988-1995). A split-block experimental design was used for the analysis.

A.

B.

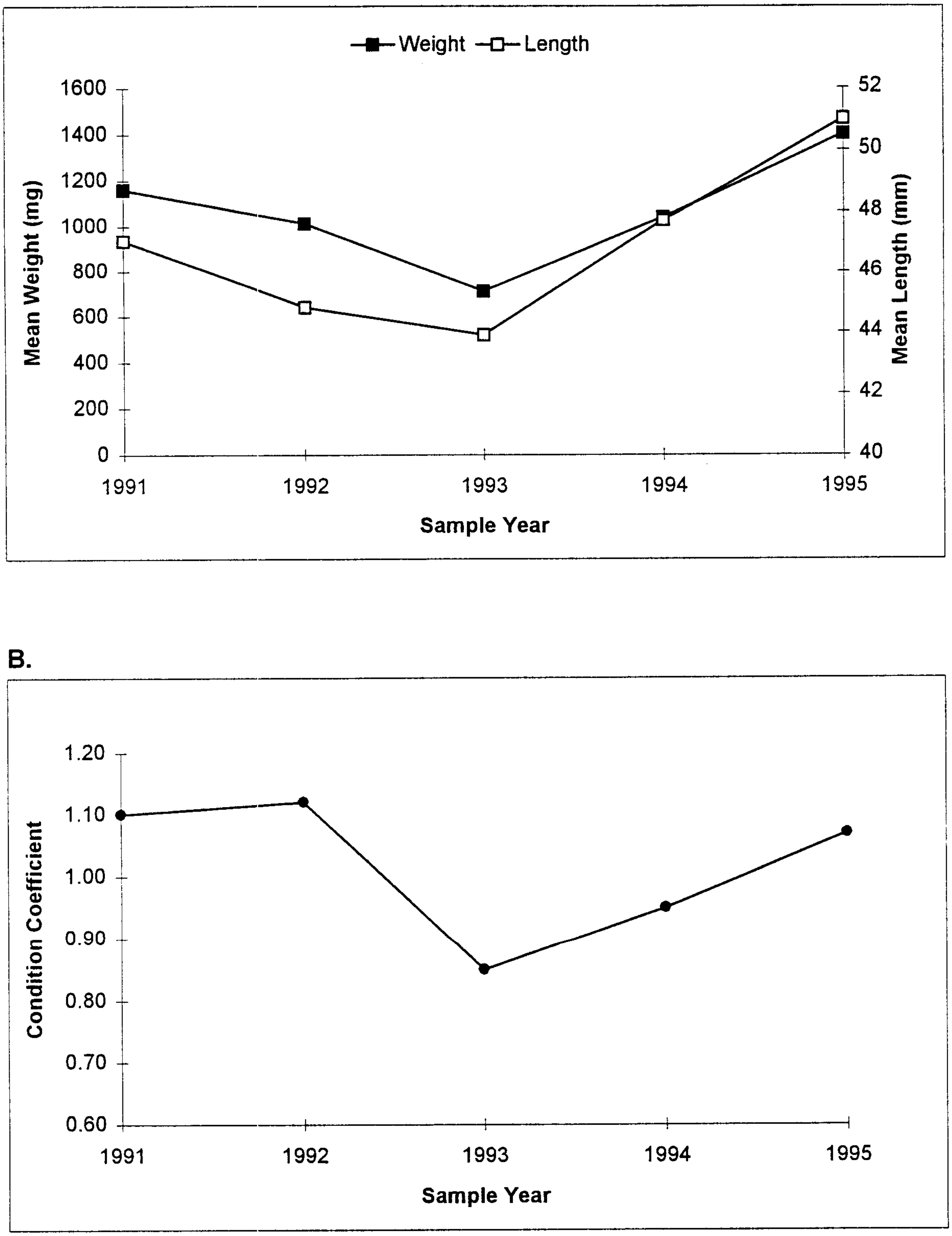


Figure 15 Mean wet weight and length (A) and condition based on wet weight (B) of age-0 sockeye salmon fry caught in late September-October from Coghill Lake, 1991-1995.

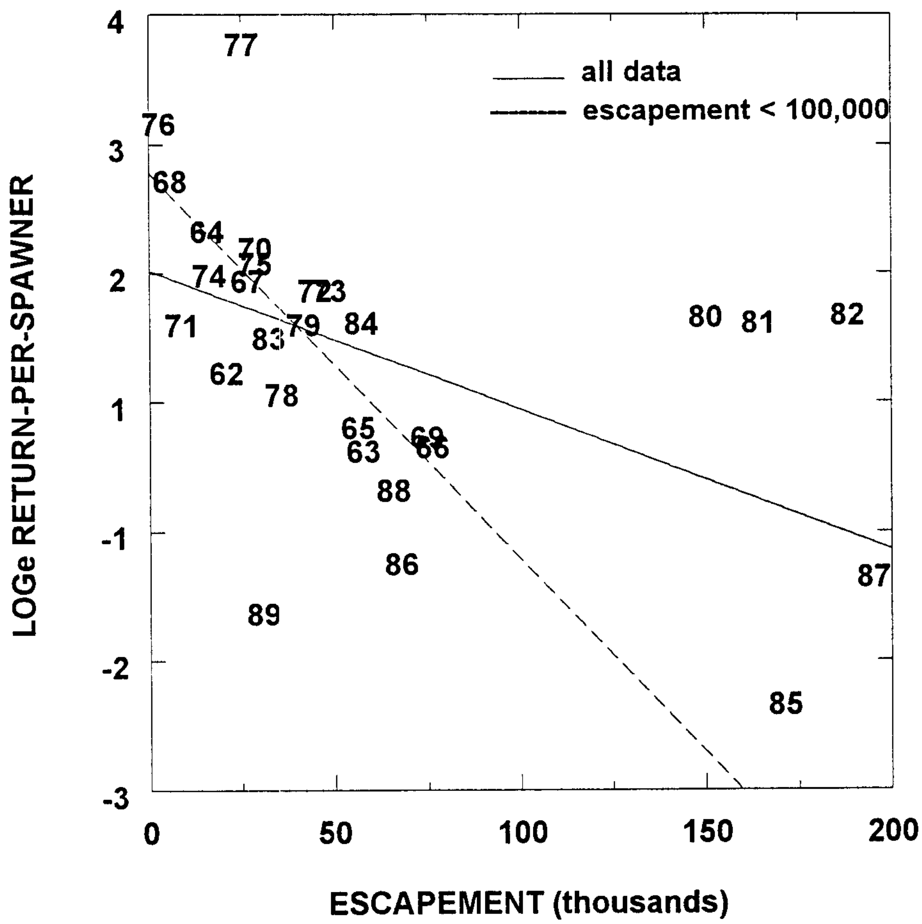


Figure 16. Results of Ricker analyses of return-per-spawner for Coghill Lake sockeye salmon for brood years 1962-1989. The solid line shows results using all historical data (r^2 = 0.26; p = 0.007); the dashed line shows results based on escapements of less than 100,000 (r^2 = 0.44; p < 0.001).

Appendices

Appendix A. Fertilizer application dates and the amounts of phosphorus and nitrogen applied to Coghill Lake in 1993 and 1994.

Application	Amount (gal)		Amount (kg)	
dates	20-5-0	32-0-0	P	N
07/01/93	495	0	51	0
07/08/93	330	0	34	0
07/09/93	495	0	51	0
07/15/93	440	0	45	0
07/16/93	495	0	51	0
07/23/93	715	0	74	0
07/24/93	550	0	57	0
07/25/93	0	550	0	886
07/31/93	0	770	0	1,240
08/01/93	0	550	0	886
08/08/93	0	165	0	266
08/09/93	0	660	0	1,063
08/10/93	0	165	0	266
08/17/93	0	1,045	0	1,682
08/27/93	0	1,073	0	1,727
Total	3,520	4,978	363	8,014
06/23/94	440	0	45	0
06/26/94	440	0	45	0
06/29/94	440	0	45	0
06/30/94	770	0	79	0
07/05/94	385	0	40	0
07/07/94	385	0	40	0
07/10/94	385	0	40	0
07/12/94	825	0	85	0
07/13/94	440	0	45	0
07/16/94	440	0	45	0
07/19/94	440	275	45	443
07/25/94	880	0	91	0
07/26/94	440	0	45	0
07/27/94	440	0	45	0
07/28/94	385	275	40	443
07/29/94	880	0	91	0
08/08/94	0	385	0	620
08/09/94	385	385	40	620
08/12/94	770	0	79	0
08/13/94	770	0	79	0
Total	10,340	1,320	1,065	2,125

Appendix B. Summary of general water-chemistry parameters, metals and nutrient concentrations, and algal pigments by sample date for the 1-m stratum of Coghill Lake,1995.

Date	Station	Cond. μmhos cm^{-1}	pH	Alk mg L^{-1}	Turb NTU	Color (PT)	Ca mg L^{-1}	Mg mg L^{-1}	Fe μg L^{-1}	TP μg L^{-1}	TFP μg L^{-1}	FRP μg L^{-1}	TKN μg L^{-1}	Amon μg L^{-1}	Nitrate μg L^{-1}	RSI μg L^{-1}	Chla μg L^{-1}	Phaeo μg L^{-1}
6/28/95	A	73	7.2	19.0	3.0	4	7.3	1.3	138	5.9	2.5	1.1	54.8	1.9	4.0	716	0.72	0.20
7/19/95	A	61	7.2	19.1	3.2	3	6.4	1.3	101	11.8	4.2	2.2	96.2	5.7	4.0	310	3.14	0.41
8/2/95	A	56	7.1	17.9	3.1	2	6.3	1.0	79	17.6	3.9	1.2	138.4	3.0	4.6	185	7.32	0.41
8/17/95	A	57	7.1	18.3	1.8	3	6.4	1.3	84	20.2	4.6	2.1	270.6	22.6	45.9	171	7.50	0.58
9/6/95	A	54	7.2	18.0	5.2	9	6.3	1.5	138	8.5	6.0	5.2	120.1	2.5	4.0	183	4.19	0.50
9/26/95	A	64	6.9	16.0	15.5	12	5.6	1.3	687	20.4	5.6	2.1	102.6	1.7	9.0	687	1.49	0.40
10/11/95	A	76	6.6	18.8	11.6	5	6.2	1.4	520	10.8	2.5	1.7	63.4	1.7	27.9	648	0.85	0.28
11/2/95	A	93	6.5	16.5	17.2	5	8.1	1.1	441	9.9	2.5	1.6	84.7	0.1	37.5	886	1.01	0.15
6/28/95	B	71	7.1	19.4	3.2	4	6.4	1.9	138	5.3	1.6	0.8	48.4	1.4	0.2	722	0.69	0.15
7/19/95	B	60	7.0	17.8	3.4	2	7.4	0.6	124	9.5	2.0	0.4	65.9	4.6	4.0	484	1.53	0.20
8/2/95	B	56	7.1	18.1	3.2	2	6.3	1.0	72	9.2	2.9	0.9	76.3	4.0	4.0	251	4.37	0.51
8/17/95	B	57	6.9	18.4	2.1	4	6.4	0.4	57	16.0	10.2	3.1	193.3	1.4	4.0	144	7.40	0.68
9/6/95	B	55	7.2	17.7	1.8	4	6.3	0.4	74	9.8	6.8	2.0	104.9	2.5	4.0	74	3.66	0.64
9/26/95	B	72	7.0	17.0	9.3	19	6.6	0.7	341	13.2	10.4	7.8	78.3	0.7	11.3	615	1.31	0.27
10/11/95	B	82	6.8	19.2	16.2	8	7.1	1.4	646	15.1	3.3	1.7	42.5	1.7	26.4	636	0.71	0.19
11/2/95	B	110	6.5	18.2	20.7	4	7.2	1.8	972	17.2	2.7	1.7	80.8	1.7	38.3	852	0.88	0.22
6/28/95	D	71	7.2	19.7	3.7	6	7.3	1.3	154	5.1	5.9	3.2	54.0	1.9	4.0	710	0.76	0.23
7/19/95	D	61	7.0	18.6	3.1	3	6.4	0.6	128	7.2	3.5	1.3	53.2	0.4	4.0	466	1.28	0.25
8/2/95	D	56	7.2	18.2	3.8	3	6.3	1.0	102	9.9	3.0	1.0	83.4	0.4	4.0	340	4.30	0.40
8/17/95	D	76	6.8	22.0	2.4	4	6.4	0.4	60		4.8	1.9		6.2	20.8	131	7.73	0.66
9/6/95	D	54	7.3	17.8	1.5	4	6.3	0.4	68	11.8	3.0	0.8	132.8	1.4	4.0	123	4.30	0.48
9/26/95	D	72	7.0	17.0	10.0	3	6.2	1.2	415	13.4	2.7	1.9	51.6	0.7	16.4	633	0.62	0.15
10/11/95	D	79	6.9	18.6	11.1	6	7.1	2.1	453	11.4	1.8	1.3	44.1	1.7	27.1	642	0.66	0.19
11/2/95	D	99	6.8	20.0	15.5	5	7.2	1.1	554	14.6	2.5	1.4	80.0	1.7	38.3	874	0.93	0.14
Average		69.4	7.0	18.4	7.2	5.2	6.6	1.1	272.8	11.9	4.1	2.0	92.2	3.0	14.5	478.5	2.81	0.35
S.D.		14.8	0.2	1.2	5.9	3.7	0.5	0.5	248.6	4.3	2.3	1.5	51.9	4.4	14.0	265.2	2.47	0.17

Appendix C. Summary of general water-chemistry parameters, metals and nutrient concentrations, and algal pigments by sample date for the 20-m stratum of Coghill Lake,1995.

Date	Station	Cond. μmhos cm^{-1}	pH	Alk mg L^{-1}	Turb NTU	Color (PT)	Ca mg L^{-1}	Mg mg L^{-1}	Fe $\mu g\ L^{-1}$	TP $\mu g\ L^{-1}$	TFP $\mu g\ L^{-1}$	FRP $\mu g\ L^{-1}$	TKN $\mu g\ L^{-1}$	Amon $\mu g\ L^{-1}$	Nitrate $\mu g\ L^{-1}$	RSI $\mu g\ L^{-1}$	Chla $\mu g\ L^{-1}$	Phaeo $\mu g\ L^{-1}$
6/28/95	A	220	6.9	22.6	3.1	3	8.3	3.9	136	5.4	1.2	0.4	47.6	0.4	71.7	858	0.09	0.05
7/19/95	A	96	6.8	19.6	2.0	4	7.4	1.9	88	4.6	1.6	0.6	35.6	4.6	40.0	750	0.25	0.08
8/2/95	A	82	6.7	20.1	2.9	3	7.2	1.9	104	6.2	4.6	2.3	33.3	4.6	29.7	696	0.30	0.13
8/17/95	A	102	6.8	20.2	1.8	3	7.3	2.2	92	6.5	3.1	1.3	43.6	0.4	38.5	877	0.37	0.11
9/6/95	A	88	6.8	20.0	1.8	5	7.2	1.5	78	5.7	4.2	2.0	49.2	0.4	30.4	862	0.74	0.17
9/26/95	A	169	7.0	20.0	110.4	15	7.5	3.2	5607	96.5	5.2	3.0	125.7	1.4	25.3	891	0.10	0.12
10/11/95	A	216	6.8	17.6	52.9	5	7.9	4.4	1891	41.5	2.8	2.0	71.3	1.2	47.2	784	0.06	0.07
11/2/95	A	129	6.6	20.9	35.6	6	8.1	1.8	1450	31.6	2.1	1.3	95.0	0.1	47.9	966	0.20	0.16
6/28/95	B	98	7.0	20.2	2.4	5	7.3	1.9	104	4.6	1.7	0.5	47.6	2.5	35.6	809	0.22	0.09
7/19/95	B	74	6.7	18.8	2.0	3	7.4	1.3	96	6.0	1.4	0.2	46.0	6.7	26.7	762	0.35	0.17
8/2/95	B	75	6.8	19.5	2.4	4	7.2	1.9	78	3.6	1.8	0.5	32.5	1.4	26.7	696	0.28	0.14
8/17/95	B	76	7.0	21.6	1.4	8	6.4	1.3	62	3.6	2.2	0.6	50.0	1.9	26.7	838	0.32	0.12
9/6/95	B	97	6.8	20.0	2.2	4	7.2	1.5	105	6.9	2.6	1.4	49.2	1.4	38.5	772	0.44	0.18
9/26/95	B	173	7.0	19.6	75.9	4	7.1	4.2	3330	78.0	2.5	1.5	88.6	2.8	40.7	861	0.14	0.12
10/11/95	B	210	6.9	19.9	50.2	4	7.9	4.4	2012	40.1	2.9	2.1	66.5	0.7	50.1	772	0.04	0.06
11/2/95	B	230	6.7	20.3	64.0	6	8.1	3.9	2400	48.8	2.5	1.7	102.1	1.7	59.0	966	0.02	0.06
6/28/95	D	147	6.9	21.9	3.0	3	7.3	2.6	130	4.9	1.6	0.5	38.0	4.6	55.5	840	0.16	0.07
7/19/95	D	80	6.7	19.0	2.5	4	6.4	1.3	145	7.0	2.2	0.8	29.3	3.0	30.4	773	0.26	0.11
8/2/95	D	99	6.9	21.0	2.0	4	7.2	1.9	97	5.4	2.9	1.4	36.5	0.4	36.4	696	0.20	0.10
8/17/95	D	104	6.8	20.4	1.0	5	7.3	1.3	54	3.2	2.3	1.1	45.2	0.9	29.7	942	0.36	0.12
9/6/95	D	88	6.9	19.6	2.3	8	7.2	1.5	109		1.9	0.8		2.5	32.6	741	0.72	0.19
9/26/95	D	184	7.0	19.9	74.7	3	7.1	5.2	3683	84.4	2.7	2.3	85.4	3.9	46.6	892	0.10	0.08
10/11/95	D	152	6.9	19.8	42.9	8	7.1	2.9	2189	37.0	2.5	1.8	63.3	1.7	43.5	741	0.05	0.06
11/2/95	D	131	6.7	19.6	31.3	8	7.2	2.5	1526	27.5	3.0	2.3	83.2	0.1	43.5	949	0.23	0.13
Average		130	6.8	20.1	23.8	5	7.3	2.5	1065	24.3	2.6	1.4	59.3	2.1	39.7	822.3	0.25	0.11
S. D.		51	0.1	1.0	31.3	2.6	0.4	1.2	1467	28	1.0	0.7	25.4	1.7	11.4	84	0.18	0.04

Appendix D. Summary of general water-chemistry parameters, metals and nutrient concentrations, and algal pigments from the 1-m stratum of Coghill Lake for 1986-1995.

Parameter	Units	1986		1988		1989		1990		1991		1992		1993		1994		1995	
		Mean	S.D.	Mean	S.D.	Mean	S.D.	Mean	S.D.	Mean	S.D.	Mean	S.D.	Mean	S.D.	Mean	S.D.	Mean	S.D.
Sp. Cond.	μmhos cm^{-1}	63	19	73	11	83	13	75	13	86	12	62	34	77	13	71	9	69	15
pH	Units	7.3	0.4	7.2	0.1	7.3	0.2	7.1	0.2	7.2	0.3	6.7	0.6	7.1	0.3	7.0	0.3	7.0	0.2
Alkalinity	mg L^{-1}	23.8	5.2	17.4	2.0	19.0	1.2	19.3	1.2	18.2	2.6	15.8	8.0	18.3	0.9	18.2	1.9	18.4	1.2
Turbidity	NTU	4.4	5.3	3.0	2.1	9.7	2.1	5.0	2.1	2.6	0.9	1.8	1.0	7.2	2.0	2.6	0.8	7.2	5.9
Color	Pt units	6	4	4	3	8	1	5	1	11	2	7	3	4	2	5	3	5	4
Calcium	mg L^{-1}	6.6	0.4	na[a]	--	7.1	3.8	8.4	3.8	6.9	0.3	7.7	3.9	7.7	0.5	7.4	0.6	6.6	0.5
Magnesium	mg L^{-1}	1.0	0.3	na	--	1.3	0.9	1.5	0.9	2.2	1.0	0.6	0.5	1.7	0.5	1.3	0.2	1.1	0.5
Iron	μg L^{-1}	222	277	141	98	410	130	301	130	148	47	83	49	300	194	93	31	273	249
Total-P[b]	μg L^{-1}	8.4	5.8	6.3	2.7	11.4	2.0	7.5	2.0	7.6	3.3	6.6	2.8	10.6	3.7	8.5	4.0	11.9	4.3
Total filterable - P	μg L^{-1}	7.0	4.2	2.5	0.5	3.1	2.1	2.6	2.1	2.3	0.5	3.6	2.6	2.6	0.8	4.5	4.0	4.1	2.3
Filterable reactive - P	μg L^{-1}	4.1	2.6	1.4	0.4	2.5	1.6	2.2	1.6	1.9	0.6	1.8	1.3	1.2	0.6	3.0	3.1	2.0	1.5
Total Kjeldahl - N	μg L^{-1}	49	6.2	48	8	48	12	42	12.4	36	8.9	55	20	77	26	97	31	92	52
Ammonia	μg L^{-1}	3.8	3.4	4.1	1.1	1.5	0.3	1.3	0.3	5.0	2.7	4.0	1.2	2.0	1.2	1.9	0.7	3.0	4.4
Nitrate + nitrite	μg L^{-1}	8.1	8.8	22.7	17.9	27.7	17.5	18.0	17.5	20.0	12.7	18.9	18.0	13.8	10.7	8.2	11.7	14.5	14.0
Reactive silicon	μg L^{-1}	813	124	710	119	701	237	748	237	710	53	563	280	626	145	430	230	478	265
Chlorophyll	μg L^{-1}	0.39	0.32	0.91	0.47	0.49	0.95	0.84	0.95	0.61	0.18	0.70	0.33	1.74	1.05	1.73	1.18	2.81	2.47
Phaeophytin	μg L^{-1}	0.31	0.16	0.24	0.11	0.21	1.90	0.85	1.90	0.18	0.09	0.27	0.12	0.44	0.19	0.16	0.13	0.35	0.17

[a]Na = not analyzed.

[b]Uncorrected for turbidity.

Appendix E. Summary of general water-chemistry parameters, metals and nutrient concentrations, and algal pigments from the hypolimnion[a] of Coghill Lake for 1986-1995.

		1986		1988		1989		1990		1991		1992		1993		1994		1995	
Parameter	Units	Mean	S.D.	Mean	S.D.	Mean	S.D.	Mean	S.D.	Mean	S.D.	Mean	S.D.	Mean	S.D.	Mean	S.D.	Mean	S.D.
Sp. Cond.	μmhos cm^{-1}	29,625	1,702	271	58	311	105	318	93	257	133	116	31	139	25	138	14	130	51
pH	Units	7.3	0.1	7.1	0.1	7.0	0.1	7.0	0.1	7.1	0.2	6.9	0.2	6.9	0.1	6.8	0.3	6.8	0.1
Alkalinity	mg L^{-1}	436.0	103.0	21.9	1.4	22.7	2.3	22.6	1.1	21.0	2.8	20.0	1.2	20.1	1.7	20.4	1.4	20.1	1.0
Turbidity	NTU	192.5	28.7	1.8	0.2	24.9	30.1	12.5	3.4	3.2	1.7	5.7	5.9	13.2	13.5	3.4	1.8	23.8	31.3
Color	Pt units	na[b]	--	na	--	8	1	7	2	12	4	7	3	4	2	6	3	5	3
Calcium	mg L^{-1}	na	--	na	--	9.7	0.8	10.4	2.5	8.7	1.3	9.8	2.3	8.7	0.9	8.3	0.8	7.3	0.4
Magnesium	mg L^{-1}	na	--	na	--	5.5	2.3	6.3	2.6	4.8	3.4	1.2	1.1	3.1	0.8	2.5	0.6	2.5	1.2
Iron	μg L^{-1}	12,518	3,029	71	5	1,051	1,294	667	357	149	94	299	403	550	557	129	77	1,065	1,467
Total-P[c]	μg L^{-1}	100.4	30.7	4.5	0.5	20.9	21.0	15.9	4.8	5.9	3.3	11.7	13.4	16.5	18.6	5.1	3.0	24.3	28.1
Total filterable - P	μg L^{-1}	53.9	51.9	2.4	0.1	2.6	0.8	2.6	1.7	1.9	0.9	2.7	1.6	1.6	1.1	2.4	1.5	2.6	1.0
Filterable reactive - P	μg L^{-1}	39.5	36.2	1.5	0.4	2.3	0.5	2.5	1.3	1.6	0.8	1.6	1.0	1.0	0.8	2.2	1.4	1.4	0.7
Total Kjeldahl - N	μg L^{-1}	12,822	1,028	39.7	5.4	49	25.8	41.4	5.2	25.1	31	48.4	20.6	56.7	20	41.4	18	59	25
Ammonia	μg L^{-1}	21,186	22,107	5.1	1.0	1.4	0.5	1.4	0.4	3.9	2.0	210.4	590.2	2.0	1.1	1.8	1.0	2.1	1.7
Nitrate + nitrite	μg L^{-1}	4.9	2.7	88.6	22.4	88.3	21.2	83.1	15.2	57.2	21.0	46.7	12.7	52.2	7.9	51.1	10.7	39.7	11.4
Reactive silicon	μg L^{-1}	4,029	1,263	843	128	818	51	902	217	809	16	743	54	793	99	810	43	822	84
Chlorophyll	μg L^{-1}	0.01	0.01	0.27	0.25	0.09	0.05	0.08	0.08	0.13	0.07	0.36	0.34	0.32	0.15	0.22	0.11	0.25	0.18
Phaeophytin	μg L^{-1}	0.16	0.22	0.13	0.10	0.12	0.07	0.10	0.04	0.10	0.06	0.26	0.13	0.20	0.07	0.09	0.05	0.11	0.04

[a]Hypolimnetic samples were collected from 60 m (1986), 25 m (1988-1991) and 20 m (1992-1995).

[b]Na = not analyzed.

[c]Uncorrected for turbidity.

Appendix F. Summary of seasonal mean macrozooplankton density, body size, and biomass by major taxa in Coghill Lake, 1986-1995.

Density (number m^{-2})

Taxon/Station	1986 A	1988 A	1988 B	1989 A	1989 B	1990 A	1990 B	1991 A	1991 B	1992 A	1992 B	1993 A	1993 B	1994 A	1994 B	1994 D	1995 A	1995 B	1995 D
Cyclops	10,347	112,887	125,772	24,011	22,819	5,175	8,137	39,791	35,881	3,254	36,600	42,992	56,805	18,100	30,536	28,394	15,226	30,530	16,576
Bosmina	88	80	186	531	109	42	308	0	53	0	0	44	136	4,178	5,891	4,204	1,905	3,708	2,401
Daphnia	77	412	212	212	159	0	0	0	35	11	319	0	0	18	0	61	53	54	0
Total	10,512	113,379	126,170	24,754	23,087	5,217	8,445	39,791	35,969	3,265	36,919	43,036	56,941	22,296	36,427	32,659	17,184	34,292	18,977

Body size (mm)

Taxon/Station	1986 A	1988 A	1988 B	1989 A	1989 B	1990 A	1990 B	1991 A	1991 B	1992 A	1992 B	1993 A	1993 B	1994 A	1994 B	1994 D	1995 A	1995 B	1995 D
Cyclops	0.78	0.79	0.67	0.68	0.72	0.85	0.84	0.60	0.66	0.71	0.76	0.70	0.72	0.70	0.63	0.81	0.77	0.73	0.71
Bosmina	0.45	0.47	0.41	0.55	na	0.40	0.44	na[a]	na	na	na	0.33	0.34	0.40	0.32	0.36	0.37	0.34	0.35
Daphnia	0.55	0.74	0.74	0.79	0.77	na	na	na	na	0.69	0.95	na	na	0.71	0.71	0.65	0.94	0.95	na

Biomass (mg m^{-2})

Taxon/Station	1986 A	1988 A	1988 B	1989 A	1989 B	1990 A	1990 B	1991 A	1991 B	1992 A	1992 B	1993 A	1993 B	1994 A	1994 B	1994 D	1995 A	1995 B	1995 D
Cyclops	22	247	191	38	41	13	20	48	54	6	73	71	101	30	42	66	32	59	30
Bosmina	0	0	0	1	0	0	1	0	0	0	0	0	0	6	5	5	3	4	3
Daphnia	0	1	1	1	0	0	0	0	0	0	1	0	0	1	0	0	1	1	0
Total	22	248	192	39	41	13	21	48	54	6	74	71	101	37	47	71	36	64	33

[a]Na indicates not available.

Appendix G1. Densities and population estimates of juvenile fish rearing in Coghill Lake by transect based on the 14 August 1995 hydroacoustic survey (down-looking data).

Transect	Fish density (no./1000 m^2)	Area ($X\ 10^3\ m^2$) transect	Area ($X\ 10^3\ m^2$) total	Weighted mean fish density (no./1000 m^2)	Lake estimate based on transect
1-a	28.83	228			
1-b	1.03	228	686	13.96	177,305
1-c	12.04	230			
2-a	1.06	223			
2-b	0.67	223	671	1.63	20,674
2-c	3.14	225			
3-a	4.77	234			
3-b	0.00	234	700	4.74	60,238
3-c	9.50	232			
4-a	16.59	461			
4-b	0.00	461	1,414	11.48	145,846
4-c	17.46	492			
5-a	0.00	385			
5-b	1.00	385	1,163	0.33	4,204
5-c	0.00	393			
6-a	0.57	414			
6-b	0.00	414	1,244	0.38	4,830
6-c	0.57	416			
7-a	3.98	517			
7-b	1.25	517	1,551	2.20	27,940
7-c	1.37	517			
8-a	1.80	558			
8-b	0.00	558	1,683	1.01	12,885
8-c	1.24	567			
9-a	1.77	532			
9-b	0.00	532	1,602	3.03	38,472
9-c	7.27	538			
10-a	0.82	287			
10-b	2.46	287	864	5.64	71,682
10-c	13.57	290			
11-a	2.55	254			
11-b	0.00	254	762	2.63	33,359
11-c	5.33	254			
12-a	0.00	118			
12-b	0.00	118	360	3.11	39,457
12-c	9.02	124			
		Total	12,700	Mean	53,074
				total estimate =	53,074
				variance =	2.52E+08
				S.E. =	15,868.2
				95% confidence interval (+/-) =	34,926

Appendix G2. Densities and population estimates of juvenile fish rearing in Coghill Lake by transect based on the 14 September 1995 hydroacoustic survey (down-looking data).

Tran-sect	Fish density (no./1000 m^2)	Area ($X 10^3 m^2$) transect	Area ($X 10^3 m^2$) total	Weighted mean fish density (no./1000 m^2)	Lake estimate based on transect
1-a	4.02	232			
1-b	19.21	232	696	14.47	183,769
1-c	20.18	232			
2-a	3.22	232			
2-b	0.56	232	691	4.41	56,003
2-c	9.56	227			
3-a	0.00	227			
3-b	0.00	227	681	0.00	0
3-c	0.00	227			
4-a	22.81	459			
4-b	8.92	459	1,381	18.79	238,678
4-c	24.60	463			
5-a	4.46	385			
5-b	4.63	385	1,157	3.76	47,802
5-c	2.21	387			
6-a	9.90	484			
6-b	14.71	484	1,464	13.13	166,708
6-c	14.73	496			
7-a	12.38	484			
7-b	25.29	484	1,458	27.90	354,381
7-c	45.82	490			
8-a	25.09	521			
8-b	11.36	521	1,569	14.12	179,352
8-c	6.01	527			
9-a	1.78	542			
9-b	39.69	542	1,632	20.42	259,347
9-c	19.80	548			
10-a	19.43	246			
10-b	18.31	246	779	18.75	238,105
10-c	18.54	287			
11-a	16.66	238			
11-b	5.24	238	722	11.83	150,186
11-c	13.52	246			
12-a	8.40	155			
12-b	15.31	155	473	14.34	182,135
12-c	19.07	163			
		Total	12,703	Mean	171,372
				total estimate =	171,372
				variance =	8.26E+08
				S.E. =	28,731.7
				95% confidence interval (+/-) =	63,238

Appendix G3. Densities and population estimates of juvenile fish rearing in Coghill Lake by transect based on the 10 October 1995 hydroacoustic survey (down-looking data).

Transect	Fish density (no./1000 m^2)	Area ($X\ 10^3\ m^2$) transect	Area ($X\ 10^3\ m^2$) total	Weighted mean fish density (no./1000 m^2)	Lake estimate based on transect
1-a	53.26	227			
1-b	48.24	227	671	41.19	523,076
1-c	21.18	217			
2-a	38.19	221			
2-b	24.64	221	657	27.29	346,584
2-c	18.81	215			
3-a	25.52	219			
3-b	21.05	219	647	21.67	275,269
3-c	18.30	209			
4-a	44.82	413			
4-b	18.17	413	1,221	31.22	396,556
4-c	30.66	395			
5-a	68.93	369			
5-b	34.77	369	1,105	39.85	506,056
5-c	15.71	367			
6-a	36.03	387			
6-b	53.39	387	1,179	37.81	480,129
6-c	24.61	405			
7-a	23.04	487			
7-b	35.05	487	1,505	32.65	414,644
7-c	39.26	531			
8-a	37.23	513			
8-b	35.91	513	1,545	31.37	398,355
8-c	21.08	519			
9-a	17.30	600			
9-b	43.91	600	1,782	30.16	383,022
9-c	29.24	582			
10-a	43.12	351			
10-b	14.28	351	1,037	19.88	252,527
10-c	1.41	335			
11-a	8.65	271			
11-b	57.41	271	807	26.36	334,739
11-c	12.71	265			
12-a	29.44	180			
12-b	22.75	180	542	20.71	263,067
12-c	10.07	182			
		Total	12,698	Mean	381,169
				total estimate =	381,169
				variance =	6.99E+08
				S.E. =	26,435.6
				95% confidence interval (+/-) =	58,184