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Taihoro Nukurangi

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CONTENTS

TABLES

FIGURES

SUMMARY

The effects of the Wilberforce Diversion on the benthic fauna and fish stocks of Lake Coleridge were studied in 1988 and 1989. During floods large amounts of silt-laden water enter the head of the lake and there was concern that this had damaged the important sports fishery for landlocked chinook (quinnat) salmon (*Oncorhynchus tshawytscha*).

Lake Coleridge contains a typical variety of benthic invertebrates, with high numbers of snails present in shallow water and worms in deep water. There was no evidence that siltation at the head of the lake had adversely affected the benthic fauna. The diversion of the Wilberforce River has probably benefitted the benthic fauna by adding organic material and by reducing fluctuations in water levels. However, this benefit has been counteracted by increased turbidity and a compression of the euphotic zone.

Common bullies (*Gobiomorphus cotidianus*) are abundant throughout the lake and the adults are large. Koaro (*Galaxias brevipinnis*) are widely spread around the lake and smaller numbers of upland bullies (*Gobiomorphus breviceps*) and longfinned eels (*Anguilla dieffenbachii*) are present. There are little data available on these native fish but comparisons with other lakes indicate that siltation has not affected the stocks.

Salmon were difficult to capture in gillnets. The few caught were large and in good condition. The salmon are mainly midwater and surface predators of insects and small fish and grow more rapidly than trout. Since 1977, when the diversion occurred, there has been no statistically significant decline in anglers' catch rates or in the growth and condition of salmon caught during the opening weekend of the fishing season.

High numbers of brown trout (*Salmo trutta*) and rainbow trout (*Oncorhynchus mykiss*) were caught in gillnets at the head of the lake in February 1988. There have been no significant changes in rainbow trout growth rates since the 1950s and trout are abundant in many New Zealand lakes with higher nutrient and sediment inputs and lower water clarity than Lake Coleridge. However, insufficient data have been collected to monitor historical changes in density.

It is concluded the Wilberforce Diversion has had no discernible effects on the benthic invertebrates or fish stocks of Lake Coleridge. However, small to moderate changes can be difficult to detect and continued research and monitoring of the fish stocks and fishery are advisable.

1. INTRODUCTION

Lake Coleridge is a large, oligotrophic lake situated in the Southern Alps, some 130 km west of Christchurch (Fig. 1). The lake supports an important fishery for landlocked chinook (quinnat) salmon (*Oncorhynchus tshawytscha*) and hundreds of anglers fish the lake during the opening weekend of the fishing season (Bowden *et al.* 1983, Flain 1986).

In 1914 the Lake Coleridge Power Station began operating and in 1921 (Bowden *et al.* 1983) the Harper River was diverted into the north west corner of the lake (Fig. 1) to increase power generation. In 1977 the Wilberforce River was also diverted into the lake via the Oakden Canal. Both rivers carry silt and the visual clarity of the lake water has declined significantly since 1977 (Biggs *et al.* 1990).

There has been concern about the potential impacts of the Wilberforce Diversion on the fish stocks and fisheries of Lake Coleridge (New Zealand Electricity Department 1975, Commission for the Environment 1976, Field-Dodgson 1980, Bowden *et al.* 1983, Jowett 1984, Fisheries Research Division 1985). However, there is little information on the ecology of fish in the lake or on their sensitivity to increases in water turbidity. The numbers and size of salmon in the spawning runs and in anglers' catches have fluctuated irregularly over the years (Webb 1982, 1990, Bowden *et al.* 1983, Fisheries Research Division 1986) and there is no evidence of any long term trends.

When the Electricity Division of the Ministry of Energy (now Electricity Corporation of New Zealand: ECNZ) applied for a 10-year renewal of water rights to divert the Wilberforce River into Lake Coleridge in 1985 (Ministry of Energy 1985), one of the conditions was that the effects of the diversion on the fish stocks and fisheries should be monitored.

Past studies on the lake were reviewed (Graynoth 1987) and a two-year research programme was undertaken using funds provided by MAF and ECNZ. The objectives were:

1. To determine whether any major changes to the chinook salmon stocks and fishery had occurred since the Wilberforce was diverted.

2. To collect benthic invertebrate samples for comparison with those collected in the late 1960s.

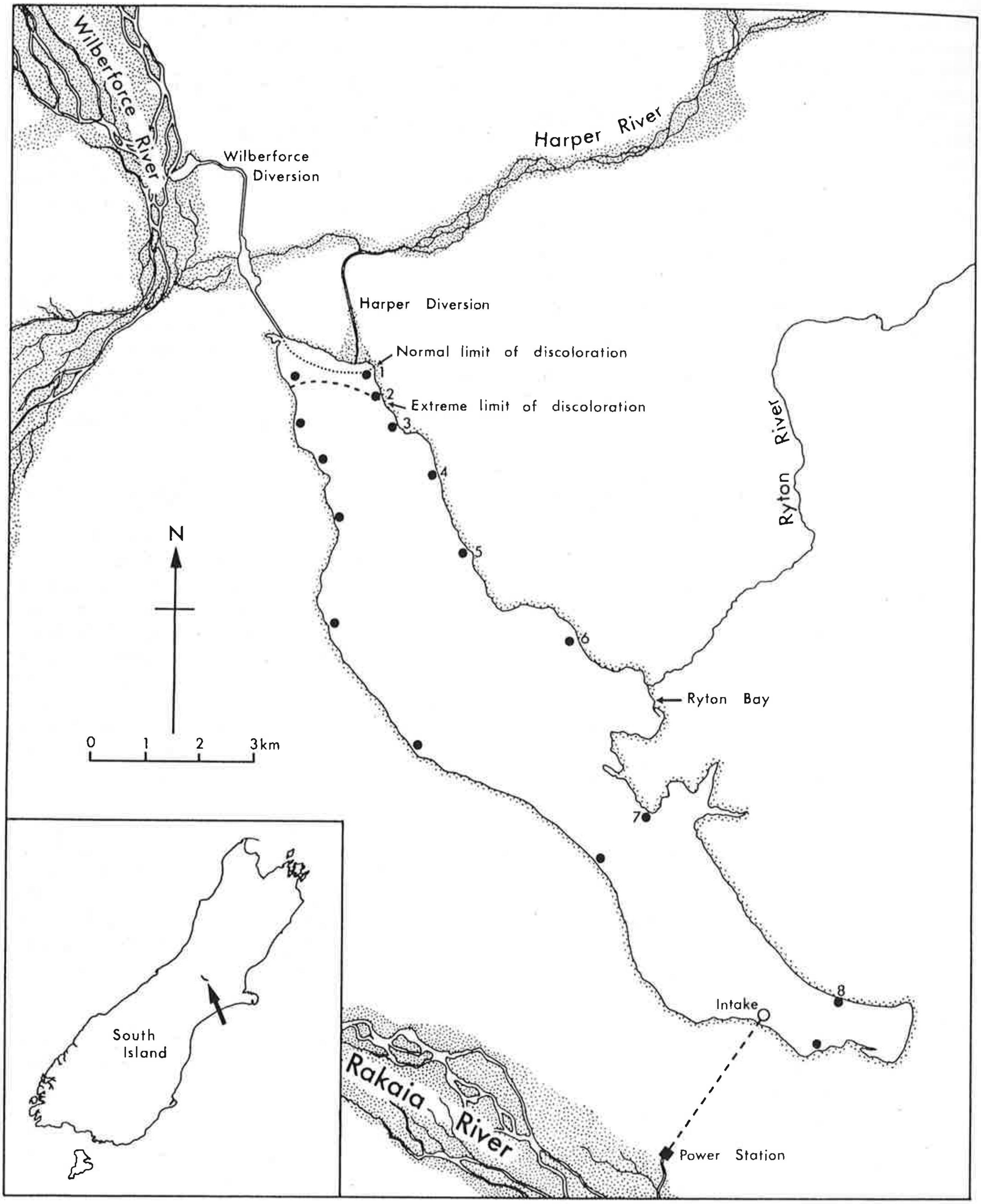

FIGURE 1. Location of Lake Coleridge and of the benthic invertebrate sampling stations in the lake.

3. To study the potential impacts of changes in water clarity and siltation on the benthic fauna and fish stocks of Lake Coleridge.

4. To assess the impacts of hydroelectric developments on the fish stocks and fishery and to suggest improvements to fisheries management and hydroelectric operations.

This is a final report on this research programme and describes the data collected, discusses the potential impacts of the Wilberforce Diversion and makes recommendations for future studies.

New data and background information on the ecology of fish and invertebrates in Lake Coleridge have been included in this report. Although some of this material is not directly relevant to the effects of the Wilberforce Diversion on the lake, it is included because there is a general lack of biological information on large alpine lakes in New Zealand.

(Note that "chinook salmon" is the preferred common name for *Oncorhynchus tshawytscha*. For consistency with the international fisheries literature, it is NIWA Freshwater's policy to use this name in all published material, rather than "quinnat salmon" which has been used within New Zealand only.)

2.1 Study area

Lake Coleridge is 17.8 km long, 3.4 km wide and has a surface area of 32.9 km^2 (Irwin 1975). It was formed by a glacier and has steep sides and a wide bed, which is mainly over 120 m deep. Maximum depth is 200 m (Flain 1970) and mean depth is 95.7 m. Its volume is 3.15 km^3 and residence time is 5.3 years, post Wilberforce Diversion (Jowett 1984).

Before power development began, only small amounts (about 4-5.7 m^3/s, Bowden *et al.* 1983, Jowett 1984) of largely sediment-free water entered Lake Coleridge from the Ryton River and several smaller streams. Nowadays the lake receives about 11 m^3/s of water and 34 000 tonnes/year of sediment from the Harper Diversion while the Wilberforce Diversion adds another 7-9 m^3/s and 100 000 tonnes/year (Bowden *et al.* 1983, Jowett 1984). Also diverted are large amounts of organic material and particulate and dissolved inorganic nutrients. From 1% to 3% of the lake area is discoloured for 43% of the time (Fig. 1 and Jowett 1984) and the secchi disc clarity of the lake has declined from 13.4 m to 8.6 m, since the Wilberforce River was diverted (Biggs *et al.* 1990, Biggs and Davies-Colley 1990). A sediment fan has developed below the Harper and Wilberforce Diversions and siltation rates in the sublittoral plant zone have increased three-fold compared to controls (30 mm per year cf. 10 mm per year, Clayton pers. comm.). Lake Stream, the natural outlet, has been dammed and most water leaves the lake via the hydroelectric intake.

Information on changes in lake water levels is contained in Table 1, which was derived from Bowden 1983 and Jowett 1984. The Harper Diversion increased lake levels by 1.1 m and increased the mean annual fluctuations from 0.9 to 2.2 m. The Wilberforce Diversion increased lake levels by another 0.3 m but, because the lake is now drawn down less over winter than in previous years, it reduced the size of annual fluctuations to 1.6 m. The annual range is usually less than the 4.15 m range permitted under ECNZ's water right (Water Right NCY 850628). During the winter (June to September) maximum flow is drawn through the power station and water levels decline to their minimum levels (Jowett 1984). From October, as electricity demand reduces and inflows increase, the

TABLE 1. Lake Coleridge water levels (Bowden *et al.* 1983, Jowett 1984).

	Period		
Statistic	Pre-Harper diversion	Post-Harper diversion	Post-Wilberforce diversion
Date start	1915	1922	1978
Date finish	1922	1978	1983
Years of record	7	56	5
Mean lake level (m)	507.6	508.7	509.0
Mean monthly fluctuations (m)	0.6	1.2	0.7
Mean annual fluctuations (m)	0.9	2.2	1.6
Extreme range (m)	3.2	4.9	2.2

water levels are increased as rapidly as possible to reach their maximum level in about February.

Strong north-west winds can increase lake edge erosion and water turbidity, especially if the lake is high (Flain 1986).

Surface water temperatures range from 8-10°C in winter, when the lake is isothermal, to 13-15°C in summer (Stokell 1934, Flain 1971 pers. obs.). In summer, water temperatures at depth are 7-10°C and there is a deep (>50 m) thermocline. The lake has been classified as ultra-oligotrophic (Burnet and Wallace 1973) and its C_{14} primary productivity is one of the lowest in New Zealand. It also has low numbers of bacteria and microbial activity (Bowden *et al.* 1983). Primary production could be limited by nitrogen and phosphorus (Mitchell 1984) as well as by carbon and trace elements (Burnet and Wallace 1973). Although high nutrient levels were recorded in one water sample collected from the Wilberforce Diversion (Mitchell 1984) very low concentrations of chlorophyll (0.37 mg/m^3) were measured in 1986 and 1987 and there is no indication of eutrophication (Biggs *et al.* 1990).

Phytoplankton is present in low numbers and consists of relatively few species of desmids, such as *Staurastrium* sp. and *Clostarium*, and diatoms, such as *Melosira* and *Synedra* (M. Flain, pers. comm., Bowden *et al.* 1983). The zooplankton consists of the copepod *Boeckella hamata* and the cladoceran *Bosmina meriodinalis* (M. Flain, pers. comm.). Although no quantitative calculations have been made it appears that zooplankton are not abundant.

In Lake Coleridge five distinct plant communities are present (Clayton 1984, Mitchell 1984). Characean algae are the predominant form and are found from the water surface to 36 m deep while native milfoils, pondweeds and *Isoetes* are present in shallow water down to 8-10 m. Lake Coleridge also contains an interesting and little-studied community of mosses and liverworts living at depths of 35-70 m, which may be vulnerable to increased siltation or a reduction in the photic zone (Clayton 1984, and pers. comm.). Surveys showed that sediment from the Wilberforce Diversion had only local impacts on the aquatic vegetation in the littoral zone and had no discernible impact on the lake vegetation as a whole (Clayton 1984). Lake edge erosion and underwater slumping and siltation are regarded as the principal factors preventing the full development of macrophyte and bryophyte communities (Clayton 1984).

The benthic fauna consists of few species and is principally composed of snails (*Potamopyrgus antipodarum*), caddisflies (*Oecetis unicolor* sp.) and chironomid larvae (Cudby *et al.* 1966). The highest densities of invertebrates (2800/m^2) were found at depths of 7-15 m.

The common bully (*Gobiomorphus cotidianus*) is the most widespread and abundant fish in Lake Coleridge (Flain pers. comm.). Other native fish include upland bully (*Gobiomorphus breviceps*), koaro (*Galaxias brevipinnis*) and longfinned eel (*Anguilla dieffenbachii*).

Lake Coleridge is the most important lake fishery in North Canterbury and of national significance (Teirney *et al.* 1982). It is one of the few lakes in New Zealand and the world which supports a landlocked salmon fishery. Salmon make up 90% of the anglers' catch and 50% of the annual crop is taken by 400-600 anglers during the opening weekend in November (Bowden *et al.* 1983, Flain 1986).

The salmon spawn in the tributary streams from April to June when most are three years old and average 450 mm in length. The fry emerge from the redds in early spring (Unwin 1976) and the majority probably migrate rapidly to the lake. The diet, survival and distribution of juveniles in the lake are unknown. Dietary studies indicate that two-year and older fish consume a significant amount of food of terrestrial origin (such as brown beetles), bullies and small amounts of benthic invertebrates (Cudby *et al.* 1966).

Rainbow trout (*Oncorhynchus mykiss*) spawn in the tributary streams from June to September. Their numbers have been monitored for many years and no significant trends in abundance have been detected to date (Bowden *et al.* 1983, Fisheries Research Division 1986). The size of fish in the runs and those caught by anglers has been stable since the early 1950s. Mature fish average about five to six years old and 550 mm in length (Flain 1986). The juveniles live in tributary streams and then migrate to the lake. Rainbow trout are probably the most abundant salmonid in Coleridge although they comprise only 9% of the anglers' catch.

Brown trout (*Salmo trutta*) spawn from mid-May to mid-August and are similar in size, age and life history to rainbow trout. Despite being abundant in the lake few brown trout are caught by anglers and not much is known of their biology in this lake.

Further information on the lake, its fish stocks and fisheries is contained in Bowden *et al.* 1983, Flain 1986 and Graynoth 1987.

3. METHODS

Prior to this study, the Lake Coleridge fishery was monitored using creel census surveys during the opening weekend of the fishing season, and by counting the numbers of salmon and trout spawning each year. This provided historical information on the size, age composition and growth rates of salmon and trout. These creel census and spawning run surveys were continued during this study. However, little was known about the distribution, movements, growth and diet of fish in the lake making it difficult to assess how the Wilberforce Diversion might influence the fish stocks. Data were therefore collected on the ecology of fish in the lake. This included sampling at increasing distances away from the sediment sources at the north end of the lake (Graynoth 1987).

3.1 Invertebrates

3.1.1 1967-68

Benthic invertebrates were collected from Lake Coleridge at regular intervals from February 1967 to March 1968 (M. Flain pers. comm.). A total of one hundred samples were taken with an Eckman grab in water depths ranging from 30 m to 223 m (mean 146 m). The grab had a mouth area of 380 cm² and the mud and invertebrates were washed through hand nets (mesh size 0.51 mm). An additional 334 samples were collected in the littoral zone by scuba divers using Wisconsin grab samplers (mouth area 929 cm², mesh size variable from 0.40 to 0.51 mm). Five samples were taken at depths of both 9 m and 18 m from each of 37 sites around the lake margin at distances ranging from 200 m to 15.3 km from the Harper River mouth. Wisconsin grabs were more effective than boat-mounted Eckman grabs which did not work on the steep underwater slopes.

Benthic animals were preserved in 4% formalin, classified to major taxonomic groups and counted; notes were made on plant debris present. The oligochaetes were sent to Professor R.O. Brinkhurst (University of Toronto, Canada) for identification. Microscopic and planktonic animals were not counted.

3.1.2 1988

In February 1988, scuba divers collected 85 Wisconsin grab samples. These were taken from just off the Wilberforce Diversion and at DSIR transects 1, 4, 6 and 8 (Biggs *et al.* 1990) at depths of 9 m and 18 m, with 10 replicate samples being collected at each depth on each transect. In the shallow water off the Wilberforce Diversion, five samples were collected at 9 m.

Samples were preserved in 4% formalin in the field. In the laboratory, samples were sorted by hand under a stereoscopic microscope and all animals extracted. Invertebrates in each sample were identified to species, where possible, and counted.

The species composition of the benthos was compared between sites using the Jaccard coefficient (J):

$$J = c(a + b - c)^{-1}$$

where a is the number of taxa in sample A, b is the number of taxa in sample B, and c is the number of taxa common to both samples. The coefficient ranges from 0 (different) to 1 (identical) and uses presence/absence data only, so community compositions were further compared using the percentage similarity coefficient (PSC), which takes into account the relative abundance of taxa.

$$PSC = 100 - 0.5 \sum_{i=1}^{k} |p_{ia} - p_{ib}|$$

where p_{ia} is the percentage of taxon i in sample A, p_{ib} is the percentage of taxon i in sample B, and k is the total number of taxa in the two samples. This coefficient ranges from 0% (different) to 100% (identical).

All other statistical tests used are detailed in Zar (1984).

3.2 Native fish

In February 1988, common bullies and koaro were collected near the mouth of the Wilberforce Diversion using a "benthic sledge" (Yockum and Tesar 1980) which was dragged along the lake bottom for about 50 m at a depth of about 10 m. A diver supervised the sledge to make certain its path was unimpeded by rocks and debris. Also in February 1988, two divers counted the number of bullies present at night along a 50 m transect across the lake bed near the mouth of the Wilberforce Diversion.

Native fish (common bully, upland bully and koaro) were caught using beach seines near the Wilberforce Diversion and in Ryton Bay from November 1988 to July 1989 (Table 2).

3.3 Salmon and trout

3.3.1 Gillnetting

Fleets of gillnets were set in 11 locations in the northern end of the lake during 8-12 February 1988 (Fig. 2). Sinking gillnets were used mainly in shallow water, whereas floating nets were set in deep water. Each fleet comprised four nets, each 25 m long and 3 m deep. The mesh sizes used were 57, 63.5, 82.6 and 101 mm. The gillnets were inspected in the morning and at dusk and the catch of trout and chinook salmon removed for measurement and sampling of scales and stomachs.

Few salmon were gillnetted in February 1988 and anglers and the North Canterbury Acclimatisation Society (NCAS) protested about the excessive number of trout which were killed. The use of gillnets was abandoned and seine nets were used instead. These were only moderately successful and we caught only juvenile salmonids and native fish (Table 2).

As ECNZ funds began to run out, a final attempt to catch adult salmon using gillnets was made in August 1989. Gillnets were set overnight at three locations around the northern end of the lake. To reduce the bycatch of trout, the nets were set 50 to 100 metres offshore, inspected at one to two hour intervals, and all trout caught were released. There were very few trout mortalities but only two salmon were caught. Overall, fewer chinook salmon and other fish were collected than originally planned.

3.3.2 Seining

Juvenile salmon and trout were caught by seining beaches along the northern end of the lake, in Ryton Bay and near the intake (Fig. 1). Multifilament seine nets, either 10 m or 30 m long, with a panel depth of 3 m, and mesh sizes of 8 mm and 25 mm respectively were used.

3.3.3 Manta board survey

In November 1988 and March 1989, attempts were made to assess the distribution and abundance of trout and salmon in the lake by towing divers mounted on a manta board at night (Flain 1989).

3.3.4 Hatchery liberations

To compensate for trout killed in gillnets, and to ensure the lake had enough salmon to study, 20 900 yearling (80 g) salmon were transferred from the MAF salmon hatchery at Glenariffe and released into Lake Coleridge in early July 1988. Five thousand of these fish were nose tagged and adipose fin clipped. An extra 7800 triploid (sterile) salmon were also released on 3 May 1989. These fish averaged 60 g and 96% had been tagged and clipped.

3.3.5 Opening weekend creel census

NCAS councillors and staff interviewed anglers and recorded details of their catches and fishing effort in November 1988 and 1989. Scale samples were

TABLE 2. Netting effort and catch in Lake Coleridge, February 1988 - November 1989.

			Catch				
Date	Netting method	Gillnet effort* or seine hauls	Quinnat salmon	Brown trout	Rainbow trout	Common bully	Other species**
Feb 1988	gillnet	36	8	57	173	0	0
Nov 1988	seine	7	47	13	31	153	32
Mar 1989	seine	34	10	5	62	155	47
Jul 1989	seine	16	1	3	9	83	1
Aug 1989	gillnet	1.5	2	4	30	0	0
Totals			68	82	305	391	80

* = 100 m of net for 24 hours.
** = koaro, upland bully and longfinned eel.

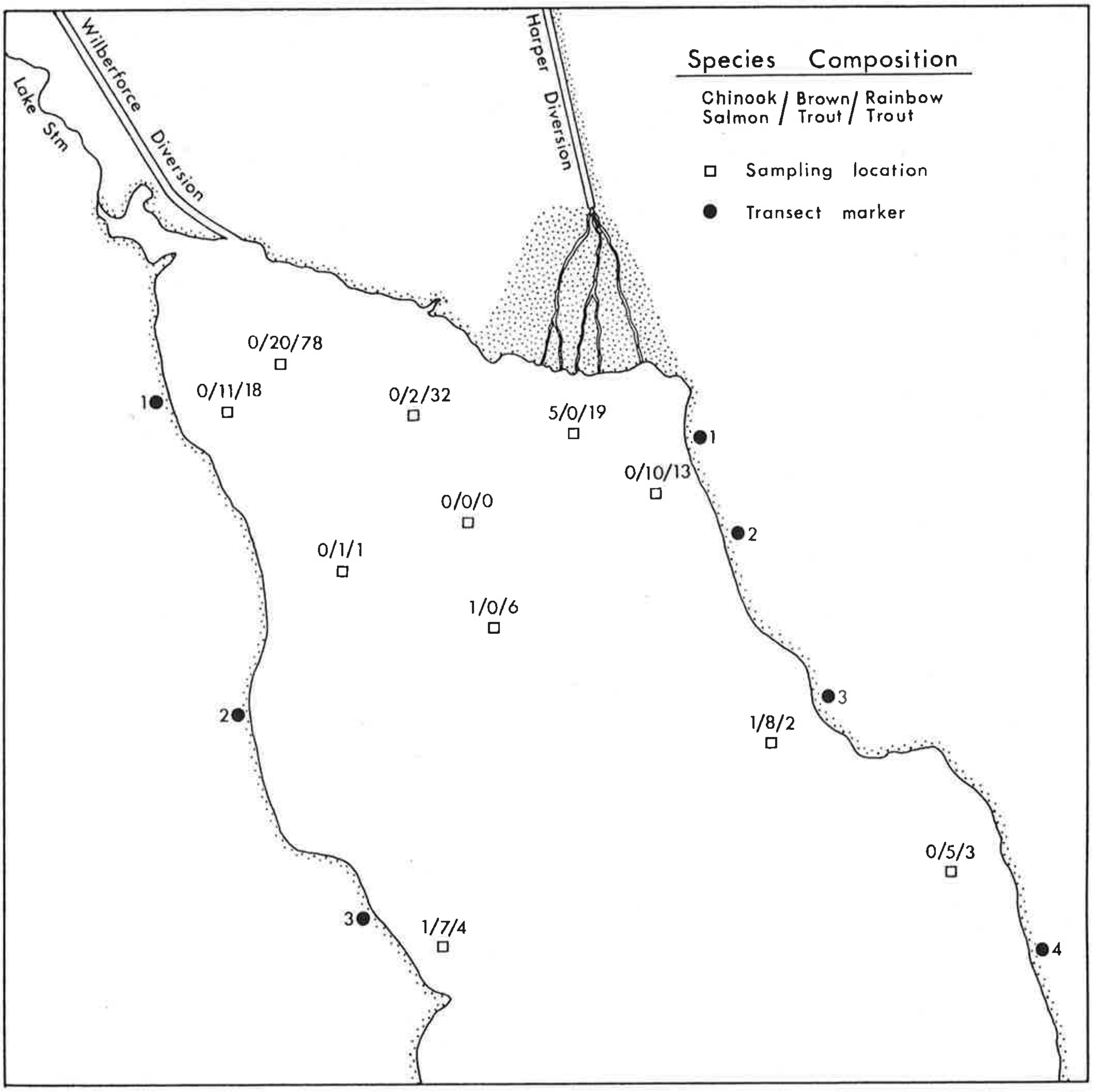

FIGURE 2. Gill net locations and catches in February 1988.

collected from most salmon caught. In 1988 and 1989, Freshwater Fisheries Centre (FFC) staff also collected 72 salmon stomachs from anglers.

3.3.6 Salmon and trout spawning surveys

NCAS staff collected otoliths from chinook salmon carcases in the Ryton River and Hennah Stream during the salmon spawning seasons of 1988 and 1989 (Ross 1988, 1989). Spawning surveys were also carried out in 1988 and 1989 to count the number of trout redds present in tributaries of Lake Coleridge (Maindonald 1989a, Webb pers. comm.).

3.4 Determination of age and growth rates

As in previous studies (Flain 1986, Webb 1982, 1989), salmon and trout in Lake Coleridge were aged using scale reading techniques (Tesch 1968). Scale samples were examined from 58 brown trout, 112 rainbow trout, and 64 chinook salmon caught in gill and seine nets in 1988 and 1989. Scales were also examined from several hundred chinook salmon caught by anglers during the November opening weekends in 1988 and 1989.

Scales were taken from the area between the dorsal fin and just above the lateral line, pressed into plastic and read using a Nikon profile projector under 50x magnification. Misshapen, damaged, badly eroded or replacement scales were ignored. Scale measurements were made, using a stage micrometer, from the centre of the nucleus, along the longest axis to each annulus and to the edge of the scale.

An annulus was defined as the last circulus of the boundary between a zone of closely and widely spaced circuli. Insufficient fish were caught in all seasons to determine when the annulus was formed. Many salmon caught in November had narrow circuli on the margins of their scales and few had wide circuli indicative of spring growth. This indicated the annulus formed in about mid November and not in May as assumed by Flain (1986). Annuli may have been formed on trout scales in August, as occurred in Lake Alexandrina (Page 1986) and Ruataniwha (Graynoth *et al.* in press).

Some fish had what appeared to be false checks on their scales (Allen 1951, Burnet 1969) which could have caused by changes in food supplies, high summer temperatures in the tributary streams or by the entry of fish into the lake.

Growth rates were determined by back-calculating the length of fish at earlier ages from scales (Tesch 1968). The radius of each annuli was used to back calculate the length of the fish at a given age using the Fraser and Lee formula (Tesch 1968). A constant of proportionality of 27 mm was calculated for brown and rainbow trout (Graynoth *et al.* in press). A constant of proportionality of 14.6 mm for chinook salmon was derived from measurements of juvenile salmon taken from the Glenariffe salmon trap (length = 14.6 + 1.958* scale width, n = 1074, r = 0.97, circuli >3).

3.5 Diet of salmonids

Stomachs were removed from large and medium sized salmonids at the lake side, preserved in 4% formalin, and assigned a unique number. Small salmonids were preserved intact. Fish fork length and location caught were noted for each specimen. In the laboratory, stomach contents were identified using keys by Winterbourn (1973), Chapman and Lewis (1976), and Winterbourn and Gregson (1981). Items of food were usually identified to genus or species level and were then grouped into one of four categories. These were:

1. Terrestrial invertebrates, presumably taken from the water surface.
2. Adult life stages of aquatic insects, usually found on the water surface.
3. Benthic invertebrates.
4. Fish, usually bullies, or rarely, koaro.

The contents of each stomach were dried on a Buschner funnel for 1 - 3 minutes and then weighed. The wet weight of the four food categories was calculated from their estimated volumes (Graynoth *et al.* 1986).

The volume and weight of each type of food was expressed as a percentage of the total. Also the percentage volume of each type of food was used to calculate the Schoener diet overlap index (Wallace 1981), i.e.:

$$\alpha = 1 - 0.5\,(\Sigma \mid p_{ix} - p_{iy} \mid)$$

where:

α = Schoener index

p_{ix} = proportion of food category i in the diet of population x

p_{iy} = proportion of food category i in the diet of population y

Schoener diet overlap indices were used to compare the percentage volume of dietary items by fish size, location and season. The index ranges from 0 (different) to 1 (identical).

3.6 Diet of native fish

Because large numbers of common bullies were caught, stratified random sampling within size classes was used to select specimens for stomach contents analysis. Upland bullies and koaro were caught much less frequently, and most specimens were used for analysis of their diet.

4. RESULTS

4.1 Benthic invertebrates

4.1.1 1967-68 samples

In the littoral zone, the benthic fauna was dominated by the snail *Potamopyrgus antipodarum*, chironomid

TABLE 3. Mean density (per m²) and relative abundance of benthic invertebrates collected from Lake Coleridge, February 1967 - March 1968.

Method	Wisconsin grab		Wisconsin grab		Eckman grab	
Water depth	(9.1 m)		(18.2 m)		(146 m (30-233 m))	
Taxon	Mean density	%	Mean density	%	Mean density	%
Total number of samples	159		175		100	
Platyhelminthes	7	+	1	+	0	-
Oligochaeta	1179	13	1237	21	2223	87
Hirudinea	8	+	1	+	0	-
Nematoda	22	+	1	+	81	3
Mollusca						
Potamopyrgus antipodarum	6049	68	3601	60	180	7
Lymnaea tomentosa	39	+	33	1	3	+
Physastra variabilis	23	+	39	1	4	+
Ferrissia neozelanica	69	1	10	+	0	-
Sphaerium novaezelandiae	399	4	88	1	20	1
Others	11	+	10	+	3	-
Crustacea	1	+	*	+	0	-
Ephemeroptera	*	+	0	-	0	-
Plecoptera	0	-	*	+	0	-
Odonata	*	+	0	-	0	-
Hemiptera	0	-	*	+	0	-
Trichoptera						
Hudsonema amabilis	111	1	75	1	0	-
Paraoxyethira	11	+	9	+	0	-
Oecetis unicolor	56	1	61	1	13	1
Other trichoptera	1	+	1	+	0	-
Coleoptera	*	+	0	-	1	+
Diptera						
Chironomidae	905	10	833	14	16	1
Others	1	+	*	+	0	-
Arachnidae	7	+	3	+	2	+
Gobiomorphus cotidianus	8	+	5	+	*	+
Mean abundance per m²	8907		6008		2545	
m² sampled	14.8		16.2		3.8	

NOTES:

* = <1 per m².
\- = not present.
\+ = <1%.

Oligochaeta - less than 40% were recorded as "tubificids". Brinkhurst identified.
Tubificidae = *Aulodrilus pleuriseta, Limnodrilus hoffmeisteri (lucasi), Rhyacodrilus (Taupodrilus) simplex.*
Phreodrilidae = *Phreodrilus lacustris, P. beddardi?, P.* new species. Naididae = *Salvina appendiculata.* One further worm was identified as *Haplotaxis smithii.*
Mollusca - *Ferrissia neozelanica* - originally identified and recorded as *Latia neritoides* but is probably the freshwater limpet based on Forsyth (1983) and Winterbourn (1973). *Lymnaea tomentosa* - originally identified as *Simlimnaea. Physastra variabilis* - could be *Physa acuta* (Winterbourn 1973).
Crustacea - included Ostracoda, Copepoda, Amphipoda. No crayfish were caught.
Ephemeroptera = *Deleatidium, Nesameletus, Ameletopsis.*
Chironomids - *Chironomus zealandicus, C. cylindricus, Pentaneura* sp. and *Polypedilum* sp. (Forsyth 1975).

larvae, worms and caddisfly larvae (Table 3). Below 18 m, worms became more abundant while the numbers of snails and all other invertebrates decreased with depth. Molluscs decreased from about 300/m^2 at 50 m to 100/m^2 at 220 m ($n = 57$, $r = -0.27$, $P < 0.05$).

In 1967-68, silt from the Harper River occasionally discoloured a small area near the head of the lake (M. Flain, pers. obs.). To test the hypothesis that siltation had increased the numbers of worms and chironomids and reduced the numbers of snails and caddisfly larvae at this location, comparisons were made between 19 samples collected within 1 km of the Harper Mouth and 91 samples collected at unsilted sites 1 to 5 km away (Table 4). The hypothesis was rejected because samples collected less than 1 km from the river mouth had lower numbers of worms, similar numbers of chironomids and significantly higher numbers of snails and caddisflies than in unsilted sites 1 to 5 km away. Therefore siltation from the Harper Diversion probably had no significant adverse effects on the invertebrate fauna in 1967-68.

4.1.2 1988 samples

A total of 25 taxa were identified from the benthos, 20 at 9 m deep and 21 at 18 m deep (Table 5). At both 9 m and 18 m the composition of the benthos was similar. Comparisons of the species lists at these depths (Table 5) using the Jaccard coefficient showed that they had 64% of the taxa in common. Calculation of the percentage similarity coefficient showed a 82.8% similarity in terms of the relative abundance of taxa. Each taxon exclusive to 9 m (*Zelandobius, Oxyethira albiceps, Hydora* and Tipulidae) occurred at a density of less than one animal per m^2. Likewise, with the exception of Cladocera, taxa which were exclusive to 18 m (Hirudinea, Cladocera, *Deleatidium, Olinga feredayi* and Eriopterini) also occurred at densities of less than one animal per m^2. The mean density of Cladocera was 3.8 animals per m^2.

Higher total densities were found at 9 m than 18 m (Table 5) and this was the case for all the taxa, except for Oligochaeta Copepoda and *Oecetis unicolor* for which densities were higher at 18 m, but the differences were not significant ($P > 0.05$). Major taxa which were consistently more abundant at 9 m than 18 m included Gastropoda, Bivalvia and Chironomidae (Table 6).

At both depths, the greatest proportion of the total density of macroinvertebrates was formed by the gastropod *Potamopyrgus antipodarum* which comprised 50.4% of the fauna at 18 m and 65.4% at 9 m (Table 5). Chironomidae was the next most abundant taxon, comprising 18.7% (9 m) - 22.0% (18 m) of the fauna. Oligochaeta was the only other taxon to comprise more than 10% of the benthos at either depth. Together, these three taxa comprised 89.0% and 90.9% of the numbers of individuals at 9 m and 18 m, respectively.

TABLE 4. Abundance per m^2 of benthic invertebrates collected by Wisconsin grab in Lake Coleridge by depth and location, 1967-68.

Taxon	Water depth (m)	Distance from Harper Diversion (km) <1	1-2	2-5	5-12	>12
No. of samples	9	11	13	25	70	40
	18	8	28	25	70	40
Oligochaeta	9	600	1700	930	1400	1000
	18	630	780	630	1200	2200
Mollusca	9	*12000	5400	4900	7500	5000
	18	4000	2200	4700	3800	4300
Trichoptera	9	*160	170	34	260	130
	18	90	97	56	120	290
Diptera	9	450	940	480	1100	870
	18	680	590	470	830	1300
Mean abundance these taxa	9	*13000	8300	6400	10000	7000
	18	5400	3700	5800	5900	8000

* = significantly higher (0.05) than samples collected at this depth at distances of 1 to 5 km using Kruskal-Wallis test.

TABLE 5. Mean density (per/m²) and relative abundance of benthic invertebrates collected from two depths, Lake Coleridge, February 1988.

Taxon	9.1 m (45 samples) Mean density	9.1 m (45 samples) %	18.2 m (40 samples) Mean density	18.2 m (40 samples) %
Platyhelminthes	2.2	+	*	+
Oligochaeta	453.2	6.8	550.1	16.6
Hirudinea	-	-	*	+
Nematoda	39.2	0.6	16.4	0.5
Mollusca				
Potamopyrgus antipodarum	4343.2	65.4	1674.3	50.4
Lymnaea tomentosa	100.4	1.5	19.1	0.6
Physastra variabilis	46.4	0.7	16.9	0.5
Sphaerium novaezelandiae	143.0	2.2	26.4	0.8
Hydridella menziesi	5.5	+	14.0	0.4
Crustacea				
Cladocera	-	-	3.8	0.1
Copepoda	83.2	1.3	113.2	3.4
Ostracoda	105.0	1.6	90.1	2.7
Ephemeroptera				
Deleatidium sp.	-	-	*	+
Plecoptera				
Zelandobius sp.	*	+	-	-
Odonata				
Procordulia grayi	2.1	+	*	+
Trichoptera				
Paroxyethira sp.	30.8	0.5	22.1	0.7
Oxyethira albiceps	*	+	-	-
Hudsonema amabilis	7.4	0.1	9.7	0.3
Oecetis unicolor	32.3	0.5	37.1	1.1
Olinga feredayi	-	-	*	+
Coleoptera				
Hydora sp.	*	+	-	-
Diptera				
Chironomidae	1244.3	18.7	730.6	22.0
Tipulidae sp.	*	+	-	-
Eriopterini sp.	-	-	*	+
Molophilus sp.	2.4	+	*	+
Total	6638		3324	

+ = <0.1%.
* = < 1 animal/m².
- = not present.

TABLE 6. Mean densities (per m²) ± 1 s.e. (in parentheses) of major taxonomic groups at 9 m and 18 m depth and increasing distance from the Wilberforce Diversion inflow, February 1988.

Taxon	Water depth (m)	Distance (km) from diversion inflow				
		0	0.3	3.4	8.5	19.2
No. of samples	9	5	10	10	10	10
	18		10	10	10	10
Oligochaeta	9	910* (221)	581 (388)	317 (79)	204 (100)	339 (119)
	18		173 (39)	402 (210)	496 (188)	974 (330)
Gastropoda	9	5112 (709)	5004 (2426)	3743 (1516)	5436 (1364)	2039 (469)
	18		2297 (613)	1048 (329)	2047 (378)	2404 (883)
Bivalvia	9	0	147 (40)	135 (55)	43 (37)	296 (171)
	18		27 (31)	06 (02)	44 (10)	73 (17)
Crustacea	9	164 (66)	123 (53)	52 (27)	46 (30)	504 (291)
	18		188 (132)	62 (22)	96 (26)	424 (258)
Trichoptera	9	38 (08)	45 (15)	91 (23)	103 (27)	74 (31)
	18		45 (13)	51 (16)	89 (20)	94 (21)
Diptera	9	558 (14.5)	1346 (258)	1389 (309)	1226 (200)	975 (376)
	18		583 (203)	379 (59)	881 (175)	875 (175)

* = significantly different ($P < 0.05$) than samples collected at this depth at distances > = 0.3 km from the Wilberforce Diversion using t-test.

There was no consistent pattern of changes in the taxonomic composition of the macroinvertebrate fauna with increasing distance from the Wilberforce Diversion (Tables 7 and 8). At both 9 m and 18 m deep, calculation of Jaccard coefficients showed that the composition of the fauna was similar along all four transects (Table 9). Similarly, percentage similarity coefficients showed that there was little change in the relative abundance of taxa with increasing distance from the Wilberforce Diversion (Table 10). There was a significant ($P<0.05$) increase in the density of Oligochaeta, and reductions in Diptera larvae, between samples collected at 9 m depth near the mouth of the Wilberforce Diversion and those collected >0.3 km from the mouth (Table 6). Bivalves were absent from the samples collected at the mouth.

4.1.3 Comparison of benthos collected in 1968 with that collected in 1988

At a depth of 9 m, the Jaccard coefficient showed that the samples collected in 1968 and 1988 had at least 56% of taxa in common. However, the percentage similarity coefficient of 89.6% showed that the communities were more closely related. Similar results were obtained when the benthic communities at 18 m were compared, with Jaccard and percentage similarity coefficients of 54% and 88.1% respectively.

The mean abundance of benthic invertebrates found in 1988 at depths of 9 and 18 m, was 74% and 55% of that found in 1967-68 (Tables 3 and 5). All the major groups declined in abundance and some species, such as *Ferrissia neozelanica*, were absent in 1988.

4.2 Common bullies

4.2.1 Distribution and abundance

Common bullies are abundant in Lake Coleridge. The Wisconsin grab samples taken in 1967-68 yielded 5-8 fish/m². In February 1988, densities of 2.9 and 8.3 fish/m² were recorded using diver counts and the benthic sledge respectively. High numbers of bullies were also seen during the manta board survey in March 1989. Adult fish were seen foraging on the surface of the macrophytes, and juveniles of various sizes were seen in the water column, although they were more abundant in the first metre above the macrophyte beds.

Many bullies were also caught when beach seining in 1988 and 1989 (Table 2).

TABLE 7. Relative abundance (%) of benthic invertebrates collected from 9 m depth at five distances from the Wilberforce Diversion inflow, Lake Coleridge, February 1988.

	Distance (km) from diversion inflow				
Taxon	0	0.3	3.4	8.5	19.2
Platyhelminthes	-	-	-	0.1	*
Oligochaeta	13.4	8.0	5.5	2.9	7.9
Nematoda	-	*	0.3	0.8	2.1
Mollusca					
Potamopyrgus antipodarum	73.7	68.6	64.0	75.2	38.7
Lymnaea sp.	*	0.1	0.4	0.4	8.3
Physastra sp.	1.6	0.2	1.0	0.7	0.2
Sphaerium novaezelandiae	-	2.0	2.1	1.0	6.6
Hydrilla menziesi	-	-	0.2	-	0.2
Crustacea					
Copepoda	1.9	1.5	0.5	0.6	2.5
Ostracoda	0.5	0.2	0.4	-	9.0
Plecoptera					
Zelandobius sp.	-	-	-	-	*
Odonata					
Procordulia grayi	-	-	*	-	0.2
Trichoptera					
Paroxyethira sp.	0.3	0.2	0.3	0.7	0.8
Oxyethira albiceps	-	*	-	-	-
Hudsonema amabilis	*	*	0.1	0.3	*
Oecetis unicolor	0.2	0.4	0.8	0.4	0.9
Coleoptera					
Hydora sp.	-	*	-	-	-
Diptera					
Chironomidae	8.2	18.4	24.3	17.2	22.6
Molophilus sp.	-	0.1	-	-	-

- = not recorded.
* = <0.1%.

4.2.2 Length frequencies

The length-frequency distributions of common bullies recorded in Lake Coleridge from February 1988 to July 1989 are shown in Table 11.

In February 1988, juvenile bullies averaged 33 mm in length. Two older cohorts with average lengths of 51 and 75 mm, were also present. In November 1988, a wide range of intermediate-sized fish was caught, probably consisting of two cohorts with average lengths about 45 and 60 mm. In March 1989, two cohorts of fry were caught, with average lengths of 17 and 32 mm, while in July only one size class of fry was caught, averaging 36 mm in length.

It was impossible to separate large fish into age classes from a study of their length frequencies as two, or possibly more, spawnings occurred each year and size distributions merged with increasing age.

TABLE 8. Relative abundance (%) of benthic invertebrates collected from 18 m depth at four distances from the Wilberforce Diversion inflow, Lake Coleridge, February 1988.

	Distance (km) from diversion inflow			
Taxon	0.3	3.4	8.5	19.2
Platyhelminthes	-	-	-	*
Oligochaeta	5.2	20.5	13.6	20.0
Hirudinea	-	-	*	-
Nematoda	0.4	0.6	0.1	0.7
Mollusca				
Potamopyrgus antipodarum	68.8	50.6	55.3	48.4
Lymnaea sp.	*	1.8	-	0.7
Physastra sp.	0.2	1.0	0.7	0.2
Sphaerium novaezelandiae	0.7	0.1	0.5	1.1
Hydrilla menziesi	*	0.2	0.7	0.4
Crustacea				
Cladocera	0.2	-	-	0.1
Copepoda	5.3	2.9	2.5	2.0
Ostracoda	0.1	0.2	0.2	6.5
Ephemeroptera				
Deleatidium sp.	*	-	-	-
Odonata				
Procordulia grayi	-	-	*	-
Trichoptera				
Paroxyethira	0.5	0.8	0.8	0.6
Hudsonema amabilis	0.1	0.2	0.4	0.3
Oecetis unicolor	0.7	1.6	1.3	1.0
Olinga feredayi	*	-	-	-
Diptera				
Chironomidae	17.5	19.3	24.0	17.9
Eriopterini sp.	*	-	-	-
Molophilus sp.	-	-	*	-

- = not recorded.
* = <0.1%.

4.2.3 Diet

Stomach contents analyses showed that bully fry (under 20 mm long) fed almost solely on crustacean plankton (Table 12). Intermediate sized fish (20-60 mm) ate plankton, chironomid larvae and snails, while larger fish ate chironomids, snails, bully fry and mayfly nymphs. The changes in the percentage composition of the diet with size were statistically significant ($P<0.01$, Kendall's coefficient of rank correlation, Sokal and Rohlf 1981). The sample sizes were too small to detect seasonal changes in diet.

The snail most frequently recorded in the diet was *Potamopyrgus* sp. although *Lymnaea* sp. were found in large bullies, in February and November 1988.

TABLE 9. Jaccard coefficients (J) comparing the benthic faunas at 9 m and 18 m with increasing distance from the Wilberforce Diversion, Lake Coleridge, February 1988.

	9 m depth				
Distance (km)	0	0.3	3.4	8.5	19.2
0					
0.3	0.63				
3.4	0.71	0.67			
8.5	0.69	0.65	0.73		
19.2	0.63	0.60	0.88	0.75	
	18 m depth				
0.3					
3.4		0.76			
8.5		0.60	0.75		
19.2		0.78	0.87	0.67	

TABLE 10. Percent similarity coefficients (PSC) comparing the community structures of the benthos at 9 m and 18 m with increasing distance from the Wilberforce Diversion, Lake Coleridge, February 1988.

	9 m depth				
Distance (km)	0	0.3	3.4	8.5	19.2
0					
0.3	87.5				
3.4	80.3	91.7			
8.5	86.6	91.3	87.7		
19.2	58.0	69.8	72.6	67.3	
	18 m depth				
0.3					
3.4		78.6			
8.5		82.8	89.7		
19.2		76.1	92.3	85.3	

TABLE 11. Length frequency distribution of common bully and koaro in Lake Coleridge, February 1988 - July 1989.

Length categories (mm)	Common bully				Koaro
	Feb. 1988	Nov. 1988	Mar. 1989	Jul. 1989	Feb. 1988 Nov. 1988
No. of fish	242	153	155	82	35
0					
5					
10			1		
15			28		
20			5		
25	6		14	2	
30	29	4	30	35	
35	22	8	12	48	
40	6	14	1	11	
45	12	16		1	1
50	16	22	1	2	11
55	4	12			54
60	3	5	3		14
65	1	9	2		6
70	1	4	1		
75	1	4			
80			1		
85					6
90			1		
95		1	1		6
100		1			

4.3 Upland bully

Upland bullies were rarer than common bullies. No upland bullies were collected in the benthic sledge in February 1988. Eleven upland bullies were caught in seine hauls at the Ryton River mouth in November 1988, and one was collected from the same location in July 1989. These fish were between 42 and 64 mm long (Table 13).

The limited data available (Table 14) indicate that upland bullies ate mainly stonefly larvae, worms and uncased caddisfly larvae.

TABLE 12. Percentage composition of diet of common bullies (by number of food organisms in 1988, by volume in 1989.)

Date	Feb. 1988			Nov. 1988			Mar. 1989		Jul. 1989
Size range (mm)	21-42	43-63	>63	30-44	45-58	59-100	17-19	35-98	27-50
No. of fish	14	31	6	9	16	8	5	13	9
Oligochaeta					11	4			
Nematoda	3	16	9						6
Mollusca[1]	11	25	34		2	4		67	58
Crustacea[2]	47	23		5			98		14
Trichoptera[3]	2	2	10		1	17	2	2	17
Plecoptera[4]				20	15	2			
Ephemeroptera				5	38	70		4	
Diptera[5]	37	33	47	70	32				6
Others[6]						2		27	
Mean number of items	8	16	24	5	3	6	-	-	-

[1] = mainly *Potamopyrgus* sp.
[2] = Cladocera and Ostracoda.
[3] = *Paroxyethira* sp. and *Oxyethira albiceps* larvae.
[4] = *Zealandobius* sp.
[5] = Chironomidae larvae.
[6] = Fish, wasp, unidentified material.

4.4 Koaro

Koaro are widely distributed around the lake and were caught in the benthic sledge in February 1988, and in seines in November 1988 and March 1989 (Table 13). During the manta board survey in March 1989, a small shoal of large (150-200 mm) adult galaxiids (presumably koaro) was seen amongst macrophytes.

There appeared to be two year classes present in the seine catches (Table 11), juveniles ranged from 45 mm to 70 mm in length and adults from 85 mm to 100 mm.

Nearly half (44%) of the food items in the stomachs of koaro were mayfly larvae, with the balance being made up from stonefly larvae, chironomid larvae, caddisfly larvae, worms and adult Empidae (Table 15).

4.5 Chinook salmon

4.5.1 Catch of salmon in nets

Only 10 chinook salmon were caught in gillnets during this study (Table 2) and catch rates were very low (<0.4 fish/100 m net/day, Tables 16 and 17). Most salmon were caught in floating nets set 100 m or more offshore from the mouth of the Harper Diversion (Table 16, Fig. 2). The salmon were either two-year-olds (referred to as 2+) or three-year-olds (3+). They ranged in length from 360 to 510 mm and in weight from 700 to 1300 g (Tables 13 and 18) and their length-weight relationship is shown in Table 19.

Although an additional 58 juvenile (i.e. 0+) salmon were caught using seines, most of these were taken from near the Ryton mouth in November 1988 and catch rates at other localities and times were very low (0.1 to 0.3 per seine haul). The salmon caught in seines included fry and yearling (i.e. 1+) fish, which ranged in length from 30 to 250 mm and from 0.3 to 120 g in weight (Table 13).

4.5.2 Life history and movements

Most salmon probably enter the lake as fry (Flain 1986, Unwin 1976) while the remainder rear in the tributary streams before entering the lake (Flain 1986). Many adult fish had "intermediate" type scales with checks when about eight to ten circuli had formed. These fish probably migrated to the lake at ages varying from two to eight months (Stokell 1934). A few fish had typical "stream type" scales with about 16 narrow circuli and

TABLE 13. Length (mm), weight (g) and condition factor of the fish caught in gill and seine nets.

Species	Measurement		Feb. 1988 Gillnet	Nov. 1988 Seine	Mar. 1988 Seine	Jul. 1989 Seine	Aug. 1989 Gillnet
Chinook salmon	No. of fish		8	47	9	1	2
	Length	Minimum	417	30	95	-	365
		Mean	458	156	123	89	380
		Maximum	506	249	149	-	400
		s.d.	31	64	19	-	-
	Weight	Minimum	710	0.35	-	-	-
		Mean	1020	55	-	7.5	-
		Maximum	1280	122	-	-	-
		s.d.	184	33	-	-	-
	Condition factor	Minimum	98	79	-	-	-
		Mean	105	109	-	-	-
		Maximum	114	160	-	-	-
		s.d.	6	18	-	-	-
Rainbow trout	No. of fish		173	31	62	9	30
	Length	Minimum	189	45	66	123	250
		Mean	392	99	179	207	380
		Maximum	590	161	350	316	500
		s.d.	69	29	58	58	69
	Weight	Minimum	100	1	-	24	-
		Mean	794	16	-	144	-
		Maximum	2420	78	-	448	-
		s.d.	425	16	-	127	-
	Condition factor	Minimum	66	85	-	119	-
		Mean	118	132	-	130	-
		Maximum	156	187	-	145	-
		s.d.	14	20	-	9	-
Brown trout	No. of fish		57	13	5	3	4
	Length	Minimum	239	118	161	169	250
		Mean	450	160	225	220	350
		Maximum	628	265	348	275	400
		s.d.	110	42	84	53	71
	Weight	Minimum	180	20	-	57	-
		Mean	1237	62	-	154	-
		Maximum	2550	220	-	289	-
		s.d.	663	55	-	120	-
	Condition factor	Minimum	85	79	-	-	-
		Mean	121	105	-	-	-
		Maximum	158	127	-	-	-
		s.d.	15	140	-	-	-
Common bullies	No. of fish		242	10	155	82	0
	Length	Minimum	22	30	13	29	-
		Mean	41	52	31	36	-
		Maximum	81	100	97	50	-
		s.d.	11	12	15	4	-
Upland bullies	No. of fish		0	11	0	1	0
	Length	Minimum	-	54	-	-	-
		Mean	-	60	-	42	-
		Maximum	-	64	-	-	-
		s.d.	-	3	-	-	-
Koaro	No. of fish		21	14	45	0	0
	Length	Minimum	52	48	-	-	-
		Mean	61	61	-	-	-
		Maximum	95	95	-	-	-
		s.d.	10	13.	-	-	-

TABLE 14. Stomach contents of seven upland bullies from the Ryton River mouth, in November 1988.

Diet item	No. of diet items	% of total diet items
Oligochaetes	4	17
Lymnea	1	4
Ephemeroptera larvae	2	9
Plecoptera (*Zelandobius* sp.)	8	39
Uncased caddisfly	3	13
Chironomid larvae	2	9
Chironomid pupae	2	9

TABLE 15. Stomach contents of 12 koaro from the Ryton River mouth in November 1988.

Diet item	No. of diet items	% of total diet items
Oligochaetes	2	6
Ephemeroptera larvae	15	44
Plecoptera (*Zelandobius* sp.)	6	17
Trichoptera		
Sandy cased caddisfly	2	6
Uncased caddisfly	2	6
Empidae (adults)	2	6
Chironomid larvae	5	14

remained resident in the tributary streams for one year before migrating.

In the early 1980s, the NCAS was concerned about the possible losses of salmon from the lake either through the Lake Coleridge Power Station or through the Wilberforce Diversion. To determine these losses 10 000 tagged salmon were released into the lake in June 1984 and 5000 in July 1988. Anglers caught 1.5% and 0.2% of these fish, all from within the lake - none was reported from any other location. This indicates that few, if any, hatchery-reared fish leave the lake, and that either few are caught or few survive to maturity.

4.5.3 Diet of salmon

The diet of juvenile salmon appears to vary seasonally (Table 20). In November 1988, stomach contents analysis showed that juvenile salmon ate benthic invertebrates (mainly *Deleatidium* nymphs, caddisfly and chironomid larvae and pupae) and adult insects of terrestrial and aquatic origin (mainly Empidae, Hemiptera) (Table 20). By contrast, juvenile salmon caught in March 1989 had eaten mainly bully fry (Schoener index 0.08).

On a volume basis, fish were the most important food of adult salmon (Table 20). Salmon gillnetted in February 1988, had eaten a fingerling brown trout and koaro, terrestrial insects (brown beetles, wasps and a moth), and adult aquatic insects (craneflies). In November 1988, adult salmon caught by anglers had eaten bullies, terrestrial insects and a dragonfly nymph, In late November an angler caught a 3200 g salmon which contained a 150-180 mm long salmon in its stomach (Webb 1989). In November 1989, salmon caught by anglers at the mouth of the Harper Diversion contained a single juvenile salmon 115 mm long, mayfly nymphs, chironomid larvae and adult beetles. Salmon taken from the Ryton Bay area contained about 10 bullies, and various benthic and terrestrial invertebrates.

In November 1988, 80% of the adult salmon caught by anglers had empty stomachs. Although some of the fish may have regurgitated their stomach contents, the contracted appearance of the stomachs and their empty intestines indicated that most fish were not eating. The average weight of food present was only 7% of that found in February 1988.

4.5.4 Growth rates

Although scale reading has been used for many years to age salmon and trout in Lake Coleridge, its accuracy has not been seriously questioned or tested until recently. In retrospect this was a serious error. In 1934, Stokell showed that landlocked Atlantic Salmon (*Salmo salar*) of known age from Lake Coleridge could not be reliably aged from their scales because they grew rapidly and had no or poorly defined winter bands on their scales. He found some salmon caught in midsummer had wide circuli on the margins of their scales while others had checks of varying widths. More recently, both Webb (1988) and Jellyman (1991) showed that scale reading underestimated the true age of large brown trout in Lake Coleridge.

We found that chinook salmon scales, especially from hatchery-reared salmon, were hard to read and interpret. For example, scales from 211 salmon caught by anglers in November 1989 were independently aged by M. Flain and D.H. Lucas at the FFC and by B.F. Webb at the NCAS (Webb 1989). About 35% of the

TABLE 16. Numbers of salmon and trout caught in gill nets by locality, February 1988.

Netting location	Transect code	No. of nets	Type*	Hours set	Catch Chinook salmon	Rainbow trout	Brown trout
Inshore	1/1	4	F	3	0	0	0
Inshore	1/1	4	S	30	0	18	11
Inshore	1/2	4	S	30	0	78	20
Inshore	1/3	5	F	40	0	32	2
Offshore	1/4	2	F	40	5	19	0
Inshore	1/5	4	S	41	0	13	10
Offshore	2/1	4	S	23	0	1	1
Offshore	2/2	4	S	42	0	0	0
Offshore	2/3	8	F	46	1	6	0
Inshore	3/1	4	S	35	1	4	7
Inshore	3/5	4	S	47	1	2	8
Inshore	4/5	4	S	19	0	3	5
Total		51		396	8	176	64

*F = floating; S = sinking nets.

TABLE 17. Catch rates (fish per 100 m of net day) of salmon and trout in gill nets by locality, February 1988.

Transect code	Catch rates CS	RT	BT	Mean km from diversions	Depth (m) water	net
1/1	*	7	4	0.7	15	5
1/2	*	31	8	0.5	10	10
1/3	*	8	1	0.5	10	10
1/4	3	11	*	0.8	20	2
1/5	*	4	3	1.1	20	20
2/1	*	1	1	0.9	70	70
2/2	*	*	*	0.9	90	90
2/3	*	1	*	1.1	110	2
3/1	*	1	2	2.1	25	25
3/5	*	1	2	1.9	20	20
4/5	*		20	2.7	20	20
Inshore	0.1	7.1	3.0	1.4	18	16
Offshore	0.4	1.8	0.1	0.9	72	41

CS = chinook salmon.
RT = rainbow trout.
BT = brown trout.
* = <0.5 fish per 100 m of net per day (2 nets each 50 m long).

TABLE 18. Length-frequency distribution of salmon and trout caught in gill and seine nets.

Length categories (mm)	Chinook salmon gill nets	seine nets	Rainbow trout gill nets	seine nets	Brown trout gill nets	seine nets
0-		8		3		
50		4		19		
100		10		20		6
150		25	1	41		10
200		10	3	1	2	1
250			6	3	7	3
300			28	1	5	1
350	1		88		4	
400	5		36		9	
450	3		14		10	
500	1		13		15	
550			6		6	
600			3			
Total	10	57	195	102	61	21

TABLE 19. Length-weight relationships (weight g = a length ^{b} mm).

Species	Date	Preservation	No. of fish	a (*10^{-5})	b
Chinook salmon	Feb. 1988	fresh	8	5.64	2.726
	Nov. 1988	frozen	47	1.96	2.878
Rainbow trout	Feb. 1988	fresh	145	5.40	2.744
	Nov. 1988	frozen	31	1.04	3.050
	July. 1989	formalin	9	0.86	3.077
Brown trout	Feb. 1988	fresh	57	5.29	2.756
	Nov. 1988	frozen	13	1.60	2.953

110 scales aged by FFC staff were aged as 1+ years old, compared to 0.5% (one fish) of the 211 scales examined by Webb. The scales aged as 1+ by the FFC had features characteristic of hatchery-reared fish and it was initially thought they were from the May 1989 release. However, this now seems very unlikely because 80% of May 1989 fish were adipose clipped but none of the fish aged as 1+ were. Also, they would have grown from 170 mm to 425 mm (60 to 800 g), in the six months since release! This growth is much faster than the many clipped and nose tagged of this release (mean length 423 mm) caught in November 1990. The fish aged as 1+ by FFC were probably two-year old and came from the 21 000 hatchery-reared fish released in July 1988.

The introduction of hatchery-reared fish into the lake during this study also complicated the situation as the scales of these fish may be similar to those of wild fish migrating as fry into the lake.

We concluded that it was difficult to be certain of the age and growth rates of salmon derived from scale analysis and back calculations and only a few of the results from fish caught in nets have been included here (Tables 21 and 22).

Growth rates of salmon were therefore derived from length-frequency analyses and from the returns of nose tagged hatchery-reared fish.

Juvenile (0+) salmon electric fished from the Hennah Stream averaged 38 mm in October 1954 and grew to 82 mm in April 1955 (Flain 1986). Juvenile salmon caught in seines in Ryton Bay averaged 40 mm in November 1988 and 123 mm in March 1989 (Table 13).

There is little information on the length of 1+ salmon. Few were present in the tributaries and yearling salmon seined in Ryton Bay in November 1988 averaged 189 mm. Most were probably hatchery-reared fish, released in June 1988, as some were nose tagged and others had "hatchery" type scales. Six fish had checks on their

TABLE 20. Diet of chinook salmon in Lake Coleridge.

					% composition of diet by volume			
					Fish	Invertebrates:		
Date	Location	n	Length range (mm)	Mean wet weight of food (g)		benthic	surface terrestrial	surface benthos
Juveniles								
Nov. 1988	Ryton	8	30-100	0.07	0	65	9	26
Nov. 1988	Ryton	37	100-250	0.57	1	64	15	20
Mar. 1989	Ryton	9	90-150	0.61	93	1	0	6
Jul. 1989	Ryton	1	90	0.02	0	100	0	0
Total		57		0.49	19	51	12	17
Adults								
Feb. 1988	Harper	8	440-510	2.68	57	0	32	11
Nov. 1988	Ryton/Harper	25	350-550	0.18	53	38	8	0
Nov. 1989	Ryton	5	350-550	1.22	38	1	55	5
Nov. 1989	Harper	28	350-550	0.93	34	59	5	2
Total		64		0.91	44	29	20	6

n = all stomachs including empty stomachs.

TABLE 21. Mean fork length when captured and back-calculated lengths at earlier ages for brown and rainbow trout and chinook salmon gillnetted in Lake Coleridge, February 1988.

			Fork length (mm) at capture		Fork length (mm) at earlier age											
					1		2		3		4		5		6	
Year class	Age	n	mean	s.d.	mean	s.d.	mean	s.d.	mean	s.d.	mean	s.d.	mean	s.d.	mean	s.d.
Chinook salmon																
1984	3+	4	479	28	87*	12	271	32	392	22	-	-	-	-	-	-
1985	2+	4	437	16	88*	4	219	20	-	-	-	-	-	-	-	-
Rainbow trout																
1981	6+	1	551	-	89	-	179	-	330	-	409	-	487	-	527	-
1982	5+	1	581	-	99	-	234	-	455	-	523	-	562	-	-	-
1983	4+	25	485	68	95	15	193	36	330	52	430	64		-	-	-
1984	3+	33	390	38	88	18	196	33	312	37	-	-	-	-	-	-
1985	2+	21	340	32	109	26	231	37	-	-	-	-	-	-	-	-
1986	1+	5	240	39	103	16	-	-	-	-	-	-	-	-	-	-
Brown trout																
1982	5+	1	569	-	78	-	180	-	322	-	489	-	549	-	-	-
1983	4+	15	518	35	97	17	211	37	385	41	480	39	-	-	-	-
1984	3+	18	444	57	97	20	217	37	383	62	-	-	-	-	-	-
1985	2+	10	285	48	78	17	169	31	-	-	-	-	-	-	-	-
1986	1+	1	258	-	107	-	-	-	-	-	-	-	-	-	-	-

* = possibly an intermediate type check in which case the 3+ fish are probably 2+ and the 2+ fish may have an annulus missing.

TABLE 22. Mean fork length when captured and back calculated lengths at earlier ages for chinook salmon and brown and rainbow trout caught in seine nets in Lake Coleridge.

				Fork length (mm) at capture		Fork length (mm) at earlier age			
						1		2	
Year class	Age	Date of capture	n	mean	s.d.	mean	s.d.	mean	s.d.
Chinook salmon									
1987	1+	1.88	35	189	20	*	-	Hatchery fish - no annulus	
	1+	11.88	1	128	-	89	-	wild	
1988	0+	11.88	10	40	9	96	10	-	-
	0+	03.89	9	123	19				
	0+	07.89	1	89	-				
Rainbow trout									
1987	1+	11.88	24	20	80	15	-	-	-
1988	0+	11.88	4	48	3	-	-	-	-
Brown trout									
1986	2+	11.88	5	198	39	84	11	-173	23
1987	1+	11.88	8	136	19	96	10	-	-

scales characteristic of intermediate type wild fish and averaged 163 mm (s.d. = 31, range 131-198). However, yearling fish caught by anglers in November were much larger (200 - 300 mm) than these seined fish (Flain 1986, Webb 1988, 1989).

Nose-tagged hatchery-reared fish grew rapidly after their release into the lake. Yearling fish released in June 1984 at about 140 mm were caught by anglers in November 1985 and averaged 459 mm in length (n = 31, s.d. = 58, range 320-590). Similarly, triploid salmon released in May 1989, at an average length of about 165 mm, were caught in November 1990 when they averaged 423 mm in length (n = 24, s.d. = 19, range 390-470).

Overall, the data show that in November,(1+, 2+ and 3+ salmon averaged between 200 and 300 mm, and 440 mm and 500 mm respectively. (It is believed the 1+ fish seine-netted in November 1988 were atypical and smaller than average.)

The mean length of 2+ salmon captured during the opening weekend of the fishing season (Fig. 3) can be used as an index of historical changes in growth rates. Before the diversion of the Wilberforce River in 1977, mean lengths averaged 449 mm (Fig. 3) while afterwards they averaged 437 mm and were not statistically different at the 5% level (Table 23). There is a slight tendency for mean lengths to decline over time at a rate of about 8 mm every 10 years (n = 18, r = 0.43, P = 0.071). The trend is small and not statistically significant at the 5% level. Fish have grown particularly slowly in recent years. During 1978-85, angler caught, 2+ salmon averaged 450 mm. During 1986-89 this declined to 421 mm.

4.5.5 Salmon abundance

The abundance of salmon can be roughly assessed from the manta board survey (Flain 1989). Twelve salmon were counted over a distance of about 8 km. If it is assumed the divers had a visual radius of 3.5 m then about 5000 salmon might have been present in the surface layers (0-20 m) of the lake. We decided that further surveys were unwarranted because we could not determine the numbers of salmon present at depth.

Attempts to assess salmon abundance by mark recapture methods were not successful. In July 1988, 20 900 yearling salmon were released into Lake Coleridge of which 5900 went into Ryton Bay. Four months later, 13% of the 1+ salmon caught here had been tagged and scale studies indicated that most (86%) of the 1+ salmon were hatchery fish from this release. Therefore abundance estimates from mark recapture studies are invalid because it seems unlikely the hatchery fish had dispersed randomly around the lake and because survival rates may be different between hatchery and wild fish.

TABLE 23. Size of chinook salmon spawning runs into the Ryton River, anglers' catch rates, and mean length of 2+ salmon before and after the Wilberforce Diversion in 1977.

	Pre-diversion 1951-1977	Post-diversion 1977-1989
Salmon spawning runs		
n	5	7
mean	207	300
s.d.	215	174
Anglers' catch rates per hour		
n	7	10
mean	0.38	0.21*
s.d.	0.11	0.07
Mean length age 2+ (mm)		
n	8	10
mean	449	437
s.d.	19	21

* = $P < 0.01$

4.5.6 Historical trends in abundance and condition factor

Historical data on salmon spawning runs and on anglers' catch rates (Flain 1986, Webb 1984, 1986, 1988, 1989, 1990) (Fig. 3) were examined to determine if they suggested any major change in salmon abundance.

There were wide annual variations in the estimates of the numbers of salmon spawning in the Ryton River and tributaries (Fig. 3). Annual variations are due to differences in survey techniques, weather conditions and effectiveness of trapping operations, as well as changes in the size of spawning runs. Before the Wilberforce Diversion the runs averaged about 200 fish (range 60-600) while post-diversion they have averaged about 300 fish (range 100-600). The difference in the average size of the runs is small and not statistically significant (Table 23).

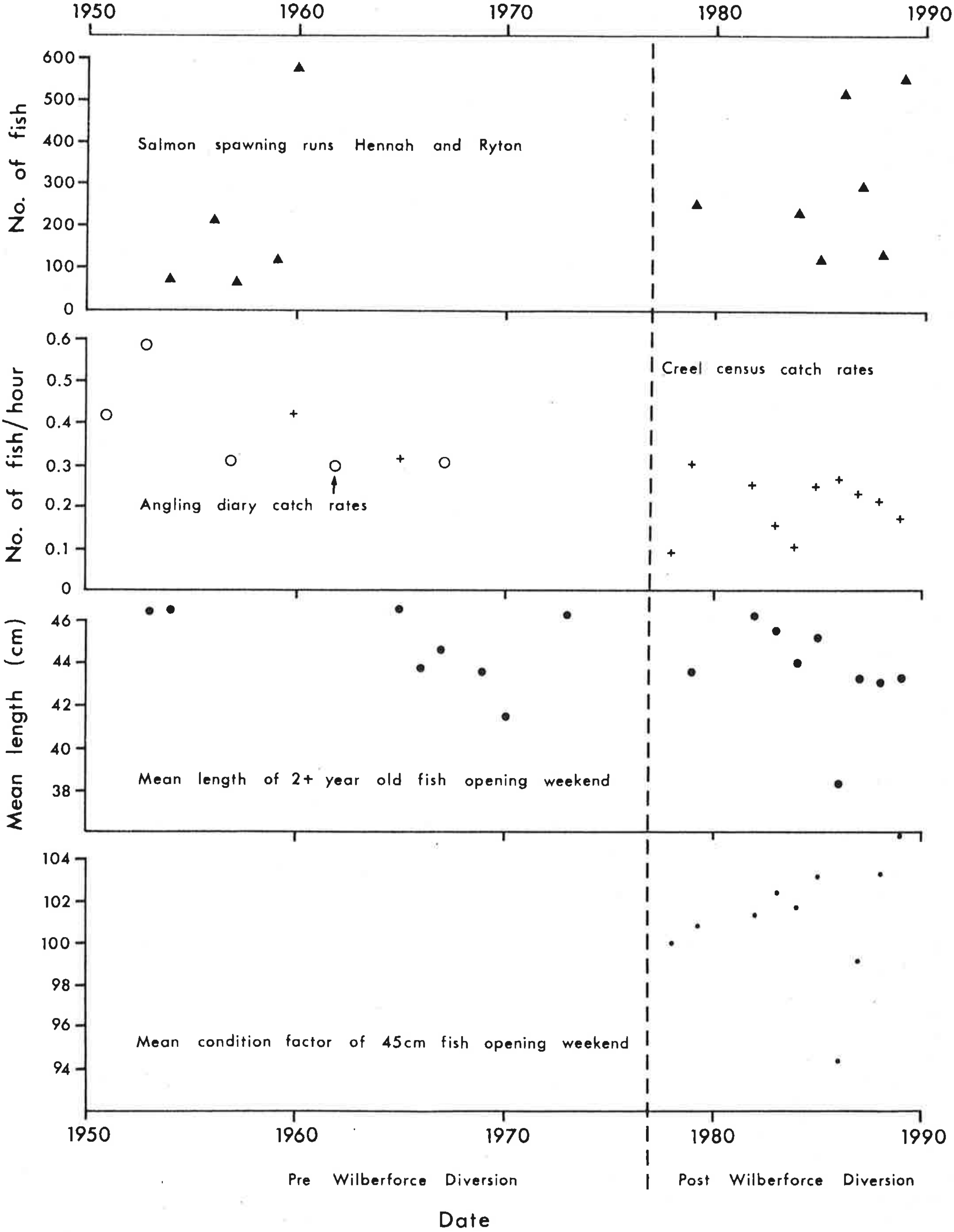

FIGURE 3. Historical fluctuations in quinnat salmon abundance and growth indices.

Anglers' catch rates vary substantially depending upon lake and weather conditions as well as upon fish density. Pre-diversion creel census and angling diary catch rates averaged 0.38 fish per hour (Table 23) and were significantly higher than post-diversion opening weekend creel census catch rates of 0.21 fish per hour (Fig. 3). However, creel census catch rates have remained relatively stable in recent years.

All historical catch rate, spawning run and other relevant information on salmon abundance from 1975 onwards contained in the NCAS Annual Reports was tabulated to indicate the relative abundance of salmon year classes. The 1982 and 1983 year classes appeared to be particularly weak while the 1975 and 1984 year classes were strong. There was some evidence for a three year cycle in abundance from 1975 onwards. The reasons for these differences in abundance have not been investigated because the data are inadequate.

Condition factors were stable over the period 1978 to 1987 (Fig. 3) except for a sharp drop during the 1986 opening weekend and to a lesser extent in 1987. The low condition factors may be an artifact of the regression system used to predict the weight and condition of the fish at 450 mm long. In 1986 the fish averaged only 384 mm long and 98 in condition which is slightly higher than the 94 predicted at 450 mm.

4.6 Rainbow trout

4.6.1 Catch of rainbow trout in nets

Over 200 large rainbow trout were caught in gillnets and another 100 smaller juvenile trout taken in seine nets (Tables 2 and 13). The fish caught in gillnets averaged 390 mm in length, 800 g in weight and were smaller than chinook salmon and brown trout. Condition factors were lower than those for brown trout (Table 13).

Rainbow trout were smaller, but more abundant, in the shallow water at the head of the lake where the diversions entered (Fig. 2) than in the other locations gillnetted. The mean size of trout gillnetted decreased significantly (Table 24) from about 490 mm and 1400 g in the main body of the lake to 360 mm and 600 g at the head of the lake. Gillnet catch rates were higher at the head of the lake and the proportion of rainbow trout in the catches rose significantly from 14% to 80% (Table 24). Most rainbow trout were caught in the littoral zone and few trout were caught in deep water either at the surface or at depth (Fig. 2).

TABLE 24. Correlations between species composition, size, and catch rates of trout in gill nets with water depth (m) and distance (km) from the Wilberforce and Harper Diversions, February 1988.

	n	Depth	Distance	Both factors
% brown trout†	10	0.32	0.77**	0.77**
Rainbow trout				
Length	171	0.14	0.30*	0.30**
Weight	146	0.18*	0.34*	0.35**
Catch rates	11	-0.45	-0.50	0.71
Brown trout				
Length	57	0.10	0.00	0.20
Weight	47	0.09	0.02	0.20
Catch rates	11	-0.57	-0.02	0.57

† = weighted by catch at location.
** = $P < 0.01$.
* = $P < 0.05$.

Juvenile rainbow trout were caught in the seine nets and ranged in length from 45 to 350 mm (Table 13).

4.6.2 Diet

Fingerling rainbow trout, less than 100 mm long ate mainly benthic foods (mayfly nymphs, stonefly, caddisfly and chironomid larvae) and the adult stages of benthic invertebrates (swarm flies) (Table 25).

Small yearling and juvenile fish ranged in length from 100 to 350 mm, and ate benthic invertebrates, juvenile bullies and terrestrial insects (Table 25). Some seasonal differences in diet occurred with insects taken from the water surface being important in February and November and bullies in March and July. Differences in diet by location (Table 25) were small and statistically insignificant.

Medium and large rainbow trout (350-600 mm) were caught in gillnets only during February 1988 (Table 25). They ate benthic invertebrates (snails and caddisfly larvae) and insects from the water surface (beetles, cicadas, craneflies, Hemiptera). There was no significant difference in diet by location and no evidence the Wilberforce or Harper Diversions influenced diet (during February 1988). Their diet was similar to that found by Stokell 1936.

TABLE 25. Length range and numbers of rainbow trout used in stomach contents analysis. Percentages that each food category contributed to total volume.

Date	Location	n	Length range (mm)	Mean wet weight of food (g)	% composition of diet by volume			
							surface foods	
					fish	benthos	terrestrial	benthos
Fingerlings								
Nov. 1988	Ryton	15	40-100	0.10	0	59	0	41
Mar. 1989	Ryton	1	70	0.08	0	25	12	62
Mar. 1989	Harper	6	70-80	0.05	26	19	33	22
Total		24		0.08	4	52	5	39
Small juveniles								
Feb. 1988	Wilberforce/Harper	21	190-350	3.29	2	57	34	7
Feb. 1988	Boat Harbour	6	140-180	3.67	16	13	33	37
Feb. 1988	Inshore	20	230-360	6.53	7	49	41	2
Nov. 1988	Ryton	15	100-160	0.36	0	60	11	29
Mar. 1989	Ryton	14	110-350	2.34	26	71	1	1
Mar. 1989	Harper	11	160-240	0.66	30	48	12	9
Mar. 1989	Intake	3	140-350	1.55	25	70	5	0
Jul. 1989	Harper	6	120-320	3.14	62	38	0	0
Jul. 1989	Ryton	3	200-250	1.22	8	88	1	3
Total		99		2.97	13	51	29	6
Medium sized fish								
Feb. 1988	Wilberforce/Harper	41	350-390	4.42	1	63	31	5
Feb. 1988	Offshore	6	340-390	3.67	25	31	28	16
Total		47		4.33	4	60	31	6
Large fish								
Feb. 1988	Wilberforce/Harper	19	400-560	11.11	7	81	12	0
Feb. 1988	Inshore	19	400-590	12.20	6	73	11	10
Feb. 1988	Offshore	11	410-530	11.65	1	63	18	18
Total		49		11.65	5	74	13	8

4.6.3 Growth

Rainbow trout fry emerge from the redds in about October (Flain 1986) and research in the lower Waitaki River (Graybill and Palmer 1990) indicates that many rapidly move downstream to the lake. In November 1988, fry seined from Lake Coleridge averaged about 48 mm (Table 13) and grew to about 72 mm by March 1989 (n = 7, s.d. = 5). They were similar in size to those electric fished from the Hennah Stream in 1954/55 (Flain 1986).

The winter check at age 1 forms when the trout range in mean length from 80 to 109 mm (Tables 21 and 22) and 1+ fish seined in November 1988 averaged 105 mm (Table 22). Growth over the second summer is rapid and by March 1989, 1+ fish averaged 181 mm (n = 51, s.d. = 26).

Growth then slowed, fish seined in July 1989 averaging 193 mm (n = 8, s.d. = 43) and the second winter check occurred when fish averaged about 204 mm (Table 21). Trout at age three and four years averaged 322 and 430 mm respectively (Table 21). These growth rates from age 1+ onwards were slower than shown by chinook salmon and those previously estimated for rainbow trout (Flain 1986).

4.7 Brown trout

4.7.1 Catch of brown trout in nets

Few brown trout fry or yearling fish were caught in seine nets (Table 2) and they appear to be less abundant than rainbow trout in the shallow littoral zone of Lake Coleridge.

Larger brown trout caught in gillnets in February 1988 ranged in length from 239 to 628 mm with a mean of 450 mm (Table 13). Their weights ranged from 180 to 2550 g with a mean of 1200 g. Condition factors ranged from 85 to 158 with a mean of 121.

In February 1988 there were no significant differences in the size and number of trout caught in gillnets at the head of the lake and those caught up to 2.7 km away (Table 24). Few trout were caught offshore in deep water (Fig. 2).

4.7.2 Diet

Small and medium sized brown trout (100-400 mm in length) mainly ate benthic invertebrates (caddisfly larvae and snails) and fish (juvenile salmonids and bullies) (Table 26). Fish (bullies and koaro) were the most important food of large brown trout (>400 mm). The samples were too small to determine differences in diet by season or location.

4.7.3 Growth

Flain (1986) showed that brown trout fry averaged about 79 mm in April and Tables 21 and 22 show that yearlings averaged just under 100 mm. Two-year-old fish range from about 170 to 220 mm, and 2+ brown trout seined in Ryton Bay in November 1988 averaged 198 mm. Growth rates of older fish then accelerated to

TABLE 26. Length range and numbers of brown trout used in stomach contents analysis.

					% composition of diet by volume			
			Length range	Mean wet weight of			surface foods	
Date	Location	n	(cm)	food (g)	fish	benthos	terrestrial	benthos
Small juveniles								
Feb. 1988	Boat Harbour	2	17-18	0.65	50	48	0	2
Feb. 1988	Wilberforce/Harper	6	24-39	0.58	6	91	0	3
Feb. 1988	Inshore (sides)	5	26-38	1.52	61	38	1	0
Feb. 1988	Inshore (north)	6	24-43	2.66	0	100	0	0
Nov. 1988	Ryton	13	12-26	0.62	19	49	0	32
Mar. 1989	Ryton	3	16-28	1.24	94	3	0	2
Jul. 1989	Harper	3	17-27	0.40	98	2	0	0
Totals		38		1.09	28	65	0	7
Medium and large fish								
Feb. 1988	Wilberforce/Harper	12	44-63	0.30	40	55	3	2
Feb. 1988	Inshore (north)	10	41-55	4.80	61	29	5	5
Feb. 1988	Inshore (sides)	14	52-62	3.76	75	15	9	1
Mar. 1989	Harper	1	35	1.08	100	0	0	0
Total		37		2.84	68	23	7	3

about 170 mm a year and three- and four-year-old fish averaged about 385 mm and 480 mm respectively (Table 21). Growth then averages as low as 17 mm a year in fish up to 14 to 15 years old (Jellyman 1991).

5. POTENTIAL EFFECTS OF THE WILBERFORCE DIVERSION ON LAKE COLERIDGE

The potential beneficial and adverse effects of the Harper and Wilberforce Diversions on the aquatic fauna and flora and on the trout and salmon fishery have been discussed in several submissions and reports (e.g., Bowden *et al.* 1983, Jowett 1984, Mitchell, 1984, Fisheries Research Division 1985, Flain 1986). Potential effects on the fishery could derive from increased and fluctuating lake levels, siltation, eutrophication, changes in benthos and fish movements from the lake.

5.1 Benthic invertebrates

5.1.1 Species present and relative abundance compared with other lakes

The benthic fauna of Lake Coleridge is more diverse and abundant in the littoral than in the profundal zone. The littoral zone, where sufficient light penetrates to support the growth of attached vegetation (Winterbourn and Lewis 1975), supports high densities of benthic invertebrates. Snails and caddisfly larvae graze the surface algae and diatoms attached to vascular macrophytes and characean algae while oligochaete worms, chironomids and bivalves live in the mud. Compared to other lakes around New Zealand (Forsyth 1978, Timms 1982), the littoral zone of Lake Coleridge appears to support exceptionally high densities of snails and average numbers of worms and chironomids. The high estimates of snail densities could be because the Wisconsin grab used by divers in Lake Coleridge collected larger amounts of weed and associated snails than the Ekman grab used by other workers.

The dark, deep-water profundal zone has a bed of soft mud containing mainly worms and low numbers of snails, caddisfly larvae and chironomids. Compared with other lakes (Forsyth 1978, Timms 1982), the profundal zone of Lake Coleridge supports average to low numbers of snails, high numbers of worms and low numbers of chironomids.

Over 25 species of benthic invertebrates were found in Lake Coleridge, which, although higher than the 14 to 21 species found in typical South Island lakes (Timms 1982), could be a result of the high sampling effort (185 samples in 1964 (Cudby *et al.* 1964), 434 samples in 1967/68 and 85 samples in 1988). No sponges, bryozoans or crayfish (*Paranephrops zealandicus*) were collected in the samples although they may have been overlooked or were, and are, present in low numbers.

The apparent absence of crayfish is interesting since they are abundant in the nearby Lake Georgina (Percival and Burnet 1963). Although crayfish can be limited by low minimum dissolved calcium levels, this is probably not the reason for their absence in Coleridge. Coleridge has calcium concentrations of 5.2 to 8.0 g/m^3 (Bowden *et al.* 1983) which are only slightly lower than Lake Georgina and similar to Lake Taupo (White 1983) where crayfish are also abundant. Indeed, one crayfish was caught in Lake Tikitapu which contains no molluscs because of the exceptionally low bicarbonate and calcium levels (0.68 g/m^3) (Forsyth 1978).

5.1.2 Historical changes in benthos

These studies showed that there were no major changes in the species composition of the benthos of Lake Coleridge between the two sampling periods in 1968 and 1988. During both periods the gastropod *Potamopyrgus antipodarum*, oligochaetes and chironomids formed a large proportion of the fauna. Although analyses of species composition, based on presence/absence, indicated that 54-56% of the taxa occurred during both periods, this is likely to be an underestimate because of differences in the level of identification used.

The decrease in overall abundance between 1968 and 1988 is probably due to sampling variability and to natural, seasonal and annual variations in abundance. Analyses of the relative abundance of different species showed a high level of similarity between the benthos collected in 1968 and that collected in 1988. It seems unlikely that increased siltation from the Wilberforce Diversion would have produced the decline noted in all the major species present.

5.1.3 Impacts of the Wilberforce Diversion on benthic invertebrates

To assess the effects of the Wilberforce Diversion, information is needed on the relative influence of ecological factors controlling the abundance of benthic invertebrates such as:

1. The trophic status of the lake and food supplies available for invertebrates.

2. Seasonal changes in dissolved oxygen, temperature and other water quality parameters.

3. Water depth and associated factors such as light penetration, weed beds, substrates and siltation.

4. The extent and frequency of changes in water levels and exposure to wave action.

5. Biological factors such as predation by fish.

There is only a coarse and imprecise relationship between the trophic status of lakes and their standing stocks of benthos (Forsyth 1978, Timms 1982, 1983). It seems unlikely that additional nutrients from the Wilberforce Diversion will have any major effects on abundance or species composition. However, wood and other allochthonous organic material will be washed into the lake by the diversions and will be a source of food to benthic invertebrates such as worms and chironomids (Wetzel 1975, Timms 1980).

Water depth is a major factor influencing the abundance and species composition of the benthos because of associated changes in ecological factors such as light penetration, weed beds, substrates, wave action and turbulence.

Weed beds support many more invertebrates than adjacent bare mud or gravel areas (Kirk and Henriques 1982) and hence the species composition, and distribution, and biomass of weed beds is a crucial ecological factor (Biggs and Malthus 1982). The maximum depth of the euphotic zone (within which plants can grow) is approximately the depth to which 1.3% of the surface light penetrates (Vant *et al.* 1986). Before power development the euphotic zone may have exceeded 43 m, and post Harper Diversion it averaged 39.2 m (Biggs *et al.* 1990) and probably varied little seasonally. Inorganic suspended solids from the Wilberforce Diversion have reduced the euphotic zone to about 30 m and there are now substantial seasonal fluctuations in the euphotic zone depth from 19 to 43 m. The precise effects of the compression of the euphotic zone on the growth of characean algae and hence on the benthos, cannot be estimated without more research on the light requirements of these plants, their distribution in the lake and the invertebrates they support. Nevertheless the reduction in euphotic depth is substantial and is likely to have had adverse effects on the charophytes and benthos.

The benthic fauna is also influenced by the composition of the lake bed. Areas of coarse gravel eroded from the lake shores support fewer burrowing invertebrates than mud rich in organic material (FFC file data collected by Cudby *et al.* 1966). Silt deposited at the head of the lake had only a local effect on invertebrate species composition and abundance (Section 4.1) and on vegetation (Clayton 1984) and had no impact lake wide.

Artificial changes to mean water levels and increased water level fluctuations have adverse impacts on the benthic invertebrate fauna living around the margins of water supply and hydro reservoirs (Hunt and Jones 1972, Winterbourn 1987). Reductions in water level can expose large areas of weed beds and kill benthic invertebrates and fish (pers. obs. in Lakes Benmore and Ruataniwha, Allen and Empson 1989). Also, the increased turbulence and wave action on normally deeply submerged mud substrates increases erosion and water turbidity, damaging weed beds and the benthic fauna (Kirk and Henriques 1982, Mark 1987). Increases in water levels can cause cliffing and serious erosion in unconsolidated sediments (Mark 1987) with adverse impacts on the benthos.

In Lake Coleridge, benthic invertebrates and plants are scarce in shallow water on the wave cut platform, because of wave turbulence and the loose broken angular rocks, stones and shingle present (Clayton 1984, Flain 1986). For example, samples taken in water less than 3 m deep contained only 4% of the numbers of invertebrates collected from water between 3.5-7 m deep (Cudby *et al.* 1966).

Hydro development since 1915 has increased natural water levels by 1.4 m and annual fluctuations by 0.7 m. Detailed studies on changes in the benthos at various sites around the lake margins and through the year would be required to assess the effects of these changes. The reduced water level fluctuations since the Wilberforce River was diverted are probably beneficial but may be counteracted by the recent increase in mean levels. Also, an extreme drawdown of the lake occurred in the winters of 1991 (505.96 m) and 1992 (minimum allowable lake level of 505.43 m). Drawdowns of up to 3.6 m below mean levels will almost certainly damage the benthic fauna and the regularity of large drawdowns could be a matter of concern.

Finally, the huge populations of bullies and the smaller numbers of salmonids and koaro which eat benthic animals, affect invertebrate biomass and productivity

(Wetzel 1975) and may prevent vulnerable species from being established. For example, in Lake Waahi an increase in water turbidity, from about 0.8 m secchi disc visibility to 0.4 m (J. Hayes, pers. comm.), reduced predation by smelt and bullies on the mysid shrimp *Tenagomysis chiltoni* which became more abundant (Hayes and Rutledge 1991). However Lake Coleridge is much clearer than Lake Waahi and it seems unlikely the increase in turbidity due to the Wilberforce Diversion would have made much impact on prey detection distances and hence predation by bullies and other fish.

In conclusion, Lake Coleridge contains a typical variety of invertebrate species, with high numbers of snails present in shallow water and worms in deep water. There is no evidence that siltation at the head of the lake has adversely affected the composition of the fauna lakewide and it is unlikely that turbid inflows have influenced fish predation on benthos. The diversion of the Wilberforce River has probably benefited the benthic fauna by adding organic material and by reducing annual fluctuations in water levels. However, higher mean leak levels and a decrease in the euphotic zone have probably had adverse effects on macrophytes and associated benthic animals.

5.2 Native fish

5.2.1 Common bullies

Little is known about the biology of common bullies in large oligotrophic lakes such as Coleridge (Stephens 1982, 1983) and nothing about factors and processes controlling their abundance,.growth, size and population dynamics. However, they are undoubtedly an important part of the ecosystem and are eaten by salmon and trout.

The Wilberforce Diversion has had no obvious adverse effects on common bullies in the lake. They are abundant and adult fish are large, similar in size to those found in other oligotrophic lakes such as Taupo (Stephens 1983). In Lake Coleridge, adult bullies had an average length of about 52 mm, with the largest (and presumably the oldest) specimens around 80 mm. In contrast, in the eutrophic Lake Waahi, adults averaged about 35 mm, with the largest not exceeding 70 mm long (Stephens 1982). Bullies are also abundant in Lake Benmore (Graynoth *et al.* 1986 and pers. obs.) (which is more turbid and has greater level fluctuations than Lake Coleridge), and in the extremely turbid Lake Ellesmere.

Bullies typically spawn in shallow water on a patch of rough bottom and their nest is made on a boulder, log, weed or other firm surface (Stephens 1983). The male fish keeps the eggs free of silt and it seems unlikely the fine glacial silt entering the lake from the diversions would harm the eggs. The most significant impacts could occur if the lake was drawn down during breeding in spring and summer, exposing nests to the air. However, the lake is normally being filled or is close to its maximum level at this time, so damage to nests appears unlikely.

It is impossible to determine whether the compression of the euphotic zone and possible reduction in weed beds and benthos has had any effect on bully abundance, growth and production. The effects, if any, are probably far less than the impact of predation from the introduced salmonids.

5.2.2 Upland bullies

Upland bullies were rarer than common bullies and were caught only near the Ryton River mouth. They are present in the river and are abundant in the nearby eutrophic Spectacles Lakes (Staples 1975). Staples showed that populations were regulated by changes in water levels, food supplies, competition, predation and parasitism. The reasons for their low abundance in Coleridge are unknown. There are insufficient data to assess the impacts of the Wilberforce Diversion on the stocks.

5.2.3 Koaro

Juvenile and adult koaro are common in the lake and adults are also present in the tributary streams and the Harper Diversion (G. Glova pers. comm.). The effects of the Wilberforce Diversion on koaro cannot be assessed as insufficient is known about their biology and population dynamics in lakes. Large numbers have died on the rare occasions when the Harper Diversion has dried up (Maindonald 1989b) and their food supplies may have been reduced by the increase in silt loads and water turbidity. However, all these changes are probably insignificant compared with the impact of predation by trout and salmon.

5.2.4 Longfinned eels

Only a few eels were caught as a bycatch while seine netting and no serious study was made of these fish in the lake, although they are probably quite common based on observations by divers (M. Flain pers. comm.)

and on studies in other high country lakes (D.J. Jellyman pers. comm.).

It is most unlikely that increases in water turbidity have had any direct impacts on eel stocks as they are commonly found in turbid waters. However, the damming of Lake Stream may have reduced the recruitment of juvenile eels into the lake and many adult eels may die when migrating out of the lake through the power station and turbines, en route to their South Pacific Ocean breeding grounds.

5.3 Salmon and trout

5.3.1 Chinook salmon

Some aspects of the life history and biology of chinook salmon in Lake Coleridge are uncertain because it proved impossible within the constraints of time and finance to capture sufficient fish to study. Nevertheless, some assessment of the impacts of the Wilberforce Diversion on the biology of the fish in this lake can be made based on the results of this study and on studies elsewhere.

5.3.1.1 Distribution

The vertical and horizontal distribution of landlocked chinook salmon has not been studied previously in deep, cold, isothermal lakes such as Coleridge. In Lake Coleridge, anglers catch salmon at the surface as well as by trolling at depths down to 15 m (50 feet). In Lake Michigan, U.S.A., salmon are caught with "downrigger" gear at depths greater than 30 m (Hardy 1971) and salmon at sea are caught at even greater depths. Trawlers off Banks Peninsula most frequently catch salmon at depths from 55 to 75 m with isolated specimens being recorded below 100 m (Unwin *et al.* 1988). Therefore, it is possible that salmon could be found throughout Lake Coleridge down to its maximum depth of 200 m, although it seems more likely they will be found where food is most available, either in the littoral zone, in midwater or near the water surface.

Insufficient salmon were caught to determine seasonal differences in vertical and horizontal distribution in Lake Coleridge. Our lack of success was not totally unexpected as only a few salmon were gillnetted in earlier studies in Lake Coleridge (Cudby *et al.* 1966), and other workers (R.T. Hutchinson pers. comm.) have found it difficult to catch landlocked chinook salmon in gillnets. We suspect the salmon either see and avoid the nets and/or live in the mid water regions of the lake which are difficult to net. The failure to catch salmon in beach seines after November 1988 is probably because older and larger salmon moved offshore. Further studies, possibly using echosounding, radio telemetry or midwater trawling techniques, are needed to determine the distribution of salmon in Lake Coleridge.

Nevertheless, based on these studies and anglers' catches there is no evidence that the diversion of the Wilberforce or Harper Rivers has had any adverse impacts on the distribution of salmon in the lake. Indeed, the converse is true as many anglers fish near the mouths of both diversions and the highest catches of salmon in gillnets were made just off the Harper mouth.

5.3.1.2 Diet

In Australian and North American lakes, small fish such as inanga (Cadwallader and Eden 1981) and smelt (Hoover 1936) are important foods of adult chinook salmon. In New Zealand, in Lake Heron, 0+ salmon (50-120 mm) caught in summer ate mainly *Daphnia* sp, terrestrial insects and benthic invertebrates (FFC file data). In Lake Wakatipu, adult salmon fed on koaro and terrestrial insects but not on bullies (R.T. Hutchinson pers. comm.). The only previous studies in Lake Coleridge were by Cudby *et al.* (1966) who found mainly terrestrial insects (brown beetles), adult flies and dragonflies and bullies in salmon stomachs. Therefore it appears that landlocked chinook salmon in lakes are primarily midwater and surface predators of insects and small fish and that benthic invertebrates are less important.

It has been shown that the Wilberforce and Harper Diversions had little effect on the invertebrates and bullies eaten by salmon, and most salmon were caught just off the Harper Mouth, possibly because insects were washed down the river into the lake or were blown into the lake by the prevailing north-west winds. Although high silt levels during floods might reduce water visibility and hence hinder the capture of food items by salmon this is a localised and infrequent event and of no consequence.

5.3.1.3 Growth rates

Although growth rates, to age 2+, were slower in 1986-89 than in previous years, the Wilberforce Diversion has had no major adverse impacts on the growth rate of salmon. The very small fish caught in November 1986 (384 mm) were the product of the strong 1984 year class and it is possible that growth rates were limited by food supplies. If this is the case then the large hatchery liberations in recent years may also reduce growth rates. Growth remains rapid compared to salmon in Lake Wakatipu (Hutchinson

1981) and is faster than brown and rainbow trout in Coleridge, possibly because salmon may enter the lake at a younger age than trout and feed on bullies at a smaller size. Indeed, the supply of forage fish is probably the crucial factor controlling growth and size of salmon.

5.3.1.4 Abundance

We have little information on the numbers of chinook salmon present in Lake Coleridge. Flain (1986) and Webb (1989) estimated stocks of 4955, 9667 and 34 450 takeable (>250 mm) fish, using mark-recapture techniques. However, these estimates are unreliable because they assume marked, hatchery-reared fish have similar behaviour and survival rates to wild fish. Spawning run counts and the manta board survey indicated that from 600 to 5000 large fish were present. Counting fish using echosounding techniques was not attempted because modern equipment was unavailable and because we suspected it might be impossible to identify salmonid species from the magnitude and location of their echoes. There seems to be no easy way to measure salmon abundance in this lake.

Therefore, we can only rely on indirect indices of abundance such as angler catch rates, which appear to have declined since the 1950s and 1960s. This is probably partly due to a change in data collecting techniques, from the early angling diary schemes, where the returns tended to come from the more skilled and successful anglers, to the later creel census schemes where all types of angler were randomly interviewed. Nevertheless, the decline may also be due to a reduction in stock densities, as some anglers we spoke to are convinced that chinook salmon used to be much easier to catch 20 or so years ago, and claimed limit bags of 10 salmon were not uncommon.

Clearly the data on stock density and historical changes in abundance are inadequate. Perhaps all that can be said is that there is no evidence of any major changes in stock density since the Wilberforce Diversion, although it is possible that some changes may have occurred.

5.3.2 Trout

Brown trout and rainbow trout will probably be unaffected by small increases in nutrient levels and siltation as good trout stocks are found in many New Zealand lakes with much higher nutrient and sediment inputs and lower water clarity than Coleridge (Jowett 1984).

5.3.2.1 Distribution

Rainbow trout were caught more often in gillnets at the head of the lake, near the where the Wilberforce and Harper Diversion entered, than elsewhere. This is possibly because gillnets were more effective on the flat, shallow bed than on the steep littoral further down the lake. However, trout may be more abundant at the head of the lake because of the entry of nutrients and invertebrates washed and blown into the lake.

5.3.2.2 Diet

In the Rotorua lakes, rainbow trout feed in winter on smelt and terrestrial insects (Smith 1959) while in summer, because of high surface water temperatures, they are forced to eat benthic foods such as adult bullies and aquatic insects (Rowe 1984). Lake Coleridge is relatively cold all year round, with a deep thermocline, and this may account for the lack of major seasonal differences in diet.

Brown trout in Lake Coleridge mainly ate fish (bullies), and benthic invertebrates. Insects from the water surface were not normally taken in large numbers, therefore brown trout probably forage close to the lake bed. Their diet in Coleridge is similar to that of brown trout in other South Island lakes (McCarter 1987, Graynoth *et al.* 1986, Pack and Jellyman 1988).

High water turbidity can cause rainbow trout in rivers to feed on benthos instead of invertebrates drifting downstream in midwater (Tippets and Moyle 1978). It is not known by how much water turbidity would have to increase before open water foraging by trout in Lake Coleridge is threatened. However, silt inputs would probably have to increase many fold before predation is affected (Burnet and Wallace 1973).

5.3.2.3 Growth rates

The growth rate of rainbow trout in Lake Coleridge in the late 1980s was compared with the growth rates measured in the early 1950s in Lakes Coleridge and Lyndon (Percival and Burnet 1963). (Examination of the original data, held in FFC files, revealed a drafting error in this paper and 0.5 years should be added to the age of fish in Figures 5 and 6 of Percival and Burnet 1963).

No historical changes were found in the growth rate of fish in Lake Coleridge although there were major differences between the growth rates of fish in Lakes Coleridge and Lyndon. Trout in Coleridge grew more slowly in the first year of life, presumably because many reside in tributaries, unlike the lake resident fish in Lyndon. Growth rates in the second year of life are fairly similar, at about 100 to 120 mm a year, but the major difference occurred in large fish (280-400 mm fork length). In Lake Coleridge these fish continued to grow at more than 100 mm a year while in Lyndon growth slowed markedly to less than 40 mm a year. This may be due to differences in water temperatures, trout density and food, such the species composition and abundance of forage fish in the lakes (common bullies and koaro in Coleridge, upland bullies in Lyndon)

Flain 1986 and Webb 1990 found no evidence of major historical changes in the mean size or length at age of rainbow trout caught in the opening weekend of the fishing season since the Wilberforce Diversion was built in 1977.

Compared to other lakes (Graynoth *et al.* in press) brown trout grew slowly for their first two years and then more rapidly so that at age four and older they were larger than in other lakes. There is insufficient historical information available to assess the effects, if any, of the Wilberforce Diversion on growth rates.

5.3.2.4 Abundance

In recent years an unusually high number of rainbow trout have been caught during the opening weekend (Webb 1990). It is possible the recent modifications to the Harper River Diversion gates and the opening up of the Harper River catchment to trout spawning and rearing may have increased stock numbers (Webb 1989). However, historical trends in rainbow trout abundance cannot be determined because of the lack of adequate data. The annual mixed redd counts conducted in September each year by the Fish and Game Council (Maindonald 1989a) are inconclusive and only indicate that no major changes in spawning runs have occurred in the Ryton River and other tributary streams.

5.4 Conclusions

We conclude that the Wilberforce Diversion has had no discernable major effects on the benthic invertebrates or fish stocks of Lake Coleridge. However, small to moderate changes can be difficult to detect because of natural variability and continued monitoring of the trout and salmon stocks and fisheries is needed. In particular the size and growth of salmon and rainbow trout caught by anglers during November should be monitored to determine if the recent declines in size continue.

Finally, strategic research is needed on lake ecosystems to understand the effects of hydro-power development on lakes similar to Coleridge. This includes research on:

1. Factors such as lake level fluctuations and turbidity which could influence benthic invertebrate distribution, abundance and productivity in the littoral zone.

2. The role of common bullies and koaro in lake ecosystems.

3. New methods of monitoring salmon and trout stocks in lakes.

4. The life history, migrations, distribution and growth of landlocked chinook salmon in New Zealand lakes.

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